United States Environmental Protection Agency

EPA/600/R-09/112 | September 2009 | www.epa.gov /nrmrl

Advanced Topics in Wet-Weather Discharge Control

Office of Research and Development
National Risk Management Research Laboratory - Water Supply and Water Resources Division

EPA/600/R-09/112
September, 2009

Advanced Topics in Wet-Weather Discharge Control

by

A. Charles Rowney
ACR, LLC
Longwood, Florida, 32750

and

Linda D. Pechacek
LDP Consultants, Inc.
Houston, Texas, 77018

in association with

Michael E. Hulley,
Hulley Holdings, Inc.
Kingston, Ontario

Thomas P. O'Connor,
U.S. EPA
Edison, New Jersey, 08837

and

Larry A. Roesner
Colorado State University
Fort Collins, Colorado, 80523

Contract No. 8C-R056NTSX

Project Officer
Thomas P. O'Connor

Technical Advisor
Richard Field
Urban Watershed Management Branch
Water Supply & Water Resources Division
National Risk Management Research Laboratory
Edison, NJ 08837

NATIONAL RISK MANAGEMENT RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CINCINNATI, OH 45268

Notice

The U.S. Environmental Protection Agency (EPA) through its Office of Research and Development (ORD) performed and managed the research described here. It has been subjected to the Agency's peer and administrative review and has been approved for publication as an EPA document. Any opinions expressed in this report are those of the author and do not, necessarily, reflect the official positions and policies of the EPA. Any mention of products or trade names does not constitute recommendation for use by the EPA.

Abstract

This report discusses four related but generally independent wet-weather flow (WWF) topic areas, namely: i) opportunities for advanced practices in WWF control technology, particularly as it applies to sewered systems; ii) tradeoffs between storage facilities (tanks) and enlarged trunk sewers (tunnels) in CSO control; iii) disinfection/sedimentation tradeoffs in primary treatment; and iv) routing methods for indicator bacteria analysis in stormwater. The literature surrounding these areas is reviewed, and then each of the four areas is developed as a separate sub-theme of the report. An evaluation of advanced practices identifies seven areas where knowledge gaps are likely to frustrate current practices, and also explores some potential development and innovation areas that offer promise in the near future for improvements in practice. Some key areas that are candidates for development include evaluation of BMP placement and performance, real time control, information management, decision support systems, and control system theory and application in the watershed context. The analysis of oversized tunnels as alternatives to storage tanks in the CSO control context is made, and conclusions are drawn as to the relative merits and potential cost tradeoffs between these options, based on available data. The report also evaluates the relationships and tradeoffs between disinfection and sedimentation in primary treatment of wastewater. The combination of these two methods is explored, and the inverse relationship between sedimentation efficacy and the need to add disinfectant is assessed. A systems model was developed to represent these two processes. Based on typical data, the model was applied to a sedimentation system and a disinfectant system individually, and then in various combinations. The model was found to be a useful and simple way to simulate sedimentation and disinfection design alternatives. The complexities of bacterial behavior in the environment, including substrate adhesion and clumping, are discussed. The results of modeling and analysis are assessed and the potential for improvements in practice is addressed. The scale dependence of BMPs and relevance to indicator bacteria controls is assessed by means of a case study. From this effort, conclusions were drawn that certain end-of-pipe BMP treatment and control may be of limited value for control of indicator organisms, and should be avoided in favor of site-specific studies of cause, effect and in-stream water quality. Also significant is the conclusion that ponds are sensitive to mixing and antecedent conditions, and that drawing down ponds via filtration or exfiltration between events may be a factor in enhancing pond performance, particularly in small events. Recommendations for future research are made.

Foreword

The U.S. Environmental Protection Agency (EPA) is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how pollutants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory (NRMRL) is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threaten human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by EPA's Office of Research and Development to assist the user community and to link researchers with their clients.

Sally C. Gutierrez, Director
National Risk Management Research Laboratory

Contents

List of Figures

Acronyms and Abbreviations

ANN	= Artificial Neural Network
BMP	= Best Management Practice
BOD	= Biochemical Oxygen Demand
COD	= Chemical Oxygen Demand
CSO	= Combined Sewer Overflow
CSS	= Combined Sewer System
CWA	= Clean Water Act
DO	= Dissolved Oxygen
DSS	= Decision Support System
DWF	= Dry-weather Flows
E. coli	= *Escherichia coli*
EPA	= United States Environmental Protection Agency
EU	= European Union
FC	= Fecal Coliforms
FWPCA	= Federal Water Pollution Control Act
HSPF	= Hydrological Simulation Program-Fortran
LID	= Low Impact Development
LTCP	= Long-term Control Plan
MEP	= Maximum Extent Practicable
MS4	= Municipal Separate Storm Sewer System
MSBM	= Multiphase Sediment/Bacteria Model
NPDES	= National Pollutant Discharge Elimination System
NURP	= Nationwide Urban Runoff Program
ORD	= Office of Research and Development
POTW	= Publicly Owned Treatment Works
RBC	= Rotating Biological Contactors
RTC	= Real-time Control
RO	= Reverse Osmosis
SCADA	= Supervisory Control and Data Acquisition
SM	= Standard Methods
SS	= Suspended Solids
SSO	= Sanitary Sewer Overflow
TC	= Total Coliforms
TMDL	= Total Maximum Daily Load
WQS	= Water Quality Standards
WWF	= Wet-weather Flows
WWTP	= Wastewater Treatment Plant
UK	= United Kingdom
U.S.	= United States
USGS	= U.S. Geological Survey
UV	= Ultraviolet

Acknowledgements

An undertaking of this type requires the dedication and cooperation of a team. The technical direction and coordination for this project was provided by the technical project team of the Urban Watershed Management Branch, under the direction of Mr. Thomas P. O'Connor, the Project Officer and Mr. Richard Field, the Technical Advisor. Special recognition is also extended to Mr. Anthony Tafuri, P.E., Branch Chief who provided technical and managerial guidance at key points. Also acknowledged is the support by Dr. Scott Struck of TetraTech, and Mr. Christopher Rowney, Research Scientist, of ACR, LLC, who provided literature search material and critical reviews to the project and by Mr. Jon Jones and Ms. Jane Clary of Wright Water Engineers who reviewed aspects of the work related to the Best Management Practice Database. Dr. Dennis Lai and Dr. Ariamalar Selvakumar of EPA performed reviews of this report and Ms. Carolyn Esposito of EPA reviewed the quality assurance project plan and the report as well. Finally, the manifold contributions of the many authors and professionals who were contacted or cited in this work are acknowledged, as it is their efforts that underlay the discussion and advances contained in the report.

Executive Summary

Control of wet-weather discharges to improve receiving water quality remains an elusive goal with the current state of practice. This report addresses advanced concepts in several broad topic areas to advance the knowledge of wet-weather control management. The four topic areas addressed are: i) wet-weather flow (WWF) technology and management development; ii) contrasting tank and in-line storage solutions; iii) sedimentation and disinfection tradeoffs in primary treatment; and iv) best management practice (BMP) pathogen removal and routing analysis.

The issues surrounding these topic areas are summarized below:

- National policy in water quality management, notably as embodied in total maximum daily load (TMDL) requirements, is increasingly putting pressure on municipalities to reduce polluted discharges to receiving waters. The cost implications of responding to this pressure are substantial, and the history of combined sewer overflow (CSO), sanitary sewer overflow (SSO) and stormwater management practices has left a legacy of structures "in the ground" that now limit options for remediation. A review of options for improved mechanisms for treatment is needed to determine if emerging technologies or practices, including international progress, might provide insights into alternatives not presently being considered. This work evaluates this potential and focuses on identifying those technologies that are not in common practice but are beyond the point of basic research.

- CSO control in the form of storage (tanks) and pipeline (tunnels) storage tradeoffs was a particular element of this review that was readily accomplished because the available literature on this subject is extensive. Coping with the variability in flow associated with CSOs is a defining requirement of CSO control. The need to cope with flows during storm periods has often driven designs towards combined sewers that were sized far larger than necessary to convey peak dry-weather flows (DWF). This leads to low flows and deposition of wastewater solids during DWF, and resuspension and flushing of deposited materials during WWF. Questions arise as to ways the elevated constituent loads during WWF can be remedied. One potential solution of interest is the use of large storage facilities (tanks) or enlarged trunk sewers (tunnels) to contain flows, and thereby contribute to balancing flows and developing more efficient and steady treatment strategies. Crucial to understanding the economies of this section is that the premise for this analysis setting is for large-scale developments and projects. The suitability of off-line and in-line processes in this context is also addressed.

- EPA's national CSO policy requires primary treatment plus disinfection, depending on state and policy context resulting in varying implementation of these technologies nationwide. Despite the fact that each element is well understood, the ways in which these two common technologies interact when used jointly have yet to be fully understood. The interdependence between these processes is significant, and the two are a net determinant of overall efficiency and cost in treatment facilities. Increasing pressures to deliver improved receiving water quality, at a time when implementation and operational funds are diminishing and must increasingly compete with other national interests, makes a deeper understanding of performance and cost containment related to these factors advisable. To avoid redundancy and duplication of prior work, the present focus was not on basic research, but was instead focused on gathering existing information, building on the understanding of unit processes, and exploring their dynamic interactions with modeling or interpretation.

- Pathogen routing remains an important topic, despite the fact that the basic issues revolving around this problem are understood. Indicator bacteria have merit as a way of discriminating between waters that are likely to be safe and those that might not be, but indicator bacteria in the environment are very highly variable, and are produced by numerous sources. It is, and will be important for some time to come, to have a method of coping with the variable nature of source, transport and presence of indicator bacteria in evaluating BMPs as control alternatives. The nature of bacteria in the environment is such that BMPs can themselves constitute sources of indicator organisms. It is known that under some conditions, indicator bacteria can survive for protracted periods in sediments over time scales that would commonly span time periods between storms; wildlife associated with BMPs can also be indicator sources. There remains a need to explore whether or not BMPs are in reality practical as solutions to the removal of indicator bacteria. This portion of the project, targeted at ways of evaluating indicator bacteria movement and prediction, attempts to present modeling that may help dealing with these issues.

The realities of microbiology underlay much of this work and appear in many contexts. Disinfection is a function of the dose and duration with which a medium is exposed to a particular disinfectant, but the factors that govern the rate and degree of disinfection are numerous. Measurement in the environment is a routine and important part of environmental monitoring, but many factors make it difficult to accurately gauge the number and type of indicator bacteria and pathogens in the environment. Other examples can be cited, and what they have in common is the complexity of the microbiological processes involved. Adhesion to surfaces, embedding in a matrix, stress responses of the indicator microorganisms, and recovery of those organisms from their stressed state all work to reduce the degree of disinfection. The survival, re-growth, re-emergence or recovery of those organisms counters the intent and efficiency of disinfectant addition. Accounting for these realities explicitly offers a potential that disinfection practices can be adjusted to remove target microorganisms more effectively.

In the present project, a four compartment model (bacteria, disinfectant, and fluid and solid phases) was developed and tested for function using arbitrary but representative input parameters, and has been shown to function well in comparison to existing data sets. The model has potential applications in real-world contexts, but modeling of this type will be limited until comprehensive site-specific data become available. Although limited in scope, the model testing done in this project underscored the importance of suspended solids (SS) removal rates on the effectiveness of chlorine disinfection. From an operational perspective, achieving a desired level of bacteria removal or treatment efficiency requires an understanding of the extent to which bacteria are removed through solids settling.

Future research into this area could be taken in a number of useful directions. Experimental investigations in this work focused on quantification of the relationships between bacteria removal and a single SS component. Extending the model to represent a wider range of settling models, kinetic phenomena and multiple sediment fractions would be useful and extend the value of the model significantly. Similarly, the concurrent evaluation of multiple microorganisms (indicators and pathogens) would be useful to provide insights into the best way to apply these results in practice. If pathogenicity cannot be strongly related to indicator removal, it may be that gross methods of determining disinfection and sedimentation are all that can be reasonably defended.

Potential practical applications of this tool are evident. The model could be used, with better data and a more extensive testing program, to establish definitive design rules for disinfection of stormwater. Guidance on ways to determine the optimum removal process could have a very significant cost implication at a national level. Given the economic significance of this area (large numbers of treatment facilities rely on sedimentation and disinfection to at least some degree nationwide) this area of research should be pursued further. It is noted that there are implications in stormwater management as well. This model, though not fully detailed in this project, could in principle be applied in stormwater contexts where settling and disinfection are applied, and this consideration merits specific future attention.

Water quality sampling is also an area of interest. It seems clear, regardless of the ways that modeling might be approached, that present practices in indicator bacteria sampling may need to be revised. The variability of the phenomenon, and the ways that mixing, treatment and transport affect indicators, combine to make it difficult to readily measure the impacts of surface water sources, BMP impacts, stormwater or CSO discharges, even though they may be very significant. A few grab samples taken without regard for the process behavior of the BMP are likely to have no meaning in terms of the inflow/outflow transformations caused by the BMP. Further research will be needed to develop a confident statement as to the preferred approach to sampling and analyzing BMP performance. It is becoming clear that monitoring requires an intensive effort. Dozens of samples per storm event, taken in a synchronized way (matching inflows and outflows according to BMP residence time), are required if a dependable matched pair of inflow versus outflow characteristics are to be developed. Describing the behavior of the BMP overall will take dozens of such monitored events under varying conditions (e.g., precipitation, temperature, season, antecedent moisture conditions). This means that it will take perhaps hundreds of samples to verify the performance of a single BMP.

If a pollutant indicator cannot be measured or will not usually be measured in practical contexts to a degree of accuracy that is meaningful, it is fair to question how meaningful the indicator is. One characteristic of an ideal indicator is that it is tractable from a sampling point of view. Present indications are that the suite of pathogenic indicators in use today may not be tractable. This in turn puts further questions on the validity of the notion of indicator bacteria sampling for monitoring performance of BMP removal given that many BMPs in question have typically been designed for removal of another class of pollutants.

This does not mean that all contaminants cannot be measured with statistical precision. On the contrary, sediments, for example, turn out to be quite tractable. There are problems associated with predicting sediments, but they are more amenable to measurement and can be inferred from sediment indicators such as turbidity. The literature review and modeling results strongly suggest that the behavior of indicator bacteria can be inferred from the behavior of sediments. Since sediments are cheaper and quicker to evaluate, it may be the case that a monitoring program based on sediments, or by proxy turbidity, will be more effective in estimating bacterial behavior than a measurement program targeting the indicators directly.

Indicator bacteria themselves are, by definition, not the issue; these microorganisms are measured because of the association between them and other microorganisms with disease potential. Since BMP performance on pathogenic indicators is difficult to deal with, it may be that addressing pathogens by means of a secondary indicator is a valid option. A hybrid program can be envisioned, in which sediments are measured to evaluate BMP performance, and those results used to estimate the impact of that level of performance on indicator bacteria removal. To provide an empirical basis for this interpretation, it would be useful if bacteria are measured along with sediments, in enough detail to develop a relationship between indicators and sediments at a particular site. This would provide a locally meaningful determination that inherently incorporates some of the factors such as soil type and water chemistry, which can have an effect on bacterial behavior.

An evaluation of the cost implications of tunnels vs. tank storage was undertaken. The principles involved in tunnel storage and tank storage are generally understood, but blanket assertions as to the relative performance of BMPs are not possible given the various factors that affect performance at any particular site. Cost curves for installation may not fully reflect all site conditions, but are nevertheless useful in interpreting the relative costs of the two approaches. The general trend that was identified appears to be that on-line conveyance/storage tunnels are economically preferred to tank storage in undeveloped placements. More generally, it appears that tunneling cost for off-line systems will tend to be higher, while tunneling in on-line systems will tend to be lower. The specifics of a particular site can reverse these trends, but available information endorses this general approximation of expected behavior.

Therefore, it appears the principal driver for selection of tunnel storage over tank storage is the ability to preserve future flexibility. An over-sized tunnel can be electively used for storage, conveyance or even both, simply by changing operating characteristics. Obviously, it is more beneficial if tunnels are installed at the onset of the construction process. Whenever placed, an oversized tunnel will tend to enable greater latitude in capture rates, implying an ability to capture, control and move higher multiples of average WWF. Therefore, the preferred course from the perspective of preserving future flexibility of function for the facility is the on-line storage/conveyance tunnel. A tank is a useful remedial alternative, but from at least this perspective, is not the preferred solution.

Generally, a variety of other approaches to CSO management were identified that have merit for further investigation and/or immediate implementation. Targets other than flow rate and elementary water quality constituents should be considered, as well as the wider environmental impacts of water infrastructure solutions, including things like energy consumption and ecological community response. So called “green technologies” in particular merit wider review by a range of disciplines, and strategies for implementation should include perspectives from other fields on a routine basis.

Chapter 1 Introduction

This report documents the results of four essentially separate but integrated topic areas. The topics are thematically linked to advancing the knowledge of wet-weather flow (WWF) control. This integration has significant advantages in terms of consolidating and relating results; however, when reading the report, it should be understood that it covers four topic areas that differ in focus. To provide a logically framed single report, but still respect the integrity of the four topic areas, a specific approach to assembling the report was adopted. Structurally, the report begins with a statement of the background and objectives (this chapter). Conclusions and recommendations provide consolidated results and outcomes of the work. It proceeds to a review of fundamental regulatory factors. This is followed by separate chapters that deal with literature reviews and analysis outcomes of each specific topic area. The exception to this format is Topic Area 1 which is only represented by results of the literature review. Topic Area 1 is the fundamental theme of the report, transcending the other technically- and analytically-orientated Topic Areas.

Topic areas and objectives are described below.

Topic Area 1: Wet-weather Flow Technology and Management Development

Overview

National policy in water quality management, notably as embodied in total maximum daily load (TMDL) requirements, is increasingly putting pressure on municipalities to reduce contaminated discharges to receiving waters. The cost implications of responding to this pressure are substantial. This response has also been complicated by the history of combined sewer overflow (CSO), sanitary sewer over (SSO) and stormwater management practices which has left a legacy of structures "in the ground" that now constrain future operational management options. EPA concluded that a review of options for improved mechanisms for treatment is needed, since there is now enough pressure to remove nutrients and other contaminants from the total waste stream. Concerns have been raised that traditional approaches to stormwater treatment may be inadequate if discharge goals are to be met. This raises the question as to what alternatives are avaiable and also what experience elsewhere might shed light on this question.

Innovations may provide options that traditional methods in the North American context may not. This work focuses on those technologies that are not in common practice but are beyond the point of basic research. Modifications and advancements to existing technologies are not part of this evaluation. Innovations in application approaches that are qualitatively different than past practice, however, might be of interest. For example, the notion of an infiltrating best management practice (BMP) for volume control is not new, but testing the feasibility of infiltration as a pathogenic indicator control mechanism was given specific consideration in this report.

Overall, the identification of new technologies proved to be a challenge because a wide range of technologies is already commonly considered or employed to control discharges. The fundamental technologies are well known, and include source controls, inflow controls, optimization methods (real-time control [RTC], storing combined sewage in existing sewers, or revising facility operations), sewer separation, improved treatment technologies, and in situ remediation such as may be accomplished by aeration and flow augmentation. Each technology has differing potential for success when considered from the perspectives of regulatory compliance, cost effectiveness, remedial efficacy, public acceptance, collateral impact, and other factors. These technologies are relatively well understood for the most part, but can be implemented in different ways or augmented by new approaches, such as storage of all forms of WWF as it accumulates with bleed-back (gravity flow) to a wastewater treatment plant (WWTP), or implementation of high-rate treatment methods, initially designed for CSO or SSO, for use with stormwater.

Complicating the interpretation of opportunities is the plethora of gray literature claims by manufacturers and sources with uncertain quality control; a recent systematic review of options in this area is not available. A review of practice in terms of what has been accomplished, including international efforts and results, i.e., the recent experiences of the European Union (EU), and the massive effort into analysis on watershed management options, is presented. The EU Water Framework Directive (more formally the Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000) commits EU member states to improve water quantity and quality by 2015.

Objective

The objective of this task was to scan the available literature and, to the extent possible, filter out standard technical solutions in the U.S. and EU so that innovative potential and emerging solutions can be identified, evaluated, and communicated.

Topic Area 2: Contrasting Tank and In-line Storage Solutions

Overview

TMDLs are driving management efforts to improve contaminant control in all elements of WWF management; CSOs are encompassed by this process. Loads from CSOs can be very substantial, but are amenable to management in a variety of ways. Management of a combined sewer system (CSS) is complicated by the highly dynamic nature of flows and loads, ranging from dry-weather flows (DWF) that vary daily, weekly and seasonally, and peak period overflows that vary arbitrarily according to rainfall. Coping with this variability is an inherent requirement of CSO control. An option particularly suitable in new urban areas is the incorporation of oversized sewer pipes (tunnels) in combination with treatment to provide a buffering volume and CSO treatment capacity. This notion gave rise to the present research.

CSO control mechanisms can include reducing inflow volumes that cause overflows, increasing storage to buffer high-rate inflows, incorporating storage to buffer high-rate outflows, using high-rate treatment, or storage for subsequent treatment at the WWTP. A concept pertinent to CSO management is that combined sewer solids are deposited during DWF periods and resuspended during WWF periods at some later time. The CSS hydraulic design is one of the root causes for this phenomenon. Because of the need to cope with flows during storm periods, combined sewers are sized far larger than necessary to convey peak dry-weather wastewater flows. Therefore, during dry-weather velocities, shear forces are too low to carry all of the suspended solids (SS), and the amounts that cannot be transported simply settle. Then, during WWF, shear forces from increased velocities are enough to mobilize deposited sediments and carry them to a discharge point where they are observed as *first flushes*[1] that persist until the deposited load is removed. The classically conceived first flush is only imperfectly observed in some situations, but the basic principle is that this phenomenon (deposition and resuspension) is a clear contributor to problematic CSO performance, not only including the sediment deposit/flush cycle, but also gas generation as well. CSO SS loadings generated during storm periods vary in proportion to annual dry-weather wastewater loadings, but have been recorded at a comparable order of magnitude. This implies that a substantial part of these untreated discharges occurs over a short period of time. It is therefore not surprising that SS concentrations of several thousand milligrams per liter (mg/l) can be encountered at discharge points. Clearly, introducing mechanisms that address this phenomenon is of

[1] This is a segment of the pollutograph that exhibit a greater degree of contamination, particularly but not exclusively, early in the runoff period.

interest because of the potential to moderate transport phenomena that exacerbate the dynamic range of flows and sediment loads in the sewers, and overflows and loadings to the receiving waters. Traditionally, combined systems were constructed at a low multiple to DWF conveyance capacity with little or no upstream controls, and many WWTP were constructed for DWF capacity with bypass during WWF events, so overflows were a common outcome of WWF events which is why the National CSO policy calls for maximization of the collection systems for storage. The question of oversized sewers is a specific issue that merits further investigation.

Objective

This topic explored several methods, that historically may have been given less exposure due to overriding concerns with WWTP operations and end-of-pipe treatment options, to generate a better understanding of how a variety of storage options can potentially provide useful solutions to managers tasked with controlling CSO system discharges.

Topic Area 3: Sedimentation and Disinfection Tradeoffs in Primary Treatment

Overview

EPA's national CSO policy requires maximizing the flow to the WWTP. Often, this shortens the detention time in primary treatment, leads to bypass of secondary treatment, and increases solid content of waters subject to disinfection. There is a very substantial literature that deals with both primary treatment and disinfection technology, and in fact they are so commonly dealt with that the basic literature will not be described or addressed in detail here. It has been noted, however, that the ways in which these two common technologies interact when used jointly has yet to be fully understood. They are known to be interdependent physically but current engineering practice, and U.S. policy and regulation do not account for that fact.

This interdependence is potentially significant because the interactions between the two are a net determinant of treatment efficiency and ultimately cost. For example, conventional disinfection best deals with free microorganisms and is not as effective when there are particles and other matter in the water (Perdek and Borst, 2000a and 2000b). This physical/chemical interaction in turn raises the question of the total amount of bacterial biomass exported from the treatment system. Driving the degree to which microorganisms are freely available in the fluid phase can change the net efficiency of disinfection. Increasing pressures to deliver improved receiving water quality at a time when implementation and operational funds are not inexhaustible and must increasingly compete with other national interests makes a deeper understanding of performance and cost containment related to these factors advisable. Given the tonnages of chemical disinfectants used annually nationwide, and the focus on aging infrastructure that currently prevails, beneficial results of improved practice in these root areas of wastewater treatment could be considerable in terms of economic impact and receiving water protection.

From a technical perspective, although the general interactions between these treatment processes are known, there is still a further need for exploration and development of a procedure to balance the extent of one versus the other in order to optimize the system's performance versus cost. To avoid redundancy and duplication of prior work, an achievable subset of the possible avenues of research was sought as this project was developed. It was considered that the governing physical and chemical equations for sedimentation and disinfection are well documented in a plethora of engineering manuals and wastewater textbooks (e.g., Tchobonoglous et al., 2003), some of which EPA has been instrumental in developing and promulgating at a national level. Therefore, the focus was on gathering existing information, building on the understanding of unit processes and exploring their dynamic interactions with modeling or interpretation.

The interactions of unit process elements in combination inherently involve analytical complexity greater than the functions of the independent process elements themselves, because the total system encompasses not only those discrete processes, but also the net system consequences and interactions as well. The benefits of improved computational capability made this mathematical complexity a reasonable target for investigation. An objective which follows is the development of a conceptual basis for this modeling. Discussions of this basis can be found in Rowney et al. (2008), which although superseded, forms a background document to this one and contains some of the early results reproduced in this report.

Simulation of process reactors and fluid transport mechanisms are generally well understood and will not be addressed further. In contrast, microbiological aspects are highly complex and even now are only partially resolved; choices must be made as to how these might be represented so that meaningful model results can be obtained. A specific example of this may be found in the interrelationships between primary treatment and disinfection efficacy. Conventional disinfection best deals with free (unattached) microorganisms and is generally not effective for microorganisms contained within larger protective solids. Therefore, primary removal which emphasizes elimination of larger particulates will tend to have a proportionately larger impact on disinfection efficiency. Primary removal and disinfection interact, and a question of which is relevant to current national policy requires a basis to establish what tradeoffs can be made between one function and the other to achieve a better net result.

Objective

Based on currently known technical principles, a modeling approach was developed to evaluate tradeoffs in the selection of primary treatment and disinfection alternatives in combination and to provide improved understanding of ways to cost effectively achieve regulatory compliance. The immediate intent was to determine:

i. Those elements of the system that are most critical in terms of information gaps, so that further research can be considered to enhance model reliability, and

ii. The apparent potential for changes in process train management in the primary and disinfection system components to achieve economies in operation and potentially develop an optimum response strategy.

Topic Area 4: Best Management Practice Pathogen Removal and Routing Analysis

Overview

This topic area was designed to support development of innovative urban technologies that might assist municipalities and utilities in the selection of appropriate technologies to control urban discharges to surface water, especially those caused by WWF or failing infrastructure and cross connections. The focus of this particular topic was to explore or develop a method by which indictor organisms can be evaluated, to determine what options exist for the management of indicator organisms, and to estimate what the efficacy of those options might be in a real-world setting.

The basic issues revolving around this problem are understood. Indicator bacteria have merit as a way of discriminating between waters that are likely to be safe, and those that might not be. Furthermore, it is known that there are technologies enabling the prediction of the general behavior of indicator bacteria in the environment. What is problematic is that indicator bacteria in the environment are very highly variable, and are produced by numerous sources other than just contamination by domestic wastewaters, which was the primary reason for adoption of indicators in the first place. These factors make it difficult to associate cause and effect, and consequently to interpret receiving water conditions either in terms of the reasons for what is observed, or in terms of how to improve that condition. Substantial advances in interpreting indictor bacteria sources ("source tacking") have been made in recent years (e.g., ribotyping or "genetic fingerprinting") and some of these have the potential to improve the state of the art in indicator bacteria interpretation and control. Even so, there are numerous issues that will only slowly be resolved despite the fact that better methods of detection and source inference may be available. First among these issues is the development of conclusive relationships between pathogenicity and indicator bacteria according to indicator source, whether the source is a result of human sanitary waste emissions or not. Added to that are the realities associated with implementing a change in the state of practice, given the existence of allied fields such as medical, health and safety that deal with responses to indicator bacteria found in our waters. It is important, and will continue so for some time to come, to have a method of coping with the variable nature of source, transport and presence of indicator bacteria in evaluating BMPs as control alternatives.

BMPs themselves are commonly proposed as solutions to indicator bacteria water quality problems among other pollutants or stressors. This has merit, but it is an imperfect solution. The nature of bacteria in the environment is such that BMPs can themselves constitute sources or perceived sources of indictor organisms. This circumstance can occur when wildlife use the BMP as habitat (e.g., waterfowl are notable generators of indicator bacteria). A second complicating effect can occur if the BMP is designed such that sediments captured in one event are resuspended by inflows from the next. It is known that under some conditions, indicator bacteria can survive for protracted periods in sediments over time scales that would commonly span inter-event time periods between storms. This means that apparent removal from one event is in reality partly only a transfer of loads to the next. This kind of effect is strongly impacted by BMP design as it governs mixing behavior. Even if neither of the above problems is an issue, BMPs dependant on settling may be only partly effective in reducing bacteria since substantial proportions of the indicator load is not readily removed by this mechanism. Different BMP types are prone to different problems, but the salient point is that there are many reasons the full suite of BMPs may be imperfectly or even poorly effective in removing indicator bacteria. There remains a need to explore whether or not BMPs in general or only specific BMPs are in reality practical as solutions to the removal of indicator bacteria. If so, BMPs may remain viable solutions from that perspective, perhaps amenable to enhanced design methods. If not, it may be necessary to concede that more active disinfection technologies should be considered.

Objective

An approach to simulating indicator bacteria transport that incorporates some of the physical uncertainties that affect this phenomenon is presented. The intent was to develop a computer simulation tool, based on a systematic evaluation of characteristic parameters, and evaluate the probable effectiveness of BMPs for pathogen removal under a range of potential physical conditions.

Chapter 2 Conclusions

Several advanced concepts related to wet-weather technologies in water and wastewater management have been explored, and conclusions have been drawn from the results of that exploration. Tunnel and storage technology tradeoffs in relation to CSO discharge management have been investigated, bacterial behavior in the environment has been researched in the literature, models of bacteria fate and transport developed and tested, and the literature and members of the relevant community canvassed for insights into emerging directions in research and technology application. Detailed conclusions are found in the main body of this report in the sections dealing with these topic areas, but the main results of this work are listed below.

The principles involved in tunnel storage and tank storage are generally understood, but blanket assertions as to the relative performance of these devices are not possible given the various factors that affect performance in any particular site, and given the still limited state of knowledge regarding some basic phenomena governing performance in these systems. Cost curves for installation do not fully reflect all site conditions, but are useful in interpreting the relative costs of the two approaches. An analysis was carried out using available information; the general trend identified appears to be that on-line conveyance/storage tunnels are economically preferred to tank storage in undeveloped placements. More generally, it appears that for off-line systems, tunneling will lead to a higher relative cost, while on-line systems tunneling will tend to a lower cost. The specifics of a particular site can reverse these trends, but available information makes them a reasonable general approximation of expected behavior.

Given the state of knowledge that was found in this topic area, it is concluded that the principal driver for selection of tunnels over storage appears to be the ability to preserve future flexibility. If an over-sized tunnel is placed, it can be electively used for storage, conveyance or both by changing operating characteristics. Tunnels are best placed before a development is completed. Trying to construct such a tunnel after development implies dealing with numerous interconnections and constraints that greatly exacerbate the problems inherent in placing such a structure. Whenever constructed, an oversized tunnel tends to enable greater latitude in capture rates, implying an ability to capture and control higher multiples of average WWF. It is therefore concluded that the preferred course from the perspective of future-proofing the facility is the on-line storage/conveyance tunnel. A tank is a useful remedial alternative, but from at least this perspective is not the preferred solution.

The microbiological response to primary settling followed by disinfection is simple in concept but highly complicated in detail. The basic notions of indicator bacteria responses to environmental stressors are well understood, but quantifying them to the point where meaningful predictive models can be applied is in practice only applied to a limited degree. Disinfection of wastewater is a function of the dose, mixing and duration but there are numerous other factors that govern the rate and completion of disinfection. Adhesion to surfaces, embedding in a matrix, stress responses of the indicator organisms, and recovery of those organisms from their stressed state all work to reduce the degree of disinfection. The survival, re-growth, re-emergence or recovery of those organisms counters the intent and efficiency of disinfection. Accounting for these realities explicitly offers a potential that disinfection practices can be adjusted more effectively to remove target organisms.

Research is needed to better determine and quantify the factors in this area and to ultimately enable better predictive capability. At present, a four compartment model (bacteria, disinfectant, fluid and solid phases) was considered justifiable given the information available. A model representing this case was developed and tested. Since suitable calibration data is lacking, this model was tested for function using arbitrary but representative input parameters, and shown to function well. The model has potential applications in real-world contexts, but modeling of this type will be limited until comprehensive site-specific data become available.

Rate constants and an agreed description of bacterial population dynamics in this context need to be developed for major advances in this kind of modeling to occur. A review of the literature and interviews with experts in the field suggests that the reversibility of the bacterial attachment reaction is unproven, and that confident estimates of rate constants cannot be made without physical experimentation.

Although limited in scope, the model testing done in this project underscored the importance of SS removal rates on the effectiveness of chlorine disinfection. From an operational perspective, achieving a desired level of bacteria removal or treatment efficiency requires an understanding of the extent to which bacteria are removed through solids settling.

Future research into this area could be taken in a number of useful directions. Experimental investigations in this work focused on quantification of the relationships between bacteria removal and a single SS component. Extending the model to represent a broader range of settling theory, kinetic phenomena and multiple sediment fractions would be useful and could improve the value of the model significantly. Similarly, the concurrent evaluation of multiple microorganisms (indicators and pathogens) would be useful to provide insights into the best way to apply these results in practice. If pathogenicity cannot be strongly related to indicator removal, it may be that current, gross methods of determining disinfection dosage and sedimentation rates are all that can be reasonably defended.

Another development direction therefore would be more rigorous model runs and formal statistical analysis to establish definitive design rules for the disinfection of WWF. Guidance on ways to determine the optimum removal process could have a very significant cost implication at a national level. The model would require specific data inputs, but given the economic significance of this area of research, and the large numbers of treatment facilities that rely on sedimentation and disinfection to at least some degree nationwide, it should be pursued further.

Although not directly within the formal scope of the work performed, it is noted that there are implications in stormwater management as well. This model could in principle be applied in stormwater contexts where settling and disinfection are applied; this consideration merits specific future attention.

Despite the preliminary nature of the data, it seems apparent that solids settling rates and chlorine dose for disinfection purposes should be addressed in combination in order to more reliably estimate disinfection efficiencies.

It is becoming apparent that regardless of the ways that modeling might be approached, present practices in indicator bacteria sampling to demonstrate BMP efficacy may need to be revised. The variability of the phenomenon, and the ways that BMPs affect indicators mixing and transport, combine to make it difficult to readily measure BMP impacts, even though treatment may be significant. Statistically based sampling is required to prove BMP indicator removal performance but few are prepared or can afford such a sampling program which currently seems to be required on a case-by-case basis to develop a track record of BMP performance with regard to indicator removal.

A characteristic of an ideal indicator is that it is tractable from a sampling point of view. Present indications are that the current pathogenic indicators are less than ideal in this regard. This in turn puts further questions on the validity of indicator bacteria sampling. On the other hand, sediments turn out to be more tractable in contrast. The results of modeling and of the literature review are strongly suggestive that some aspects of the behavior of indicator bacteria can be inferred from the monitoring of sediments. Since sediments are cheaper and quicker to evaluate, it may be the case that a monitoring program based on sediments, or by extension to turbidity, might be more effective in estimating bacterial behavior than a measurement program targeting only indicator organisms directly.

A hybrid program in which sediments are measured to evaluate BMP performance and to estimate the impact of that level of performance on indicator removal could be developed. To provide an empirical basis, indicator bacteria could be measured along with sediments in enough detail to develop a relationship between indicators and sediments at a particular site. This would provide a local empirical determination that inherently incorporates some of the factors such as soil type and water chemistry that can have an effect on bacterial behavior.

Analysis indicates that there is reason to expect that ponds which operate either as a filter or as a discharge to groundwater through the bottom may be a more efficient alternative, particularly for smaller more frequent events, than ponds that attempt to detain volumes temporarily or retain volume for as long as possible, especially for indicator organisms, or by extension, dissolved constituents. Water quality managers need to weigh the benefits of varying goals for water quality treatment of the BMPs placed in a watershed. Pathogenic indicator organisms are only one water quality indicator. Use of multiple BMPs with different hydraulic functions, and therefore water quality and quantity control functions, may be warranted. The use of a treatment train approach has been proposed to address multiple stressors like solids, nutrients and pathogenic indicators.

A variety of approaches to CSO management were identified that have merit for further investigation and/or immediate implementation. It is concluded that targets other than flow rate and elementary water quality constituents should be considered. The wider environmental impacts of solutions, including things like energy consumption and ecological community response, should be considered. Green technologies in particular merit wider review by a range of disciplines, and strategies for implementation should include perspectives from sociology and other fields on a routine basis.

There are also a range of technical solutions that have been identified as worthy of future consideration. Developments in materials and information technology offer increased opportunities for system management. It is now possible to implement control systems that will enable adaptive management of systems in real time. Leak detection could be much earlier, and problem determination and response could be much more proactive. This in turn would translate into fewer, smaller and less destructive discharge events. Some specific technologies that merit further development include intelligent, real-time quality based systems, and virtual management asset systems.

Another possible development is optimizing the way that current computer models approach WWF management. Some of the analytical technologies that are routinely deployed at present involve principles that are hydraulically detailed but that do not lend themselves to preservation and management of water balance as a primary principle. Unfortunately, an excellent knowledge of the details of hydraulic losses at pipe junctions does not by itself lead to an excellence in the management of water in the system. There is a need to promote the use of existing and new tools that put priority on the water balance, including groundwater and ecosystem responses, along with hydraulic evaluation. This is a high priority need because it underlies a preponderance of the solutions being considered at this time in water resources engineering.

More generally, the review and testing of in-stream indicator bacteria behavior, based on the models and data used in this work, offered a strong indication that end-of-pipe ponding controls of bacteria may be of value but the design of those facilities needs to take event sequence and the known processes of removal into account for best results. The ubiquitous nature of these organisms in the environment, and the time scale over which they are active, are such that substantial investments in BMP placement and sizing to target this pollutant may be questionable if removal process mechanisms are not respected as a BMP design factor. Case by case evaluation of the circumstances that prevail at each site appears to be an important success factor. Finally, it is noted that the notion of scale and sequence dependence that is introduced by this concept is applicable to other contaminants, and should be considered. Setting anthropogenic end-of-pipe requirements without supporting in-stream impact analysis is likely to be an expensive and ineffective management approach given the multiple natural sources of indicator organisms.

Chapter 3 Recommendations for Potential Innovation Areas

One intended outcome of this project was to consider which technologies might provide the best avenues forward to meet emerging challenges. Some of the discussions carried out in association with this project led to useful insights into this outcome. It is noted that in developing this content, some basic principles were followed. New technologies that are still in the realm of pure research were not investigated, as these were outside the scope of this work. Instead, those approaches that are known to be, or at least likely to be, fully possible but which have not yet reached the state of common practice were the focus. Similarly, areas of technology that are within the realm of common practice were considered to be out of scope, since the intent of this discussion was not to reinforce existing practice but to develop opportunities to move the state of the art forward. The interest area, therefore, is in technologies that are proven in concept but not yet commonly practiced.

As information was gathered, specific efforts were made to find instances of:

- Increased storage for stormwater through in-pipe storage or off-line storage.
- Onsite treatment of stormwater.
- Discharge to the WWTP.
- Controlled storm systems.
- Source control to reduce loadings from sources such as cross connections, inappropriate land use, excessive sprinkler system use, car washing, and swimming pool discharges.
- High-rate intermittent treatment.
- Storage with bleed-back to WWTP for treatment during low-flow periods.
- Swirl/vortex separators with underflow discharged to combined systems.
- Street storage accomplished through regulator modification.
- Catchbasin cleaning for storm drainage systems.

Since innovative methods were targeted, standard BMPs applied in the regular stormwater context were excluded from further reporting, as were examples of the above technologies that do not constitute expanded, extended or innovative approaches to CSO control. It was discovered that few can be constituted truly innovative technologies as most elements identified constituted direct application or extension of existing approaches. However, there were some new methods discovered, and these are described in detail below.

Drivers of Needs for Innovative Practice

To provide a context for evaluation of possible emerging technologies, some of the drivers for future practice were developed. These provided an important basis for evaluation of ideas, because emerging methods can best be identified in the context of emerging needs. A full prediction exercise of global megatrends was beyond the scope of this work; however, some basic questions were identified that provided insights into this aspect.

As a part of this work, issues of pressing need in developing water resources solutions were discussed with practitioners in the field of water resources (Rowney et al., 2008). The primary needs identified, given the context summarized above, are that there are significant gaps in the following areas:

- Calibration process. There are many established methods of adjusting models to better represent reality, but the state of practice in this area remains limited, even with existing tools.

- Planning framework. The notions of practice are well understood, and the profession dedicated to this is mature, but projects can be impacted by uncertainties arising from neighbors or boundary conditions that are unpredictable. In short, some planning problems are unsolvable because the context is uncertain.

- Results interpretation. There are many graphical and statistical methods available to evaluate and communicate solutions, but even so, a lack of consensus or understanding can yield poorly represented or hard-to-interpret results.

- Theoretical underpinnings. There are limitations in the basic theory used to describe BMPs, watersheds and other factors of interest.

- Numerical methods. The applied math is extensive and sophisticated, but the state of practice in this area is limited, in that numerical limitations are not generally discussed as a part of solution development, even though the scale of resolution limitations may be comparable to the scale of solution impacts.

- Future conditions. There are groundswell changes in many basic factors, including economic, resource, population, and climatic, that pose challenges to practice.

- BMP adequacy. Some BMPs, even though they have value in maintaining mass balances, may not be highly effective in controlling all water quality parameters[2]. BMP selection is dependent on intended function, i.e., is not interchangeable, and most likely requires a series approach or treatment train of BMPs to reach higher performance targets.

---

[2] The BMP database has provided insights into performance, but added research in this area is important. Jane Clary (personal communication, August 17, 2009) has suggested "Properly designed, constructed and maintained BMPs can provide significant water quality and load reduction benefits. The effectiveness of BMPs with regard to particular water quality constituents varies based on the characteristics of the constituent and the unit treatment processes present in the BMP. More research is needed in some areas to develop a clearer understanding of factors affecting BMP performance. For example, performance of BMPs with regard to bacteria varies widely. In the case of grass swales and extended detention ponds, which tend to perform poorly for bacteria, it is unclear whether this is due to ineffective unit processes, resuspension of deposited material, regrowth of bacteria, or introduction of additional pollutants due to BMP use by geese and dogs, or some combination of these and other factors."

Those factors that are of major significance in terms of treatment technologies are discussed in more detail below.

Forecasting May Be Increasingly Difficult

Current practice is to base WWF forecasts on estimated or targeted future conditions and test them against rainfall, flow and other records measured in the past. Water usage and wastewater rates are based on per capita rates and assumptions of steady population growth. This type of forecasting based on precipitation records and population trends is common. Forecasting of receiving water quality is also possible. For example, the significant value of databases that many municipalities have in place due to WWTP sampling and reporting requirements can be used. This data in the form of mass balance analyses can provide significant cost savings when field monitoring, i.e., sampling and flow-metering, and laboratory analytical costs are high (Mueller and DiToro, 1977). There is clearly substantial value in the use of historical records to understand present conditions and past trends. Such records will by definition provide a bench mark of past conditions as we move into the future.

The question that arises is how robust are these data, and whether it will become questionable that past records provide a good reflection of what might be experienced in the future. Basic behaviors are well understood, but for some problems, the ability to define the consequence of a specific parameter shift is limited. This is not a new consideration, although it is often overlooked. Tholin et al. (1959) noted that depression storage in even carefully graded urban lots can arise from grading practices. It follows that as lots or barriers (walkways and driveways, for example) change, so can depression storage.

The notions of water quality as a driver of environmental protection are certainly well embedded in our thinking, for good reasons. However, deciding on ways to evaluate the state of a waterway based on specified water quality parameter levels may become questionable in the future. It is already difficult to predict ecosystem responses to a shift in a water quality constituent. Global warming will likely shift precipitation patterns, in ways that are hard to predict. So past events may still be useful to predict future performance, but may not reflect the frequency of conditions of interest. Further, as ecosystems change in response to shifts in climate, the modeling of precipitation into runoff and water quality parameters will also tend to change even if the physical characteristics of an area remain static. Other examples of future change include population distributions in any area. As human behavior adapts to new needs, populations may increase or decrease in any area and preferred habitat forms may shift as well.

The future has always been uncertain; however, given the notion that global changes in climate will lead to changes in ecosystems, we are additionally faced with predicting the response of the system not only as a result of our actions, but also to the changes in basic context. Furthermore, if the system is going to change as a result of global changes, it is not obvious that solutions that pre-suppose the status quo as a future target or expected baseline are based on a valid assumption.

A consequence of this factor is that methods of modeling or prediction that are robust in the face of changing state, response and forcing functions are of interest. Methods that are general extensions of existing predictive methods may not be as useful. Similarly, technologies that are robust in the face of uncertain future requirements or return periods may be advantageous compared to technologies that are brittle in the face of changes. Analytical methods that are targeted at evaluating systems in terms not just of simple quality parameters but in terms of future ecosystem behavior may also be required.

The Scale of a Planning Unit May Shift

There is a basic pattern of behavior in water resources practice that planning on a watershed basis is based on a logical foundation, and local designs are usually based on long-term expectations developed at a wider scale. However, if the rate of development decreases, and if the future is uncertain, then the likelihood of a fully built condition matching the assumed broad scenario may be limited. If the population patterns become unpredictable and consequently drainage requirements are uncertain, it is reasonable to wonder why we should design and build assuming long-term static conditions. More directly, for example, we can question the need for a 100 year design period for pipes that may be obsolete in decades or mere years. Certainly, the large-scale need will be there for major trunks and linkages, and for surrounding features like transportation corridors or other long-term infrastructure components; however, the need may be questionable on more limited scales. Methods that are readily replaced and updated, or that are scalable over a wide range of uncertainty, may be preferable in the future.

Our Understanding of Best Management Practices May Evolve

Some of our BMPs are well understood. Hydraulically, we have a significant ability to fully predict the way that water flows through a system. Others technologies, however, are less amenable to predictive analysis. Water quality transformations through the system can be difficult to predict, and even sediments are still imperfectly represented in models. It is likely that as data increases, the ability to predict certain outcomes will improve because algorithms can be devised to reflect observations. In the mean time, methods are needed to predict outcomes that are effective given the limited mechanistic understanding of BMPs. Some BMPs are more sensitive to this phenomenon than others. Filter type BMPs, for example, have proven to be quite robust in terms of sediment capture performance despite the limited ability to describe other details of BMP function.

It is concluded that methods of sizing and placing BMPs in the face of a prevailing uncertainty as to their quantitative functioning are of interest. Furthermore, BMPs that function well reliably and predictably are of significant interest.

Wider Technical Foundations May be Useful

Engineers, and more recently players from allied sciences (e.g., biology, chemistry, geology, geography, planning), have a strong and increasingly effective culture of collaboration that underlies the approach to water resources problems. The breadth of capability is inherently sound; however, there are fields that are brought into play much less frequently. Some examples of this are sociology, applied mathematics, materials science, computer science and industrial engineering.

Sociology may be valuable in helping predict and shape popular behavior in ways that are environmentally preferred. The realities of adoption and support are a critical known limitation to success, and it is not guaranteed that engineers and scientists are equipped to cope with this aspect of project development and implementation. Commonly, there are local entities that attempt to support this element, but professional competencies are not always demonstrated on this element. In France, practice now commonly includes sociology as a formal skill set in project development. This resource could be readily codified and implemented in U.S. practice. This could potentially translate into the ability to deploy distributed solutions that are dependent on public support with greater confidence.

Mathematicians may be useful in developing mathematical methods that are useful in predicting change in the face of the uncertainties noted above. It is noted that mathematics and mathematicians have played a fundamental role in developing the solutions that are in place now, and that this is not the point at hand. There are branches of mathematics that may be significantly outside present practice and that may add benefits to the water resources arena if brought to bear. The other fields noted may support improved practice in communication, control and performance beyond what is possible with current technologies.

Developments in technical fields outside the normal realm of water resources science and engineering may offer benefits not presently part of established practice, and consideration should be given to extending practice by embracing these kinds of new areas.

Some Potential Avenues for Technical Development

It is interesting to note that most of the results developed from this exploration do not pertain to technical developments, but instead to changes in practice or approach. Nevertheless, some new technologies were identified. Both categories are useful results of this work.

Management or Practice Oriented Approaches

Some of the more illuminating management and practice approaches are noted as follows:

- Implementing emerging targets and embracing multifunctional strategies. Besides peak flows, water quality constituents and other factors, there is also interest in employing new environmental targets. Evaluations of the CO_2 footprint or ecosystem diversity are examples of this approach. Green buildings/infrastructure ideas are increasingly important technologies with impacts in these areas that are beyond analysis of flow or water balance. It may be that new development which generates results in terms of microclimate or global contributions are advisable future directions.

- Extending planning and architecture disciplines so that they understand green technologies in more than the amenity or aesthetic context. The benefits of some of these technologies are only minimally experienced if they are green but attractive. They need to be sound from a wider perspective if they are to be fully functional. Extending the technical capabilities of allied disciplines may improve the end result of implementations.

- Extending water resources disciplines so that new concepts become part of the lexicon. There is a tendency towards the status quo in WWF treatment and control design. A major factor limiting proven technologies in professional practice is not the validity of the technology, but the lack of promotion. Professionals cannot implement what they do not trust or understand, and may be reluctant to implement what is not backed up by peer practice. Confidence may be built and technologies embraced, but this requires education, proof of concept projects, and regulatory encouragement.

- Incorporating redundancy into designs. With uncertainty a reality, sole solutions are risky. Intentional incorporation of redundancy is a risk-coping mechanism.

- Incorporating evolutionary adaptation into designs. Also, as a response to uncertainty, it is advisable to plan and implement facilities in ways that can be staged. If immediate needs are met and if subsequent adaptation is possible, it may be that effective solutions can evolve to meet current needs. For example, the need to abate emerging contaminants is on the horizon. If our control and treatment facilities are designed to enable upgrading for emerging contaminant removal, an important need will be satisfied.

- Avoiding blanket policies. The presumptive approach in the national CSO policy has resulted in a level of control for dischargers. However, the convenience of the presumptive approach enabled stakeholders to avoid the demonstrative approach that would presumably lead to a more efficient local approach, and therefore, in sum, a global approach. Shifting to a stance where presumptive standards are avoided might be preferred in this regard. The EU Water Framework Directive emphasized this approach and could be considered an example of an alternative regulatory stance.

- Embracing the need to manage emerging contaminants. These compounds are known to exist, and are likely to be significant in the future. Current designs based on present contaminants may be of limited efficacy tomorrow. Therefore, there is a built-in obsolescence and multiple future expenditures are likely in order to deal with these new contaminants. Although, as mentioned, incorporating evolutionary adaptation into designs is an approach that will address this feature, an alternative is to review contaminants, sources and control options today, and propel the range of decision factors forward in practice to match what is known or can be inferred from science that is available today.

- Lead with integrated urban planning. Rather than compartmentalizing skill sets, urban landscapes should be designed from the outset so that there is a balance between ecosystems, transportation, recreation, industry and other elements. The current approach incorporates water needs after the fact, which increases difficulty, and often leads to needless redundancies, missed opportunities or other negative outcomes. Yet, the knowledge currently exists to avoid these difficulties. The procedural limitations that forestall a shift in integrated planning are significant, to the point where a change in practice may be a forlorn hope. However, the point should not be lost, as it is not only limited in common practice, but clearly understood well past the point of theory and research.

Device or Technology Oriented Approaches
Some of the newer technological advances include:

- Routinely deploy intelligent materials. There are numerous options for integrating sensing and control systems into pipe networks so that they have a dynamic ability to sense and control flows. Currently, it is not common to build fiber optic or other communication links into pipe system layouts. For example, in the many miles of piping being constructed in the United Kingdom at this time, little has been attempted in this regard. Yet, this practice is well-established and it opens up wider options for subsequent adaptation and control. Implementation of intelligent materials might require different processes in retrofitted and new systems, but in principle it would be possible either way.

- Capitalize on available information technology and model capabilities. RTC tends to be based on rule curves or simple local feedback loops that may have been developed based on a larger analysis of the system. The potential exists, however, to implement control systems that are system wide, using intelligent feedback and prediction methods. If local sensing and control systems are built with adaptability in mind, the life cycle of the system could be based on evolutionary management. Rather than periodic development of new rule curves, real-time adaptation of operation may lead to a net long-term benefit. Continuous adaptation rather than step function adaptation is a desirable and achievable outcome.

- Develop virtual asset management/materials repositories. Virtual management systems (those that represent the system in three dimensions in a visual representation which has physically real characteristics) can be employed, and are routinely used in some industrial contexts. However, these systems are incomplete or non-existent in most water resources applications. As-built or design drawings supplemented by ad hoc analysis as operating management tools still predominate. Virtual systems could provide an expanded ability for managers to “see” the system as a whole and make better informed decisions.

- Implement innovative condition assessment technologies. This is a responsive measure rather than a proactive one, but there are developments that make such things as leak detection easier. Resonance techniques have been used, for example, to detect pipe fractures or irregularities remotely. These may play a role in retrofitting some of the concepts noted above in systems that are not proactively fitted with sensing and communications networks.

- Adopt water balance oriented models. Water balance is a central and fundamental requirement for ecosystem preservation, and attention to water balance could be an immediate and effective step forward. Unfortunately, many tools available today simply do not enable analysis of water balance, as they are limited to rate or concentration. Moving towards a toolset that inherently embraces water balance as a primary outcome would make visible the limitations of practice in some situations. The precedent for this is ample (note for example the Canadian WBM effort (Stephens et al., 2008)), so a shift in practice is immediately and easily possible.

- Use existing facilities. Often during upgrades or system expansions, older facilities or technologies are abandoned or demolished in favor of new approaches. However, older facilities, when properly retrofitted, may offer additional capacity during storm events.

Chapter 4 Regulations and Requirements

Wet-weather Flow

WWFs, including CSO, SSO and stormwater discharges, are one of the leading causes of water-quality impairment in the U.S. today and improvement of WWF controls remains a priority water focus area of the EPA. Pollution problems stemming from WWFs are extensive throughout the country. Problem constituents in WWF include visible matter, pathogens, biochemical oxygen demand (BOD), SS, nutrients, and a variety of known and potential toxicants (e.g., heavy metals, pesticides, petroleum hydrocarbons and emerging pollutants). National estimates have projected costs for WWF pollution abatement in the tens of billions of dollars (APWA, 1992). Municipalities need alternatives to control the high costs of WWF treatment prior to release.

In urban areas, the space to implement stormwater controls is limited. Land is more valuable when developed than when devoted to stormwater control devices, and those devices typically require significant surface area so the implicit costs of control can be high. Therefore, innovative technological advances and risk management approaches are necessary to reduce the severity of pollution and the cost of treatment in urban settings. There are numerous areas where research into application and practice in this area are needed (EPA, 2007a and 2007b).

The design of stormwater systems is still evolving. With the separation of stormwater from sanitary systems, a new pollution routing mechanism was created. Originally, stormwater conveyance systems discharged collected runoff to the nearest receiving water, without storage or treatment, which increased downstream flooding. To rectify this problem, detention basins were mandated for new developments to temporarily store runoff prior to discharge. However, stormwater systems and detention basins increased pollution to receiving waters, so the extended detention concept was introduced to overcome limitations of flood-control detention ponds. Extended detention provided more and better control of the smaller and more frequent storm events that just passed through flood basins; however, questions remained as to the efficacy of extended detention to protect receiving water as discharges may contribute to geomorphic changes to the receiving streams. Currently, low impact development (LID) practices are being implemented to control runoff at the source. In general, over the years, BMP designs that have been developed to address the complexities of storage and treatment, and receiving water quality, have yielded varying results.

Many municipalities are increasingly subjected to TMDLs on solids, nutrients and pathogenic indicators. While existing BMPs may be capable of reducing the amount solids released to receiving waters, the benefits of installing BMPs for other TMDLs are not proven. BMPs typically are not optimally designed to remove nutrients or pathogenic indicators. A more active approach to stormwater management may be warranted for urban areas subjected to TMDLs, using proven concepts for routing, storage and treatment developed for sanitary systems and WWTP subject to intense WWF. This is especially true of many urban areas that still discharge stormwater without any retention and/or treatment and do not have the available space to implement traditional stormwater BMP designs.

Many European nations opted to continue using CSS instead of switching over to separate sanitary and storm sewer systems so as to treat stormwater centrally with high-rate treatment mechanisms. The practice of installing new combined sewers in the U.S. is not popular. However, collecting a portion of the urban stormwater that is driving implementation of TMDLs and pumping it through an aggressive treatment, or pumping to a WWTP, may help address TMDLs.

The following discussion provides a summary of the current regulatory framework governing CSO and municipal stormwater discharges and is relevant to all four topic areas. The CSO regulatory discussion is followed by a summary of the municipal stormwater regulatory program. Four separate literature review chapters are then presented that relate to each specific topic area.

U.S. Combined Sewer Overflow Regulatory Background

Although the technologies embraced by this report are trans-jurisdictional, the application context is for the U.S. and it is national in scope. Accordingly, it is useful to recap some of the fundamental drivers related to effluent discharge of the type being considered. It is noted that interpretations, amplifications or supplementary requirements that may be in place at a state level are beyond the scope of this review. It is also noted that the text that follows is patterned closely after the cited documents but may not be identical to those works, and should not be interpreted as an amendment or intentional exception to them. Reference should be made to the original documents for determination of existing policy.

EPA's 1994 CSO Control Policy (EPA, 1994) accelerated compliance with Clean Water Act (CWA) requirements and established a consistent national program to control CSO discharges through the National Pollutant Discharge Elimination System (NPDES) permit program. Policy provisions included discharge characterization, implementation of technology-based controls and development of a long-term control plan (LTCP) that evaluates CSO control alternatives to meet CWA compliance with water quality standards (WQS) and protection of designated uses. The policy reiterated objectives to minimize the water quality, aquatic biota, and human health impacts from CSO discharges.

Four key principles are presented in the policy to ensure CSO controls are cost-effective and meet the objectives of the CWA. These principles include:

1. Clear levels of control that would be presumed to meet objectives.

2. Flexibility to consider the site-specific nature of CSOs and to determine the most cost-effective means to reduce pollutants and meet CWA requirements.

3. Allowance for a phased approach given a community's financial capability.

4. Review and revision of water quality standards and implementation procedures when developing CSO control plans to reflect the site-specific wet-weather impacts of CSOs.

The policy provides a national framework for a comprehensive and coordinated planning effort to achieve cost-effective CSO controls that meet local objectives. It acknowledges the site-specific nature of CSOs and provides flexibility to tailor planning efforts to local conditions. It requires CSO permittees with CSS that have CSOs to immediately undertake a process to accurately characterize their sewer system, demonstrate implementation of nine minimum controls, and develop and implement a CSO LTCP. The policy notes that the CWA requires immediate compliance with technology-based controls, and states that a compliance schedule for implementing the nine minimum control measures, if necessary, should be included.

These nine minimum controls include the following:

1. Proper operation and regular maintenance programs for the sewer system and the CSOs.

2. Maximum use of the collection system for storage.

3. Review and modification of pretreatment requirements to assure CSO impacts are minimized.

4. Maximization of flow to the publicly owned treatment works (POTW) for treatment.

5. Prohibition of CSOs during dry weather.

6. Control of solid and floatable materials in CSOs.

7. Pollution prevention.

8. Public notification to ensure that the public receives adequate notification of CSO occurrences and CSO impacts.

9. Monitoring to effectively characterize CSO impacts and the efficacy of CSO controls.

Suggestions for evaluating control option alternatives include performance-based options, such as setting a maximum allowance of overflow episodes permitted per year, providing controls that achieve a designated capture rate, or expansion of the POTW secondary and primary capacity.

Given that the final CSO LTCP becomes the basis for NPDES permit limits and requirements, the selected controls are intended to be sufficient to meet CWA requirements. Examples such as enlarging a sewer trunk line or adding storage tanks are acceptable CSO control alternatives. Both alternatives increase the storage capacity of the sewer system, thereby decreasing the sanitary and stormwater flow volume that could otherwise overflow prior to discharging into the treatment plant.

U.S. Stormwater Regulatory Background

The possible negative impacts from urban runoff were recognized during the drafting of the CWA. EPA established the Nationwide Urban Runoff Program (NURP) in 1978 to delineate urban surface water quality impairments and review existing control strategies. The NURP study was implemented to provide credible technical information to guide stormwater management policy. NURP data were used in a planning context that resulted in a practical approach for planning decisions regarding urban runoff.

The NURP study characterized types of pollutants, loads and receiving water quality effects, and evaluated stormwater control structures to remove pollutants in urban runoff. A principle objective of the nationwide assessment study was to characterize conventional pollutants found in runoff. This list contained solids, oxygen-consuming constituents, nutrients and heavy metals. Although the NURP (EPA, 1983) study confirmed urban runoff as the transport mechanism of contaminants, it concluded that land-use categories could not predict differences in site contaminant values.

Clean Water Act

Originally, this act was entitled the Federal Water Pollution Control Act of 1948 (FWPCA), and prescribed a regulatory system consisting mainly of State-developed ambient WQS applicable to interstate or navigable waters. In 1972, FWPCA amendments established a system of standards, permits and enforcement aimed at the "goals" of attaining "fishable and swimmable waters by 1983" and "total elimination of pollutant discharges into navigable waters by 1985." (33 U. S.C. § 1251 (a) (2)). Further amendments were passed in 1977, when the Act was officially named the “Clean Water Act.”

Concern over the accumulation and distribution of diffused pollutants into the nation’s waterways led to the passage of the Water Quality Act of 1987. This Act included new CWA amendments that required EPA to develop regulations governing stormwater discharged from municipalities and certain industrial activities, including construction. Pursuant to section 402p(3) of the CWA, large and medium sized municipalities were required to enact controls to reduce the discharge of pollutants to the *maximum extent practicable* (MEP). Stormwater associated with industrial activity was required to meet all applicable provisions of sections 402(p) and 301 (Effluent Limitations). Thus, the MEP standard was set for municipal stormwater discharges, whereas technology-based effluent limitations were set for industrial stormwater discharges. The MEP term was never defined in the CWA.

For a given water body or segment, WQS usually include a designated use, water quality criteria to protect the designated use, and an antidegradation policy. Water quality criteria are levels of individual pollutants, characteristics or descriptions that, if met, will protect the designated use(s). WQS are discussed in several CWA sections. Section 101(a) specifies that the standards should provide sufficient water quality for the protection and propagation of fish, shellfish and wildlife, and water recreation. Section 303(c) states that WQS should be established for water bodies while taking into account their uses and value for public water supplies, fish and wildlife, recreational, and agriculture. Section 303(d) requires that all states identify water bodies that do not meet applicable WQS for designated uses. States compile these impaired water bodies and estuaries on a list, typically referred to as the state 303(d) list. The law requires that priority rankings be established for these listed waters.

The CWA also requires that a TMDL be developed for parameters affecting the 303(d) listed waters from meeting the state WQS. The results of a TMDL study are three-fold:

1. Identify the sources of the impairment.
2. Calculate the allowable load that various sources can discharge to a water quality limited stream and still meet state stream standards.
3. Allocate the effort among these same sources to reduce their current loads that discharge into the stream.

The TMDL allocation has to be written to achieve the state stream standard. The stream standard is based on its designated beneficial uses, and criteria are developed to protect the designated uses of that stream. Following the approval of a TMDL, by both state and federal agencies, a TMDL Implementation Plan must be developed that specifies the necessary steps to achieve the load allocations and reductions set forth in the TMDL. After the Implementation Plan is completed and fully implemented, it may yet take years for the receiving water quality to show improvement.

Stormwater Permit Programs

On November 16, 1990, the EPA promulgated NPDES stormwater permit regulations for discharges associated with industrial activities and large and medium municipalities with separate sanitary and storm sewers, deemed Municipal Separate Storm Sewer Systems (MS4s). MS4s were required to submit individual permit applications covering stormwater discharges and develop and implement a stormwater management program to mitigate the impacts of urbanization on receiving waters. Specified industrial activities or categories, including construction sites disturbing more than five acres, were required to apply for coverage under general stormwater permits and develop and implement a site-specific stormwater pollution prevention plan. The NPDES stormwater program was expanded in 1999 to include discharges from smaller MS4s, which can be covered by a general permit, and construction sites that disturb one to five acres. The EPA noted in the 1990 regulations that BMPs were appropriate controls to limit stormwater pollutant discharges.

Combined Sewer Overflow and Stormwater Permit Programs

Common attributes of the CSO and stormwater discharge programs are evident. Both programs require the development of a plan or program to reduce the discharges of pollutants into federal waterways. No specified treatment technologies are prescribed in either the CSO or stormwater permit program, providing a flexible planning capability for local jurisdictions to mitigate pollutants in their respective discharges. The systems have long been regulated separately and considered mutually exclusive. However, over time, MS4 operators may find that portions of their storm sewer, especially those sewers operating in older, denser areas of their jurisdictions, or areas experiencing flows even during dry periods, behave more like combined sewers. The potential for program overlap provides municipal operators with a broader array of tools and research literature to investigate as stormwater program elements are adapted to further reduce pollutant discharges and meet state stream standards.

Chapter 5 Topic Area 1: Wet-weather Flow Technology and Management Development

The literature is replete with references to existing technologies that are at a stage of maturity or that have left the realm of innovation to become standard practice. Some newer technologies offer valid opportunities for research, and productive efforts in this regard are under way. Vortex separators and other hydrodynamic devices, for example, have been a practical option for well over a decade, but are still being researched and developed further. Boner et al. (1994) showed that modifications to conventional vortex separators provide performance improvements that suggested an increased efficacy when used in the CSO context. EPA's Office of Research and Development (ORD) has documented progress on this technology, especially the EPA swirl, from the 1970s onward (further information about the swirl can be obtained by going to http://www.epa.gov/ednnrmrl/). This effort is valuable, but peripheral to the content of the present report.

Although the general literature in this area is vast, it turned out that technologies suitable for exploration in this project were relatively limited because of the intended scope of work. Almost by definition, technologies that are not a part of accepted practice may appear as bench- or pilot-scale studies, and to a much more limited degree, full-scale studies in research journals, but not in the popular literature. In the normal course of events, those topics that are valuable and are past the point of pure research are developed and productized by industry, and are widespread in the literature accordingly. Those that have not reached the stage where they have had this aggressive focus, yet are credible alternatives, and are harder to find in the literature. However, there are some areas that appear to offer well founded promise as innovative extensions to current practice, and these are related in the following discussion.

Stormwater Best Management Practices

One area that seems to fit the intent of this work is biofiltration, also known as "rain gardens", especially when incorporated into individual or privately-owned lots. The principles are known and readily stated, in that standard filter type BMPs seem to have the potential for improved removal if biological elements such as grasses, ornamental herbaceous or shrubs are incorporated in the filter matrix. Planted systems also reduce WWF volume through evapotranspiration over other types of non-biological BMPs. However, the state of this technology might be described as beyond research but short of generally accepted practice from a quantitative perspective.

A field visit to facilities in the City of Austin, TX conducted during the course of this project attests to this. The facilities, "Austin sand filters", are some of the earliest implemented WWF controls for both quantity and quality and have a long history of continuous operation. These BMPs are sand filters with a stilling forebay (Barret, 2003). Some of these facilities have emerged as bona fide bioretention facilities, with mature and stable biological communities evident on the filter surface. Digging into the soil on this surface showed that a rich soil matrix exists, with a well established rooting zone. Performance of the system was observed in wet conditions, and hydraulic impairment was not observed. So this seems to be a case in point demonstrating the validity of the bioretention concept as a way to reduce runoff volume and enable biological treatment concurrently. However, it was also found that the systems are not monitored or measured in a way that currently enables a confident interpretation of the influence of biological community on the removal processes. It is interesting to speculate whether the facilities observed in Austin may have demonstrated hydraulic conductivity losses had biological communities not emerged over time. The University of Texas at Austin is conducting research into this effect at this time (Barrett, 2009).

These observations are mirrored in the bioretention literature. Le Coustumer et al. (2008) noted in a study of biofiltration systems over a period of one and a half years that significant hydraulic conductivity losses could be experienced in infiltration systems due to clogging. They also found that although vegetation does not generally affect this result, some vegetation could actually increase hydraulic conductivity over time. They note the relationship between sizing and clogging, with smaller (relative to catchment size) facilities more readily clogged than larger ones. The finding that the Austin facilities were not visibly hydraulically impaired by vegetation seems to support Le Coustumer's finding that vegetation could increase conductivity over time.

While indicating that the use of bioretention over the last 15 years has increased in the U.S. with a growing amount of research literature supporting volume reduction and pollutant removals, Davis et al. (2009) noted that this is still an immature technology as design questions still exist, and long-term operation and maintenance practices are not defined. Hatt et al. (2008) discussed biofiltration devices and presented the results of field-scale studies, citing the relatively limited availability of full-scale data at this time. Their results, however, confirmed the findings of laboratory-scale efforts. In a closely related paper, Lewis et al. (2008) described the impact of vegetation on maintenance of soil infiltration capacity and on increasing evapotranspiration in such systems. The need for research is also detailed by Barraud et al. (2002), which described an effort to examine the long-term impact of infiltration-based BMPs. They cited the time scale involved as being in the order of a decade, which underlines the point that the biology, ecology, hydrology, chemistry and soil science implications of these devises are still relatively unknown.

During hydraulic and water quality monitoring of an extended detention wet pond incorporating various sand filter designs, Vollertsen et al. (2008) analyzed the data and found that algae, which affects dissolved oxygen (DO) concentrations, became active starting in early March. Turbidity of the pond water was directly impacted by the larger events. The researchers also observed that the pond behaved more like a completely mixed reactor than plug flow reactor.

Combined Sewer Overflows

Field et al. (2004) summarized characteristics of CSOs, including a description of impacts of CSOs, as well as the resources spent and technologies used by municipalities to reduce impacts. It established a baseline of related data concerning sewerage management and describes typical technologies and operational practices to reduce CSO impacts. Summaries of major discussion topics follow.

Collection System Controls

Collection system controls maximize the capacity of the sewer system to store or transport wastewater through hydraulic control point adjustments to maximize system storage capacity while minimizing the volume of infiltration and inflow (I/I) into the system undergoing treatment. The controls may include maximizing flow delivered to the plant for treatment, disconnecting stormwater discharges into the collection system, developing a more effective system using RTC to monitor flow rates and more effectively managing the system's storage capacity while maximizing the flow volume directed to the plant during WWF, and rehabilitating the sewer system.

Storage Facilities

In-line or off-line storage options provide additional capacity when sewer system capacity is unable to transport or provide full treatment for WWF. In-line storage of WWF is provided within the sewer system and includes the use of flow regulators, in-line tanks or basins and parallel relief sewers. DWFs pass directly through these facilities. Flow regulators optimize in-line storage capacity by adjusting the flow into or out of the facility during WWF. In-line storage capacity can be supplemented by the installation of parallel relief sewers or replacing older pipes with larger diameter pipes. Field et al. (2004) note that areas of mild slopes provide the best opportunity for in-line storage facilities, while observing that this method can potentially increase wastewater basement back-up and street flooding. The mild slope may also promote sedimentation and debris accumulation within the sewer. The traditional solution to prevent solids deposition within the collection system is to have design wastewater flow velocities high enough to continually flush sediments and prevent solids accumulations within the pipe.

Off-line storage facilities store WWF in near-surface tanks and basins or deep tunnel locations. Off-line facilities can be adapted to numerous site-specific designs and settings relating to basin volume, inlet and outlet structure, and disinfection process. Flows are routed around the off-line facility during dry weather, whereas during wet weather, wastewater discharges are pumped or flow by gravity into the storage facility. Overflows can arise if the system's capacity is exceeded. The primary utility of the facility is storage of WWF and solids settling when storm flow volume exceeds storage capacity.

On-site storage at the wastewater treatment plant can also be used as a control where the capacity of the wastewater collection system exceeds that of the treatment facility. The two most common types of on-site storage are flow equalization basins and the conversion of abandoned or outdated treatment units, such as clarifiers or lagoons.

In areas where in-line storage is not attainable or unavailable, the cost of off-line storage is usually more expensive than in-line storage. The costs associated with on-site storage are typically lower than the construction of near surface off-line facilities because the on-site storage facility is typically located on land already owned by the facility. Expanding conveyance capacity is usually the most expensive storage development option (though analysis later will show that this presumes expansion in developed areas: see Chapter 6).

Treatment Technologies
In those collection systems where WWF exceeds the sewer conveyance and treatment facility capacity, end-of-pipe controls may be used in lieu of or in addition to storing excess flow. Different pollutants, such as solids, bacteria or floatables, use specific treatment technologies. The disinfection of excess WWF is used as an end-of-pipe treatment for bacteria removal, whereas hydrodynamic separators, such as swirls and vortex, are used for solids and floatables removal. Given the assumption that DWFs are treated at the wastewater treatment facility, these technologies are assumed to operate only during wet weather or storage dewatering conditions.

Supplemental Treatment
These technologies supplement treatment during WWFs. An example of such a supplement would be the installation of a parallel treatment process at a wastewater treatment plant that is only operated during WWF. Potential supplemental treatment technology options for excess WWF include ballasted flocculation or chemical flocculation to accelerate the settling of solids, deep bed filtration using anthracite and sand, and microscreens. These technologies must be dependable, be responsive to intermittent flows, and be able to handle variable flow regimes and influent pollutant concentrations.

Plant Modifications
Plant modifications to existing treatment process configurations or operations can increase the WWTP's ability to handle and treat WWF. Such examples include providing an even flow distribution between process treatment units, baffle installation to prevent hydraulic upsets in clarifiers, adding flocculants to accelerate SS removal, switching a portion of flow delivery from the primary to bypass the secondary units, and switching from series operation during dry weather to a parallel operation of unit processes during WWF (blending). Performance evaluations are necessary to confirm whether additional treatment capacity developed for WWF blending may adversely impact the pollutant removal and treatment process for DWF. Issues relating to an increased pollutant concentration of the plant's effluent during dry weather can occur given the retooling of the plant to optimize WWF treatment processes if the plant modifications are not properly designed.

Disinfection
Application of disinfection to CSO discharges has been limited, when compared to the disinfection unit process used

in WWTP. High flow rates and partially treated wastewater may adversely impact the disinfection process if the exposure of the disinfection agent to the wastewater undergoing treatment is reduced. Chlorine disinfection is the method most often used to disinfect WWF. Toxic residual chlorine and disinfection byproducts limit the usefulness of chlorine disinfection in those areas that have high organic solids in their effluent. It is suggested that ultraviolet (UV) light may be an alternative disinfection method for WWF that first receives a minimum of primary treatment or equivalent; however, the efficiency of UV also is subject to the reduction of solids (Chapter 7 discusses relationship between solids and disinfection technologies).

Vortex Separators

Vortex separators are designed to separate and concentrate settleable solids and floatables. The diluted effluent is then discharged for further treatment or to the receiving stream. These separators have limited capability to reduce concentrations of indicator bacteria, other small and light particles, or dissolved contaminants. These devices should not be placed in a treatment train downstream of other units that provide the same function, i.e., primary treatment, though disinfection treatment prior to discharge has been performed.

Low Impact Development Techniques

LID techniques can be used to attenuate stormwater runoff discharging into the sewer collection system, thereby potentially reducing the volume or occurrences of CSO events and capacity of downstream control facilities. LID controls provide runoff volume storage opportunities and include technologies such as swales, porous pavement, bioretention facilities (rain gardens), green roofs, and rain barrels or cisterns, and other water conservation practices. Incorporating LID controls into the footprint of urban developments decrease the storage volume capacity required in sewer collection and CSO control.

Technology Combinations

Some technologies work well when applied together. Some of the combinations suggested by Field et al. (2004) are:

Low Impact Develoment Designs Coupled with Structural Controls

Both controls reduce the peak flow rate and quantity of runoff that enters the sewer collection system. The runoff volume and peak flow reductions allow for the size of downstream storage control structures to be reduced, managed more efficiently or even eliminated.

Disinfection Coupled with Solids Removal

Numerous pollutants in wastewater discharges can interfere with and reduce the effectiveness of disinfection processes. These pollutants include high concentrations of BOD, ammonia and iron, which consume or prevent the disinfectant from interacting with the microorganisms. Larger solid particles can shield microorganisms located in the particle's interior from the disinfectant's effect. Physical shielding by insulating solids can be significant for all disinfecting processes, including UV, chlorine, chlorine dioxide or ozone.

Solids removal enhances disinfection by settling out shielding particles and the clad pathogens. Using effective solids removal controls can improve the performance of disinfection process units treating CSO discharges. For systems using chlorine, solids removal reduces both chlorine demand and subsequent de-chlorination. Off-line storage facilities, vortex separators and supplemental treatment facilities have demonstrated additional benefits by removing solids out of the wastewater stream prior to disinfection.

Sewer Rehabilitation Coupled with Sewer Cleaning

Sewer cleaning techniques should be conducted or at least considered before scheduling the rehabilitation of the sewer collection or CSO control facility systems; in this way needless and expensive infrastructure replacements are avoided when simple maintenance and cleaning are all that is necessary.

Real-Time Control (RTC) Coupled with In-line and Off-line Storage Tanks

RTC technology is used to maximize flow to the treatment plant and storage within the sewer. Both outcomes serve to reduce the volume and frequency of untreated overflows. RTC uses operating rules, monitoring data, and software (Supervisory Control And Data Acquisition [SCADA] systems) to dynamically operate system components to optimize wastewater routing, treatment and storage. System components of RTC include weirs, gates, pumps, valves and dams. RTC is most often employed in sewers that have considerable in-line storage using large pipes designed for excess WWF. Off-line storage facilities, such as tunnels or basins, can also be operated by RTC. The dynamic operations resulting from RTC optimize the sewer storage volume available for excess WWF.

Real-time Control

Kurth et al. (2008) reported on an Artificial Neural Network (ANN) to predict the hydraulic characteristics of CSO discharges. The purpose is to develop a monitoring, modeling and operational strategy that uses input from weather radar to predict hydraulic performance of CSO assets. Three United Kingdom (UK) drainage areas with local rainfall data were used to predict consequences on water depth and weir crest elevation. The data were used to train the system; the system was then validated and tested using a hidden three-layer, feed-forward perceptron. The work seeks to eventually distinguish between root causes of rising water patterns, such as flow from rainfall or an anomaly (e.g., a spill) and to input the information into a decision support tool incorporating a phased alarm system. Preliminary results from the first drainage area were positive for predicting the performance of a normal CSO structure 15 minutes ahead using three time steps.

Seggelke et al. (2008) introduced an integrated control approach whereby treatment processes are continuously monitored and models are applied to predict treatment capacity of the plan. The controller is meant to continuously adjusts the plant’s inflow rate with the object to reduce CSOs and collect early information on the plant’s critical process conditions. The controller is rule-based; simulations are used to test scenarios, analyze the storm events and adjust the control approach. Results indicated a fixed value of the maximum WWTP inflow does not provide the best use of treatment capacity at the plant.

Implementation of the EU Water Framework Directive requires ecosystem status evaluation. Blumensaat et al. (2008) elaborated on the reduction of model uncertainty given online water quality data. Long-term discharge and water quality monitoring data were used to develop reliable simulation results used for decision support input to facilitate system optimization. The researchers reported that the assumption of constant biomass conditions and the disregard of particulate water quality parameters in a sediment compartment appeared to limit the conclusions drawn, particularly for the long term.

Monitoring methods are continually being improved or developed, and this too may offer options for advanced practices. Grüning et al. (2002) reported on continuous sensors of dissolved and particulate solids and DO, used to reliably determine CSO releases in a site studied by them. They developed a statistical relationship between the two solids parameters and the chemical oxygen demand (COD). They asserted that the methods tested enable the real-time control of sewer systems on the basis of the pollution carried in the combined sewage. They suggested that this enables the collection or retention of CSO flows based on composition, offering an extended means of optimizing performance. By saving collection volume for the worst event periods, the system can maximize capture and treatment of constituents.

There may also be some insights into innovative ways to deploy existing facilities and technologies, including more active approaches to mitigating CSO impacts which are the converse of normal practice. For example, instead of regulating outflows to match receiving water assimilative capacity, the option exists in some cases to regulate the receiving water to match CSO discharge needs. Achleitner and Rauch (2007) evaluated increasing river base-flow in manmade low flow sections to dilute the adverse impacts caused by upstream CSO discharges. The increase in flow is conducted through upstream operation of retaining structures such as hydropower intake structures. Such an operation requires close coordination between the municipalities and the energy producers. This scheme fits the EU’s

Water Framework Directive for basin-wide improvement approaches, although it seems this solution may simply be the equivalent of older discharge management approaches that were simply based on dilution. The focus of the approach is on the mitigation of the acute pollution but it does not address the accumulation of pollutants. However, it does embody the RTC framework by using a model-based predictive control. An algorithm developed for the operation was tested off-line using a semi-virtual catchment. Costs for the measure were estimated as equivalent costs due to losses in energy production. Costs for annual spilled water as well as peak flows generated in the river system were also considered.

Lacour et al. (2008) continuously collected turbidity, conductivity and flow measurements at one-minute intervals for a year at two sites located in the Paris CSS to monitor the evolution of pollutant flow discharges. Turbidity and conductivity are both convenient indicators of the amount of constituent mass being transported. Results highlighted the variability of turbidity dynamics during WWF events. Attempts were made to predict turbidity responses from hydraulic flow characteristics. Flow and turbidity measurements were compared for each storm event and no relationship was found between hydraulic flow dynamics and turbidity. It was suggested that the knowledge of turbidity dynamics could potentially improve wet-weather system management when using continuous pollutant measurements because turbidity incorporates information neither predicted from nor included by hydraulic flow dynamics.

Abda et al. (2008) experimented with simultaneous measurements of real-time sewage velocity and suspended particle concentration measurements within sewer lines. The intent of the instrumentation was to provide on-line data availability of velocity profile measurements, water height measurements, estimates of suspended solid concentrations and granulometric size class distributions of the solids. Results compared well with other gauging methods, showing good agreement for velocity, water height and suspended solid concentrations. Further refinements are needed to improve the SS granulometric analysis.

Guillon et al. (2008) evaluated on-line tanks in the HAUT-de-Seine sewer network to reduce CSOs to the River Seine. This system has over 100 CSOs to the River Seine of which 22 have been outfitted with automatic gates to regulate flow. These 22 are locally operated as RTC to optimize in-line storage. The remaining fixed regulators were evaluated for replacement with automatic gates. The main CSO locations were evaluated in terms of annual overflow volume, in addition to the evaluation of the remaining storage capacity of the whole network. Both the locations where the first overflow occurred, and when flooding began were identified. Selection criteria eliminated those CSO locations that have small pipes or steep slopes, where storage opportunities are at a minimum. CSO locations were identified where mild slopes and large pipes were present. The next phase of the study will prioritize those locations for the installation of real-time automatic gates to maximize the storage volume within the network.

An EPA report (2006) provided a summary and a broad introduction to several different aspects, including hydraulics, instrumentation, remote monitoring, process control, software development, mathematical modeling, organizational issues, and forecasting of rainfall or flows but does not elaborate on them in great detail. The main goal of the report was to provide a guide on RTC technology to facilitate its understanding and acceptance by the user community.

Taken together, these papers illustrate a general finding of interest to this project. It may be that the combination of monitoring, control and analysis have emerged as a viable area for further development. Each is relatively well developed, but the pursuit of combining all three to manage a physical watershed system still has areas worthy of more development. RTC is old news in this field, but the basis for RTC still tends to be hydraulic. Applications of RTC that embrace monitoring both the discharge stream and receiving water, or making predictions based on water quality control targets were not found in the literature. There were examples of models that can apply collected data and then develop rule curves, but not of real-time monitoring and control at a time-scale relevant to discharge event management. A very current effort supported by substantial funding and technical excellence can be found in the work of Guillon et al. (2008). Their work involves numerous on-line tanks in the HAUT-de-Seine sewer system, and communications with the author during the course of this project determined that the managers had proceeded with independent pre-determined rule curves and that they were only recently considering movement to a global approach to discharge management. This would seem to be a strong case in point for this project. The individual elements of this project are all in place and proven, but the combinations are not.

Multicriteria Decision Support Systems

Numerous researchers have developed methods using multicriteria parameters to evaluate the technical performance of BMPs. The parameters selected are multicriteria in nature and supplement the technical domain with ecological aspects. The use of these decision-support methods represents a new application of technology to the evaluation and selection of BMPs. Scholes et al. (2007) proposed to theoretically assess BMP pollutant removal performance. This benchmarking method combines primary removal processes within 15 different BMPs and evaluates each process to remove a pollutant. A value representing the pollutant removal potential for each BMP is developed. Jensen et al. (2008) applied this benchmarking methodology to systematically evaluate the effectiveness of four BMPs. General criteria relating to environmental, economical and technical attributes were used to develop the specific parameters. This method links the compatibility of technology with the urban setting. Specific parameters included identifying the fine fractions of SS, dissolved metals, maintenance frequency and expected life span. The results can be used as inputs to existing urban hydrology models and applied to evaluation and prioritization of pollutant removal technologies. The researchers also discussed the limitations of BMP design.

Cardosa and Baptista (2008) proposed a decision making method to use in the preliminary planning phase of development to evaluate alternatives for urban waterways. The method included indicators related to hydrologic, hydraulic, environmental, sanitary and socio-economic attributes. They noted that conventional solutions for flood control or transportation infrastructure development traditionally contain and suppress surface waters from urban landscapes and lead to negative environmental impacts.

Moura et al. (2008) reported that the long-term sustainability of infiltration systems has not been established and their real performance must be assessed. A multicriteria decision support system (DSS) has been proposed to evaluate infiltration systems. A set of performance indicators integrating technical, economical, environmental and social attributes was developed for use in the DSS. The method was demonstrated on a test case and indicated low sensitivity and high robustness to parameter variation.

Ultimately, improving receiving water quality is the ultimate goal of implementing WWF treatment. The effluent quality load discharged from BMPs in the U.S. will come under greater scrutiny as more TMDLs are imposed to control WWF discharges. Mietzel and Frehmann (2008) reported that efficiency alone is an unsuitable parameter to assess the performance of stormwater treatment facilities, i.e., the amount of pollutants retained in the facility, because it is dependent on the influent's characteristics. The researchers noted that this parameter is often used to benchmark the performance of stormwater BMPs.

Alfaqih et al. (2008) described an environmental decision analysis framework to identify potential *Escherichia coli* (*E. coli*) sources into the upper portion of a major water supply lake during high stream flow conditions. The framework's focus is to assist decision makers in the early stages of a project to collect, assimilate and incorporate an interdisciplinary set of theories and methodological approaches to address all data, stakeholder concerns, and constraints.

Chapter 6 Topic Area 2: Contrasting Tank and In-line Storage Solutions

Managing Sewer Solids and In-line and Off-line Storage

Research is continuing toward the development of an integrated understanding of sewer solids, hydraulic characteristics and associated biological/physical/chemical processes.

As discussed in the review of CSO regulatory background, in-line and off-line storage options provide additional capacity when a CSS is unable to transport or provide full treatment for WWF. In-line storage of WWF is provided within the sewer system, allowing DWF to pass directly through these facilities. Areas of mild slopes provide the best opportunity for in-line storage facilities, but this method can potentially increase wastewater basement back-up and street flooding. The mild slope may also promote sedimentation and debris accumulation in the sewer. The accumulation of sediment and debris can affect hydraulic properties of the CSS. The traditional solution to prevent solids deposition within a sanitary collection system is to have design wastewater flow velocities high enough to continually flush sediments and prevent solids accumulations within the pipe; this is not always the case for CSS.

Off-line storage facilities retain WWF in near-surface tanks and basins, or deep tunnel locations. Off-line facilities can be adapted to numerous site-specific designs and settings relating to basin volume, inlet and outlet structure, and disinfection processes. Flows are routed around the off-line facility during dry weather, whereas during wet weather, wastewater discharges are pumped or flow by gravity into the storage facility. Overflows can arise if capacity is exceeded. The primary utility of the facility is storage of WWF for later treatment at the WWTP, although solids settling may occur during periods overflow.

On-site storage at the WWTP can also be used as a control where the capacity of the wastewater collection system exceeds that of the treatment facility. Field et al. (2004) cite some common types of on-site storage at WWTP, including expanded primary tanks, flow equalization basins (which are designed to reduce diurnal variation and maintain a constant flow through the plant, but can also augment WWF), and the conversion of abandoned treatment units, such as clarifiers or lagoons.

In-pipe Processes

In-line options are typically used to provide temporary storage of excess WWF and often are not intended to provide direct opportunities for treatment. Nevertheless, there can still be water quality impacts. In-pipe sewerage solids can settle out within the CSS as flow velocities vary and as they become low enough that deposition occurs. This is not a one-way process as deposited materials may also become resuspended during subsequent WWF events contributing to loadings to receiving waters by CSOs. The mechanics of this cycle are understood in general, but research is still being done on the ways that constituents are routed and transformed through the system. A greater understanding of sewer operational issues relating to in-pipe solids deposition and erosion processes can assist in the development of more efficient collection system operations. This same understanding offers a wider opportunity, which is to intentionally pursue the development of potential treatments in the pipe as a major design factor.

Sewer Solids

Many aspects of solids in sewer systems have been under investigation by European researchers over the past two decades (e.g., Ashley et al., 2003). Some of these relate to traditional sewer operation and functions that are affected by deposition/resuspension processes, while some are related to emerging problems of interest. Solids deposition can inhibit conveyance which can itself exacerbate deposition, and excessive deposition in sewers can lead to more frequent CSO due to reductions in capacity. Traditionally, foul flushes that occur during the first part of the storm event have been related to the largest SS phase of the discharge flowing through a CSO structure. More recently, European researchers have focused attention on the mechanics of fluid-solids interactions, which is attributable to concerns regarding pollutant releases from in-pipe sediment depositions during WWF (Ashley, 2005).

Most of the observed load originates from eroding of the sewer solids bed resulting in increased solids at the treatment facility or a potential pollutant release during overflows (McIllhatton et al., 2002). The near-bed solids are re-entrained into the wastewater flow together with solids from the bulk bed and account for large changes in the SS concentrations during variable time and flow conditions. Research by Schellart et al. (2005) indicated that microbial activity can influence the physical release of in-pipe sewer sediment. Erosion was evaluated under aerobic and anaerobic conditions and two temperature settings.

A new application of a maritime technology measured both the cross-section and longitudinal sediment profiles in large sewers (Bertrand-Krajewski et al., 2008). A marine sonar unit with an attached laser meter on a floating frame was successfully tested in a large sewer. Each cross section sediment profile measurement was immediately available for review on a laptop by the operator. Data processing entailed Excel® and AutoCAD® software to automatically correct the raw data files so as to draw simplified two and three dimensional views and calculate sediment areas and volumes. The prototype provided results accurate to approximately ±1 cm, which was anticipated to be highly effective in CSS tunnel and sediment assessment contexts.

Banasiak et al. (2005) evaluated the impact of biological processes on physical properties such as bulk density, water content, deposited-sediment structure and erosion behavior as a function of bed shear stresses. The researchers indicated that bio-processes weaken the strength of the in-pipe sediment deposits and a significantly weaker layer was observed during deposition with oxygen-rich and quiescent conditions. The deposited material had a low shear strength which may be the source of the foul flush sediments when flow rates are increased. Rushforth et al. (2007) conducted full-scale hydraulic testing that used sewage and real in-sewer sediments. Bed-load transportation rates resulting from different steady-flow discharges and flow depths were measured and evaluated against existing transport-rate predictions. The comparisons indicated that the mobility of the sewer sediments was likely to be significantly under-predicted by the existing transport-rate prediction method, particularly if a range of grain sizes is present. Biggs et al. (2005) investigated the change of particle sizes eroding from a previously deposited pipe sediment bed using a rotating annular flume. The simulator’s environment was controlled to investigate the effect of varying consolidation time, temperature and sediment characteristics on the amount and particle sizes that erode from the sediment bed under increasing shear conditions. The median size of the particles eroded did not vary significantly with temperature, although the SS concentration of the eroded particles was greater for the higher temperature under the same shear stresses, indicative of a weaker bed deposit. Using different types of sediment had a marked impact on the particles sizes that eroded.

Flushing Technologies

Within the realm of sewer solids research, European researchers also continue to delve into the hydraulic and environmental problems related to removal of in-pipe solids accumulation. The traditional approach to limiting the deposition of sewer solids has been to specify a minimum self-cleaning velocity as part of the sewer design process. However, it is noted that designating a minimum velocity does not provide insight into sediment characteristics and concentrations or other aspects related to the ability of sewer flows to transport sediment (Butler et al., 2003). The recognition that both hydraulic and environmental problems are associated with sewer solids has provided a renewed interest in mitigating solids accumulation and new design methodologies are under development. Proper application and management of flushing technologies is important so that flushed solids either reach the WWTP or are flushed past an overflow discharge outlet in the sewer network, otherwise flushing accumulated solids downstream may be viewed as just moving the pollutant load downstream, rather than providing for its removal.

Campisano et al. (2007) investigated the scouring effect of flushing waves produced by hydraulic flushing gates. Simulations using a dimensionless numerical model based on the De Saint Venant-Exner equations were conducted to provide indications of the design and positioning of the flushing gates.

Sanitary wastewater SS deposited in CSSs can generate H_2S and methane gases due to anaerobic conditions. Sulfates are reduced to H_2S gas that can then be oxidized to sulfuric acid (H_2SO_4) on pipes and structure walls by further biochemical transformation creating hazardous conditions and sewer degradation. The deposits of these SS are discharged to the receiving water during WWF events causing degradation of water quality. A state-of-the-art report on field experience indicated that sewer flushing by manual means (water-tank truck) was a simple, reliable method for CSS solids removal in smaller diameter laterals. However, the most effective device for removal of settled sediment in trunk sewers was the construction of an in-line flushing gate system (Pisano et al., 1998).

Ashley et al. (2002) reviewed progress on modeling and interpreting the mechanics of in-line retention as a means of combating CSO releases that exceed regulated limits. The buildup of grit and material and associated behaviors in the line and flushing effects at the plant, along with other consequences of buildup along the length of the system was analyzed. They also cited the fact that no truly comprehensive model suitable to represent the range of phenomena observed existed at the time of this research.

Williams (2008) investigated sediment deposition when using a generic flow control device. Test results indicated that the type of sediment load has a significant effect on the nature of the deposit formed. In catchments dominated by the suspension of fine solids, the use of an active flow device results in uniform flat bed deposition directly upstream of the flushing gate. Williams (2008) noted that a single downstream flushing gate may only be suitable in systems with fine material, since fine sediment deposits are more easily entrained in a flush used to control sedimentation.

Tanks

Research also continues toward the evaluation of treatment efficiencies for storage facilities, including tanks and basins. A duality of function exists here as it does for in-line options, in that tanks can provide hydraulic buffering capacity or can provide treatment, but hydraulic effects have in many cases been the dominant concern. In the U.S., the primary use of these structures has traditionally been to store excess WWF volume, as opposed to active development and placement of treatment capability. In Europe, the EU's Water Framework Directive is stimulating additional research into the quantification of treatment efficiencies for these facilities.

The management of filling and emptying cycles for off-line WWF storage tanks in combined sewers was based on different running rules and the analysis was based on numerical simulation of real rainfall series from Milan (Paoletti et al., 2008). Several combinations of parameters for different networks, catchments, and climate scenarios were evaluated, which culminated in three site-specific, running "RULES" of storage. Intercepted volumes, mean annual overflows, and numbers of filling-emptying cycles of tanks were compared and multi-regressive relationships were calculated given the approximations. The researchers noted that the simulation was limited by too few sets of real rainfall data.

Balistrocchi et al. (2008) assessed the long-term efficiency of CSO capture tanks with a new probabilistic rainfall model calibrated using five sets of continuous simulation time series data. Three efficiency indexes, derived by using simplified hydrologic models, were also developed. The sequence of storm events is described stochastically by three independent variables: the runoff volume, rain event duration, and antecedent dry period. These variables are assumed to be independent from each other and distributed as a suitable probability density function. Preliminary evidence from this study indicates that volume and the antecedent dry period cannot be represented as an exponential distribution, so after different distributions were evaluated, the Weibull 3 parameter, a generalization of the exponential distribution was deemed the best fit. This semi-probabilistic model with continuous simulation data inputs was tested on an urban catchment with runoff and overflow results in reasonably good agreement. Because this semi-probabilistic approach requires many simplifying hypothesis, continuous simulations are necessary to verify reliability of the methodology and to support assumptions. One assumption is that the efficiency index for the mass pollution reduction is based on the first flush concept. Simulated overflows were mitigated by buried tanks coupled with overflow devices assuming the tanks are designed to intercept the first part of the runoff event and pump the stored volume into the sewer network for treatment after the storm event. The author stated that this problem fits into the wider context of the estimation of the mitigation gain of the structural sustainable urban drainage systems. However, the assessment of the effectiveness of several design and management considerations remains an issue and further investigation should be continued. An associated difficulty lies in the definition of a design precipitation given the great variability of nonpoint source transport mechanisms.

Schroeder et al. (2008) described the dimensioning of a river-based WWF tank (in-receiving water storage) by means of long-term numerical simulation. This multi-faceted project, started in 2007 and to be carried on through 2021, includes construction and operation of a pilot tank on the River Spree in Berlin. Overflows from the CSS will be stored, treated and discharged from these river-based tanks. The tanks will be equipped with usable platforms and sealed to prevent odor emissions. An innovative concept presented relates to the marketing of the platforms as a method to help finance the storage tanks. Dimensioning analysis initially estimated the effect of the 1,000 m^3 storage tank and pollutant loading of one overflow per year. A model of the sewer system and ancillary structures including a pump station was used to carry out long-term simulations using a 30-year set of rainfall data for two scenarios. The first scenario was based on the current drainage system, while the second is based on a significant system rehabilitation that will increase the in-pipe storage by 100% and is planned to be finalized in 2020. Tank operations relating to peak inflows, average number of annual filling/emptying cycles, average and maximum duration of empty tanks phases and duration of WWF staying in the tank were also evaluated. Model calibration was based on data available from the SCADA system and included both WWF and DWF. Additional temporary measurements were collected from the main trunk line located near the site of the proposed river based tank and used for calibration purposes. Results from the 30-year long-term simulation estimating the impact of using a 1000 m^3 tank to mitigate CSOs reported that it would reduce the average number of overflows by 92% annually. It was also reported that a 2000 m^3 tank is needed to achieve an annual, average recurrence interval of overflow of once per year.

Pisano et al. (1998) evaluated tipping flushers and flushing gates in a detailed examination of 18 facilities in Germany, Canada and the U.S.; both were deemed the most cost-effective means for flushing solids from CSO storage tanks.

Overflowing Tanks as Partial Treatment

The hydraulic design of clarifier-type CSO tanks, also referred to as retention treatment basins, was evaluated by Brombach et al. (2008). These off-line tanks only fill during storm events and have a clarifier overflow allowing excess WWF to discharge to the receiving water. Sedimentation occurs as the fluid slowly flows through the tank. After the event, the remaining flow volume and sludge are discharged to the WWTP. The research evaluated tank geometric proportions and acceptable surface loading rates. The knowledge about the efficiency of the tanks is limited and design standards can be confusing. The standards are based on sound structural and hydraulic engineering practice rather than set water quality criteria. In particular, sedimentation and resuspension of sewer sediments are not fully understood and more research is needed. The paper emphasized that an emergency overflow is needed to bypass flows exceeding acceptable clarifier hydraulic limits which allows discharge of treated volumes. The research reported that a careful matching of both overflows is required. If the clarifier is a Poleni-type weir, it will cause excessive flow and re-suspend settled sludge, so the authors recommended that slot-type overflows or self-regulating clarifiers were more appropriate. The paper also provided historical content by noting that pretreatment of excess WWF not flowing to the treatment works was introduced in the 1970s and consisted of a combination of detention, sedimentation and floatable debris removal. Tanks were designed for first flush; however, retention treatment basins have become more widely accepted as CSO treatment structures. There are now more than 30,000 decentralized CSO tanks in operation in Germany.

A water treatment system was tested under real-time conditions to assess the efficiency of sedimentation tanks using particle tracer tests to examine the separation efficiency within the various columns comprising the system (Maus et al., 2008). Real-time particle size distributions of suspended particles were identified with a submersible field instrument composed of a laser diffraction particle size analyzer with settling columns developed for in-situ observation. The in-situ settling characterization method worked automatically and indicated that particle sizes in the outlet differed from the inlet. The authors suggested that larger mineral particles with higher densities were transported as bed material, whereas organic material particles with much lower densities were transported as suspended load. The system differentiated the total SS into different settling fractions. The ratio of the fraction masses can indicate if a high total removal of SS is based on a well constructed tank or on a great amount of settleable solids. However, it was concluded that the system was limited at estimating the separation efficiency, although it did provide a better understanding of the treatment process. It was also reported that the instability of SS primarily was the result of agglomeration during sample storage. Other settling velocity studies (Dalrymple et al., 1975, and Aiguier et al., 1995) had also noted the phenomenon of differences in laboratory measurements of samples as a function of elapsed time and storage procedures. Dalrymple et al. (1975) discussed development on settling curves and relation to settling devices for WWF, particularly the EPA swirl.

Shepherd et al. (2008) quantified the performance of stormwater tanks providing storage and sedimentation at treatment facilities for excessive WWF. Historically, flows entering European treatment facilities are limited to six times the mean daily DWF by using CSOs and an emergency overflow at the entrance to the WWTP. Approximately half of the inflow, three times DWF, undergoes full treatment and the remainder is discharged to the storage tanks. Once the tanks are full, the excess discharges into the receiving water. After the storm event, the tanks are emptied and the discharge undergoes treatment at the WWTP. The basic functions of these tanks include acting as temporary storage which retain excess flows for later treatment, reducing overflows to receiving waterways, providing treatment by settling resulting in a more diluted overflow from the tanks, increasing the time of concentration such that discharges occur at a later time which may coincide with increased flows to the receiving waterway (thereby potentially providing additional dilution in the receiving stream), and retaining the first flush (if pronounced) from the watershed. To achieve acceptable performance, the tanks require sufficient depth for the settled solids to remain immobilized so as not to be resuspended into the receiving stream during WWF.

In the UK, design guidelines are based on calculated design tank volume, and do not take into account the length to width to depth ratio for rectangular tanks, or the diameter to depth ratio for circular tanks. The guidelines also do not specifically consider the need to retain pollutants in the runoff or to characterize the pollutant retention processes that occur in the tank. It is noted that a tank's geometry may influences a tank's hydraulic performance, and in turn this may influence the pollutant retention processes occurring in the tank. However, the design guidelines do not influence the matching of the hydraulic performance of the tank with its dimensions and miss an opportunity to optimize the configuration for enhancing pollutant removal effectiveness. The aspect ratios' built-in practice must also reflect access, maintenance and constructability preferences; however, guidance on the priorities between these factors and process priorities is not definitive.

Historically, the efficacy of rectangular tanks has been based on surface load clarification theory for an ideal rectangular tank based on sanitary loadings. Simply, the particle settling performance is related to the nature and type of particles and surface overflow rate. Starting with Hazen (1904) and expanded by Camp (1946), the characteristics of an ideal continuous-flow settling basin depends on the selection of V_c, which essentially defines the surface overflow rate or surface loading rate, and is related to V_s, the settling rate which defines the removal efficiency (the proportion of the inflow load that is retained within the tank over the duration the tank is in operation). Germany specifies a surface loading rate of 10 m/hr and a 2 to 1 length to width ratio for sizing a rectangular tank for CSS. In the U. S., the specified loading rates typically range from 0.5 m/hr for small populations for separate sanitary loadings assuming no inflow, to 5 m/hr which includes WWF (Tchobonoglous et al., 2003). EPA (Driscol et al., 1986) has a method to estimate sediment removal under WWF or dynamic conditions based on the original work of Hazen (1904) and Camp (1946).

Full- and laboratory-scale model tanks were evaluated to compare the deviation of the measured residence times from theoretical residence times, assuming idealized plug flow conditions (Shepherd et al., 2008). Testing was carried out at a range of constant flow rates. A fluorescent tracer was used to estimate the true residence time with measurements of tracer concentration on entry to the tank, adjacent to the inlet weir and on exit at the spill weir using a full-scale tank in the field. Results show that the spill tracer is much more lagged and attenuated than the inlet trace, suggesting that mixing processes were occurring in the tank.

The example solute trace showed different definitions of residence time. Although there are numerous techniques to define the residence time from the tracer data, the one selected was the time for 50% of the tracer to pass through the tank, which compares with a theoretical residence time of 65% tracer pass-through (Shepherd et al., 2008). A series of laboratory experiments tested a Froude-scaled tank situated within a controlled and repeatable environment. The design of the lab tank was based on tanks commonly used in the UK and configurable in different geometrical arrangements, including the full-scale tank used in the field. The methods used in the laboratory were the same as in the field, which included the monitoring of a fluorescent tracer at it traveled at constant flow rates through the tank. One advantage of the laboratory testing process was that it provided more accurate control for hydraulic conditions, resulting in less noise in the measurements, greater ability to determine the start and end of the tracer slug, and ultimately, a more accurate residence time calculation. The resulting data showed a power law trend with the discharges, and the measured residence times were always less than the idealized residence times.

An experiment to quantify SS was also conducted using entrained crushed olive stone at a constant rate and sampling at the tank's inlet and overflow location (Shepherd et al., 2008). Crushed olive stone has been used previously as a surrogate to represent wastewater solids. The results indicated that when the residence time is correctly estimated, retention efficiency can be estimated with good accuracy provided classical sedimentation is used. A comparison between the full-scale field and lab 50% residence times show reasonable agreement when flow variations present in the field tests are recognized. In both cases, the 50% residence times are always less than the theoretical residence times. The author suggested that the ratio of the theoretical to measured residence times be used as a correction factor, including estimates for calculating the solids retention efficiencies, provided that such ratios are quantified for a full range of tank geometries at various flow rates. The measurements of the 50% residence times measured in the field tank and in the lab essentially agreed providing confidence that residence times measured for a range of tank geometries can be successfully investigated using Froude-scale models. Storm tanks designed without accounting for imperfect mixing may be over-predicting solids retention.

Analysis Overview

Significant research is underway in the EU to determine the influences of the hydraulic and geometric properties of storage structures on pollutant deposition/resuspension processes regardless of whether the structure is situated in-line or off-line. In-line storage structures are most appropriate in mild slope applications that allow DWF to directly pass.

Active investigations are focused on the mechanics of fluid-solids interactions attributed to the mobility, propagation and discharge of in-pipe sewer solids during runoff events. Recent studies have explored influences of microbial activity on the physical release of in-sewer sediment. Other queries have examined the role of grain size variability and physical factors under increasing shear conditions. Hydraulic flow responses relating to particulate size distributions of both suspended and accumulated sediments are continuing although comparisons of flow characteristics and turbidity measurements collected continuously over a one-year period were unable to detect a predictable relationship (Lacour et al., 2008).

A structure's geometry can greatly influence hydraulic properties, including losses, conveyances and efficiencies. Rectangular tanks and oversized circular pipes are often used for in-line WWF storage. Milder slopes can lead to increased sedimentation and a decrease in hydraulic conveyance if the solids continue to accumulate over time. In-pipe solids can be re-suspended during WWF events. Mixing effects in highly turbulent conditions are a secondary effect to consider. Given the same volume, and assuming that other variables are held constant, a circular pipe would need to be configured longer and shallower than a tank in order to perform the same function.

Innovative measurement techniques to estimate sediment profiles in pipes are also under development. Improving techniques to measure sewer solids bed material, in addition to the real-time data capture of hydraulic properties, suspended pollutant data, and in-pipe bio-processes, together provide a more dynamic analysis of in-pipe fluid-solids processes. Investigation into sediment characteristics, such as range of particle sizes, concentrations and chemical/biological attachments are ongoing.

The EU is also directing further scientific inquiry into enhancing treatment capability and efficiency of storage tanks through its Water Framework Directive. This research includes using real-time SCADA systems to optimize system storage and tank cycle operation sequences to reduce the annual average recurrence intervals of tank overflows. Long-term simulations indicate high reductions of overflows can be achieved.

Current research has indicated that the matching of hydraulic properties of the tank's geometry with its length-to-width ratio if rectangular, or length-to-radius ratio if circular pipe, are important design components. Other significant influences include access, maintainability, constructability and preferences. Correction factors for residence times (ratio of theoretical to measured residence times) for tanks and circular pipes are still being investigated in the laboratory.

Despite the extensive literature that surrounds the topic of sedimentation, there remain elements that are not yet fully understood or amenable to analysis. Kutzner et al. (2007) stated there is a lack of knowledge about the treatment efficiency of differently designed sedimentation tanks. The paper also noted that no deterministic model is available to simulate sedimentation tanks with acceptable reliability. Support for the notion that transferability of sedimentation tank design remains a somewhat intractable analytical problem can be found in Luyckx et al. (2005), who identified issues of transferability of removal efficiency from physical models to large-scale tanks; continuing hydraulic research, predominately in the EU, attempts to define fluid-solid dynamics found in in-line systems.

Comparative Costs of Tanks vs. Tunnels

One element of combined sewers that has been effectively measured and quantified is the actual construction cost of both tanks and tunnels. It is interesting to consider the problem in this light, although it is not possible to predict with confidence the costs of tunnels as opposed to tanks for several reasons. The site conditions and context have too big an impact on the final costs to make a meaningful assessment of cost tradeoffs possible without specific reference to a particular site. As pointed out by Feroz et al. (2007), there is a range of alternatives for tunnel implementation, including drill and blast, mechanical excavation and use of a tunnel boring machine. In cases where tunnels are smaller and shallower, cut and cover methods can be used. Whatever the case, the point raised is one amply born out in common experience, which is that site conditions have a major impact on the best option for laying a tunnel and on the costs of that effort.

VanWeele (2009a) noted recent experiences in industrial contexts around the globe, and supported the notion that variability in site conditions, local capability and project circumstances can have a major impact on implementation costs. However, he also noted that experience has shown that general relationships between tunneling and cost can be devised. Figure 1 provides an indication of trends in unit cost against tunnel size. As noted by Feroz (2007) the impact of ground conditions is clearly a significant determinant of unit cost, as is the size of the tunnel itself.

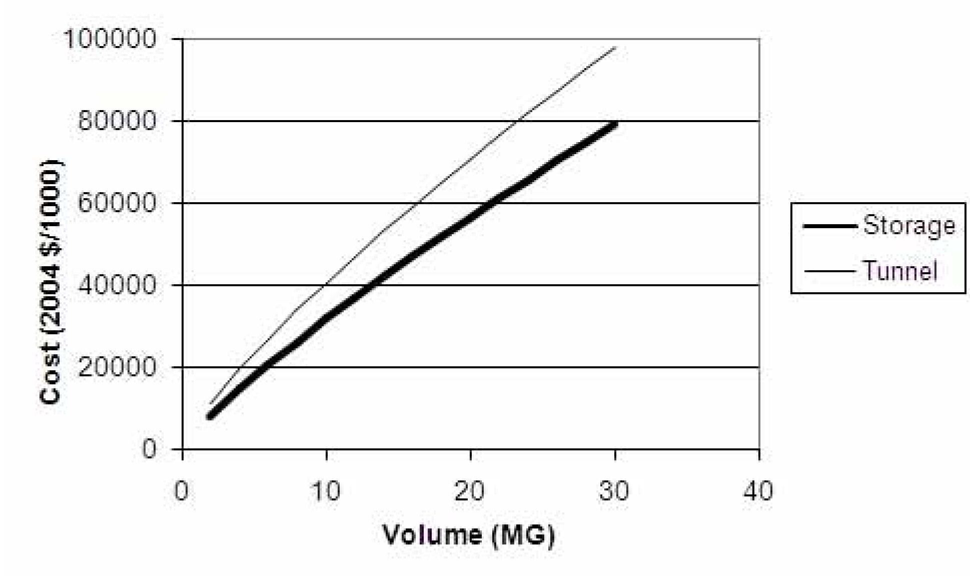

Figure 1 Recent experience in global tunnel costs

Also provided by VanWeele (2009b) was information originally derived from literature sources (e.g., Engineering News Record) and assembled to provide a broader view of tunnel costs. This content, shown in Figure 2, indicates that there is a relation between diameter of the facility and cost, but that for smaller diameter facilities, the unit cost relationship changes.

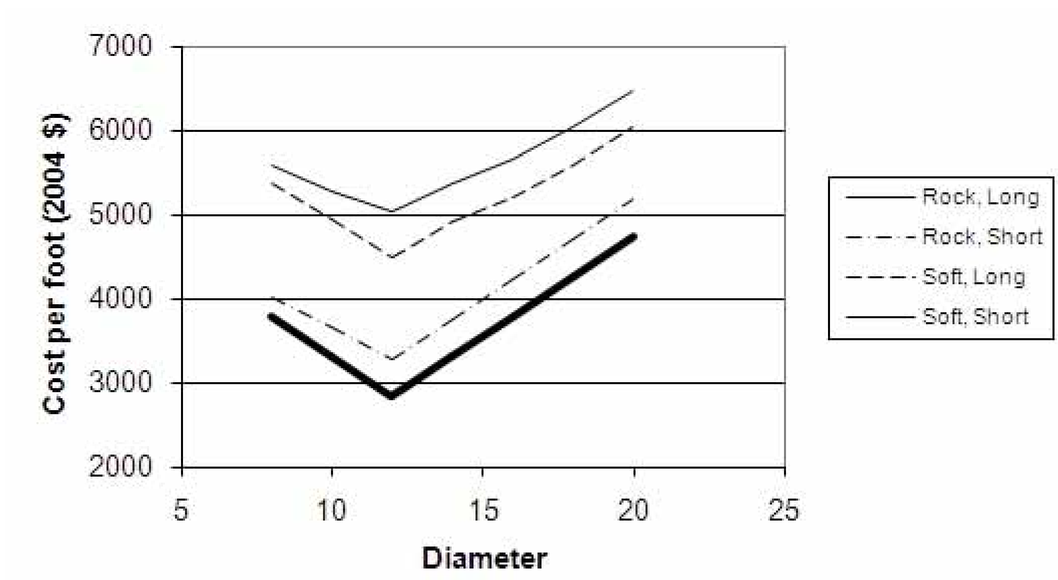

Figure 2 General relations between tunnel unit cost and diameter for various lengths and ground conditions

Also clear in this representation is the fact that economies of scale are present (longer tunnels cost less per foot than short tunnels) which is a logical outcome of the cost of mobilization and demobilization, and possibly of the greater relative importance of disturbance at entrance/egress points. The impact of ground conditions is visible in this figure as well, with hard ground less costly for construction purposes than soft ground (because of the greater difficulty in excavation and lining). Associated costs of land and other relevant factors important to determining the net outcome are not visible in this representation, but the general relationships illustrated provide some useful insights into costs.

Costs in Figure 2 are generic and while Figure 1 costs, borne of recent experience, are at least a magnitude level of agreement, particularly in the smaller size range; however, for larger projects, costs are typically larger than those developed based on generic cost information.

In considering the problem at hand, the costs of storage are also important, as they need to be compared to tunnel costs if conclusions are to be drawn regarding these two options. Heaney et al. (2002) and Sample et al. (2003) completed an excellent review of cost estimation methods in the context of urban drainage, and suggested an approach to generating cost estimates that is sensitive to local considerations and applicable for specific cases. In doing this, they also conducted a literature review that provides insights into cost factors. Two relationships, from Gummerman et al. (1979) and Walker et al. (1993), provide quantitative relationships between volume and cost for tunnels and storage units, respectively. These are shown in Figure 3. The curves are normalized to 2004 dollars for consistency with the rest of this report. They are illustrative rather than definitive, but they do at least illustrate two representations of cost as a function of storage. This information is of limited inherent value, but can be extended for exploratory purposes.

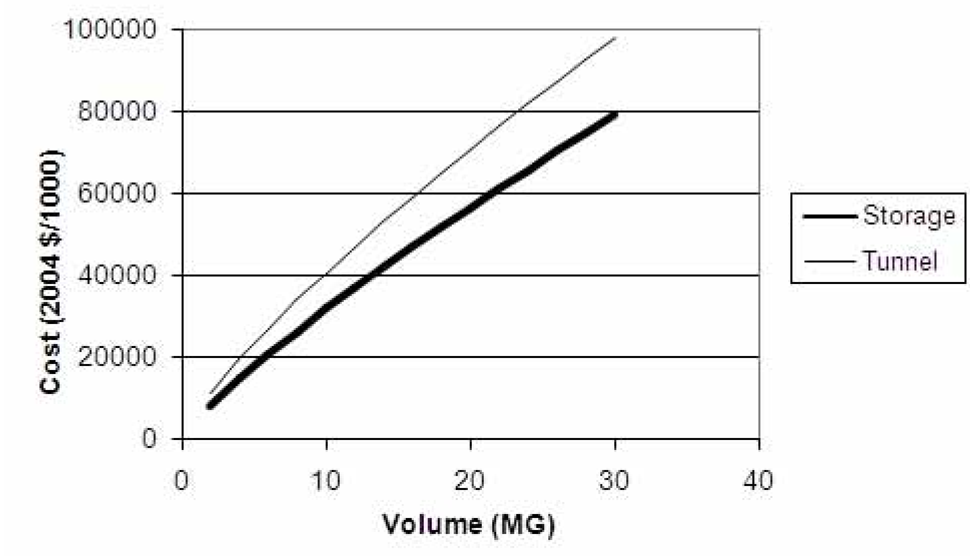

Figure 3 Cost curves for storage and deep tunnels

It is possible to apply relationships (Gummerman et al., 1979) for cost, normalized to 2004 and converted to a cost function against length for any particular tunnel diameter. For purposes of illustration, this is provided in Figure 4 for a 3 m (10 ft) diameter tunnel. In developing this curve, the original relation was normalized to 2004 dollars by applying a cost factor obtained from Sahr (2004). This curve differs from the preceding information (roughly comparable to the empirical basis of Figure 1 and somewhat comparable to the generic curves in Figure 2), and is only one of a family of curves dependant on diameter, but it is nevertheless useful. Given the length of time since the original derivation and the variability of site conditions, this curve can be viewed only as a rough approximation, but it can be used to develop further insights into the tradeoffs between storage and tunnel storage.

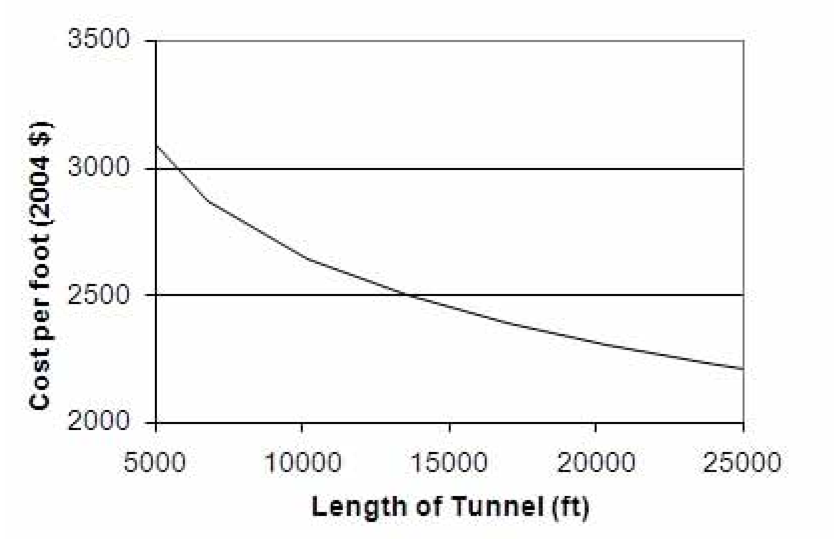

Figure 4 Cost per foot for a ten-foot diameter tunnel

This was done by applying the two relationships included in Heaney et al. (2002) based on earlier work. Without attempting to explore all the cost implications and possible cost models to this problem, something beyond the limits of the present evaluation, it was possible to consider the implications of the basic relationships between cost for storage and for tunneling. From this source, the relevant cost functions, adjusted to 2004 dollars, are:

$$Cs = 4699Vs^{0.83}$$

where

$Cs =$ storage cost, 2004 dollars

$Vs =$ storage volume (MGal)

and

$$Ct = 6437Vt^{0.80}$$

where

$Ct =$ tunnel cost, 2004 dollars

$Vt =$ tunnel volume (MGal)

It immediately follows that a relation between tunnel cost and storage cost exists, as follows, by equating the tunnel volume and the tank volume:

$$Rts = 1.37V^{-0.03}$$

where

$Rts =$ ratio of tunnel cost to storage cost

$V =$ stored volume (MGal)

The relationship which emerges is as shown in Figure 5[3]. For a given volume, in the size ranges shown and using the cost curves that have been reproduced in Heaney et al. (2002), there is apparently a significant but not overwhelming tendency for tunnels to be more expensive than a storage solution for an equivalent volume.

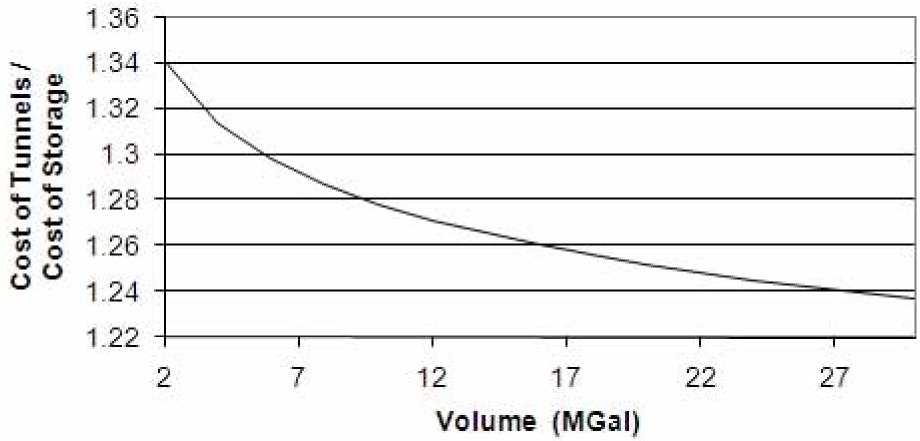

Figure 5 Relative cost of tunneling and storage

This interpretation must be treated with caution, as it does not allow for variability in land costs, antecedent conditions or other factors:

- The magnitude of the cost differential is modest compared to the variability that can occur from, for example, soil conditions. The cost differential in Figure 5 ranges between 34% for smaller volumes and less than 25% for larger ones. In contrast, reference to Figure 2 suggests that the difference between tunnel costs in soft ground and rock can approach 100% depending on tunnel size. In a similar vein, the empirically-based trend shows differences of about 100% for smaller tunnels, reducing with tunnel size. If this apparent finding is reflective of general experience, then the cost distinction between tunnels and storage on a volumetric basis is in some cases eclipsed by other considerations.

- It is also interesting to note that the form of the curve in Figure 5 suggests that the cost differential increases for smaller tunnels. This seems reasonable and consistent with the left-hand side (smaller tunnels) of Figure 2, which shows that tunneling unit costs increase as smaller sizes are considered. Again, numerous factors will work to affect this picture one way or another, but the basic sense of the information is reasonable.

In many cases the problem does not relate to a single purpose storage tunnel as compared to a storage tank. The case of most interest in the CSO context is when urban development dictates the tunnel alignment, and the storage function is accomplished by increasing the diameter of a conveyance tunnel; this dual purpose of conveyance and storage may justify comparative cost, not analysis of a single purpose storage volume in place.

In that event, the end area of the pipe is increased from that which is needed for conveyance to that which is needed for storage. The ratio of end area of a conveyance tunnel compared to end area of a combined conveyance/storage tunnel will depend on the design wet-weather carrying capacity, site conditions, and the design retention volume. Both features are very site-specific, but what emerges is that for a tunnel, the cost of storage is the cost of an incremental increase in volume, not the total cost of the tunnel.

---

[3] It is noted that the form of the curve in this figure depends on the value of the exponent and is therefore sensitive to the significant digits in the underlying cost curves. This does not affect the basic conclusions but should be considered as these results are interpreted.

It is possible to relate the incremental cost of the tunnel to the cost of an equivalent volume of storage. This has been done in Figure 6, which shows a family of curves that relates the relative cost of a storage/conveyance tunnel to the cost of a storage tank. The abscissa axis indicates the volumetric capacity of a tunnel, and the ordinate axis the ratio of providing added storage cost in the tunnel to providing added storage cost in a tank, for each curve of added storage volume. In the example shown, if a pipe required for transport initially has a volume of 16 MGal, and an added storage volume of 4 MGal is required, increasing the size of the conduit to obtain that added volume would cost 50% of what would be required to add a storage tank providing the same added volume.

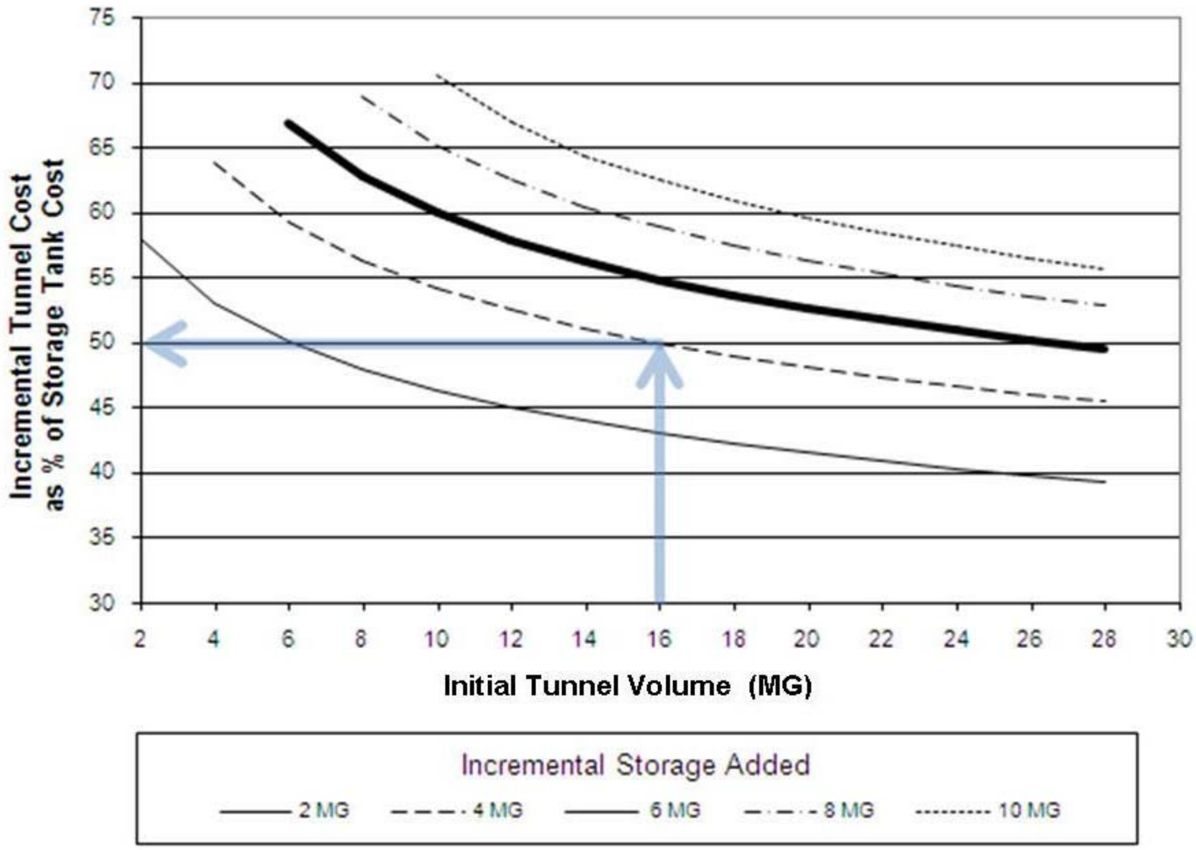

Figure 6 Relation between cost of incremental tunnel storage and cost of a storage tank

More generally, the relative increase in the cost of providing storage by increasing the tunnel size is less than the cost of adding that same storage by placing an equivalent tank. It is also noted that as the size of the incremental volume goes up, the relative cost of adding storage goes up accordingly. Again, numerous factors can materially affect this picture, including site-specific conditions, the volume of the flow hydrograph and so on, but the general point that the cost of increasing the conveyance structure will have different implications than adding a new discrete storage seems reasonable and merits consideration in practical cases.

It is also noted that in every case considered in this simplified analysis, the cost curves being compared indicate a different picture from what is found if an off-line facility is considered. The tendency is for an off-line storage tunnel (one not inherently used for conveyance) to be greater than for a tank, and the cost of an on-line storage tunnel (one used for conveyance when excess volumetric capacity is not used for storage) to be less than that for an equivalent tank. In either case, however, the magnitudes are similar and other factors could either increase or reduce this tendency markedly.

Either way, it is noted that the cost comparison for the tunnel is true for primary construction only, as the costs for replacing a tunnel that is part of the existing infrastructure involves removal costs, staging expenses and other factors that can cause cost estimates to depart markedly from situations where no prior construction has existed.

Topic Area 2 Conclusions

Overall, the principles involved in tunnel storage and tank storage are generally understood, but there are numerous aspects of design and operation that have yet to be researched to the point where blanket assertions as to the "best" approach can be made. Case by case considerations will still have to be evaluated in order to determine the best course of action. This will also be true regarding costs as well, although the available information demonstrates that general trends can be used as an approximate guide, provided the variability that results from differing local conditions is recognized. It is interesting to note that the general trend appears to be that on-line conveyance/storage tunnels are economically preferred to tank storage in undeveloped placements. More generally, it appears that for off-line systems, tunneling will tend to a higher relative cost, while in on-line systems, tunneling will tend to a lower cost.

The principal driver for selection of one method over another, therefore, would seem to be the ability to preserve future flexibility. If an over-sized tunnel is placed, it can be electively used for storage, conveyance or both by changing operating characteristics. Trying to implement such a tunnel after the fact implies dealing with numerous interconnections and constraints that greatly exacerbate the problems inherent in placing such a structure. Further, an oversized tunnel will tend to enable greater latitude in capture rates, implying an ability to capture, control and move higher multiples of average WWF. So the preferred course of action from the perspective of future-proofing the facility is opting for the on-line storage/conveyance tunnel. Tank storage is a useful remedial alternative, but from at least this analysis perspective, may not be the preferred solution.

Chapter 7 Topic Area 3: Sedimentation and Disinfection Tradeoffs in Primary Treatment

Microorganisms

There is a variety of microorganisms of concern in surface waters and wastewater. Bacteria are unicellular microorganisms that play a fundamental role in the decomposition and stabilization of organic matter in nature and in biological sewage treatment processes. Bacteria can range in size from 0.4 to 14 µm in length and 0.2 to 1.2 µm in width. Many types of enteric pathogenic bacteria occur in wastewater and are also found in water supplies. Enteric waterborne bacteria are known to cause gastrointestinal illnesses with common symptoms of diarrhea, nausea, and cramps. Some more severe infections from pathogens are spread through the body from the intestinal mucosa and cause systemic infections known as enteric fevers (e.g., typhoid fever). Enteric pathogenic bacteria transmitted by water and wastewater include *Campylobacter*, *E. coli* O157:H7, *Leptospira*, *Salmonella*, *Shigella*, *Vibrio cholerae*, and *Yersinia entercolitica*. *Legionella pneumophilia*, while not enteric, is a pathogenic bacteria distributed in the aquatic environment (Perdek et al., 2003).

Protozoa are one-celled microorganisms which live in many animals and survive in cysts (protective shells) when outside of an organism. These microorganisms vary in size from 2 to 100 µm. Protozoa reproduce rapidly inside a host organism, so ingestion of only a few can cause infection. Protozoa can survive for several weeks in water, and even longer in ice. The waterborne pathogenic protozoans of greatest concern in North America temperate climates are Cryptosporidium and Giardia. Oocysts of Cryptosporidium and cysts of Giardia occur in surface water. Oocysts and cysts are both very persistent in water and are very resistant to disinfectants commonly used in drinking water treatment (Perdek et al., 2003).

Viruses are infectious agents that require a host to replicate by using the host cell's reproductive mechanism. After replication, and subsequent death of the host cell, viral particles are spread to neighboring cells resulting in infection to the host organism. Viruses are the smallest and most basic life form, ranging in size from 0.02 to 0.09 µm. Lipoprotein, virus protein covering, allows some viruses to survive for long periods of time outside host organisms. Enteric viruses which are found in the gastrointestinal tract of infected individuals are of greatest concern to water quality. These viruses are excreted in the feces of infected people and may directly or indirectly contaminate water intended consumption or contact (Perdek et al., 2003).

Surface waters are tested for indicators that serve as a proxy for harmful pathogens during contact recreation to protect public health. The concentrations of these indicators are used to determine the potential for fecal contamination and to compare to public health-based thresholds. Indicator bacteria are used because it is difficult to measure the variety of specific pathogens themselves, due in part to timeliness, labor, expense, complexity, and analytical limitations. The most common indicator bacteria microorganisms tested by public health agencies include fecal indicator bacteria such as total coliforms (TC), fecal coliforms (FC), fecal streptococci, *E. coli*, and enterococci. Like the pathogens they represent, fecal indicator bacteria are found in feces from both human sources (e.g., sewer discharges, and failing septic systems) and non-human sources (e.g., pets, waterfowl, and farm animals) (Struck et al., 2006). Historically, total and fecal coliforms, as well as fecal streptococci, have served as the preferred indicators, i.e., EPA had recommended that states, as a bathing/contact WQS, not allow fecal coliforms to exceed 200 organisms/100 ml. However, there are efforts to substitute enterococci and *E. coli* for water quality monitoring because of higher correlation with gastrointestinal illness (Gray, 2000). EPA (1986b) has recommended that states revise the recreational water quality microbial criteria to use enterococci for marine waters (35 enterococci per 100 ml) and *E. coli* or enterococci for freshwater bodies (33 enterococci per 100 ml and 126 *E. coli* per 100 ml, respectively). Suggested criteria are 35 enterococci per 100 ml for marine waters and for freshwater bodies (EPA, 1986b). If a single sample exceeds 235 *E. coli* per 100 ml in freshwater or 104 enterococci per 100 ml in saltwater, the EPA recommends that a swimming area be closed, or a warning be posted until levels are lower. Many states continue to use the traditional indicators for a variety of reasons, particularly FC, because of its historic use; however, several states have established policies that post advisories at more protective levels of indicator bacteria (Struck et al., 2006).

Microorganisms in Wastewater Systems

To evaluate the impact of primary treatment and disinfection, a variety of factors need to be considered. At an elementary level, the more sediments that are removed, the more efficient the disinfection, as less disinfectant reacts with sediments. Furthermore, particles occlude and carry bacteria, and might therefore be a source of post-disinfection bacteria. This occlusion factor will be included in the modeling analysis as one of the primary decision factors, and needs little further discussion here. Secondary effects, however, may also be important as bacteria are known to move between fluid and solid phases of a solution, which has an effect on disinfection efforts. The following discussion provides a brief review of some factors relevant to these secondary effects.

Figure 7 provides a simple representation of one concept of varying bacterial grouping modes in a wastewater stream. As indicated, bacteria can be free and independent, clumped in groups or associated with a substrate (a fluid or solid distinct from the medium in general) either on the surface or embedded in the matrix.

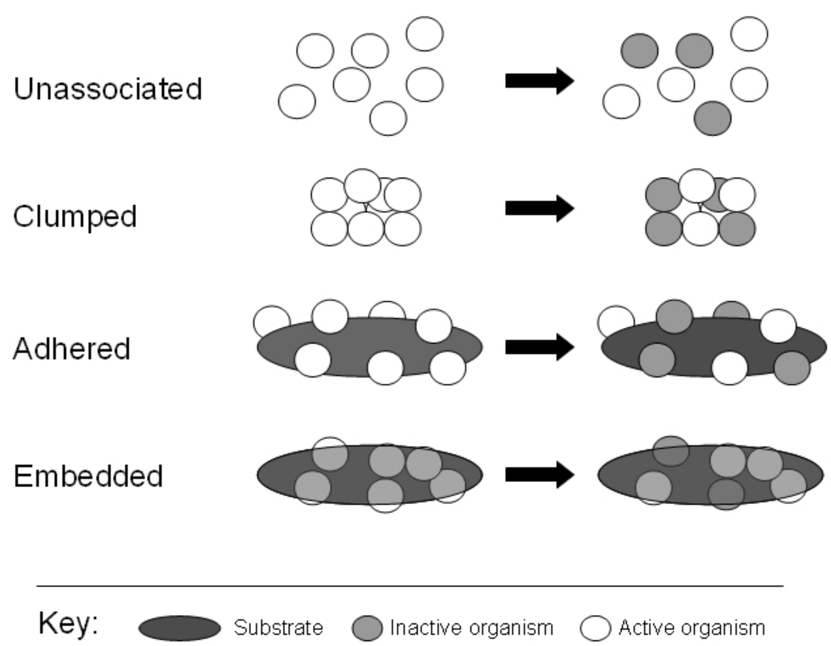

Figure 7 A simple representation of bacterial grouping modes

This concept has a key implication when it comes to predicting the movement of microorganisms through the wastewater stream. Those associated with a high density particulate substrate, for example, will be transported with the stream differently than those that are free-floating and independent. The various ways microorganisms may be present also can have an impact on measurement, since typical detection and quantification methods may measure clumped and single organisms differently. Since the microorganisms may be active or inactive, the ways they associate can have an impact on the interpretation of numbers in the sample. Microorganisms that are freely present, for example, may be easier to culture and count as distinct individuals than those that are clumped. Agitation or other actions that can break up a clump or separate microorganisms from the substrate can have an impact on the numbers counted. Basically, the state of the microorganism population in the waste stream and the ways that they are sampled and analyzed can all have an impact on the numbers that are inferred.

The illustrative image in Figure 7 is vastly simpler than reality. The microorganisms involved are highly varied in their forms and behaviors. Useful examples of this can be readily found on the internet.[4] As noted, bacteria have several different means of moving independent of fluid currents or other physical forces (gliding, using flagella, or a spiral motion). Different types of microorganisms will behave in different ways as physical forces and their own actions dictate. Furthermore, the physical/chemical makeup of the waste stream, and the existence of predators will have a pronounced impact on the lifespan of microorganisms. Gannon et al. (1991b) noted the transport of bacteria through a porous medium can be influenced by salinity as well as by the medium characteristics. They suggested that the transport of bacteria can be affected by the chemical composition of the carrying solution. Ghayeni et al. (1998) found a similar relationship between the adhesion of wastewater bacteria (three genuses of Pseudamonas) to reverse osmosis (RO) membranes, in that low ionic strength (deionized) water led to minimal adhesion but that increased salt concentrations increased adhesion. The state of the bacteria themselves can also have an impact on adsorption, as noted by Camper et al. (1993) who found that motile, non-motile and starved *Pseudomonas flourescens* demonstrated differing adsorption rate constants. Beyond the variability of the fluid composition, the substrate relationships that are possible within the wastewater stream are numerous, since the substrate components that exist are themselves numerous and varied. Camper et al. (1993) demonstrated a difference between sorption rates on polycarbonate and glass. Other work by Searcy et al. (2005) showed that Cryptosporidium oocysts were removed from the fluid column much more rapidly in the presence of suspended sediments and the rate of oocyst sedimentation depended primarily on the type of sediment with which the oocysts were mixed and not on the background water composition.

Microorganisms are not merely passive particles, but use biofilms, as a survival mechanism, to join together to form a community. Bacteria produce glycocalyx, an excretion of adhesive slime. Glycocalyx formation is a polysaccharide matrix that plays a role in protecting bacteria directly, while also allowing bacteria to adhere to other bacteria to form communities or to solid surfaces. It also has an impact on indicator bacteria susceptibility to environmental factors, and even in apparently pristine or near pristine waters it has been observed to have a pronounced impact on indicator bacteria presence and growth (Hunter et al., 2006). At a macroscopic level, this phenomenon is associated with the biofilm upon which some treatment processes depend. In rotating biological contactors (RBC) it has been observed that the adhesion of bacteria by means of this mechanism is the major mechanism for removal, greater than actual dieoff (Tafwik et al., 2004). Skraber et al. (2007) reported that biofilms within WWTPs can capture enteric microorganisms present in the fluid phase and as a result, their fate may change depending on whether or not interactions with biofilms occur.

Fully defining the behavior of bacteria in the highly heterogeneous conditions characteristic of a wastewater treatment facility, and the linkage of that behavior to engineering design, is challenging. The association of bacteria and sediments, and the interactions between them, suggest the utility of considering a multi-phase model for use in representing bacteria in a treatment plant. The nature of the glycocalyx or the chemical dependence of isotherm associations suggests that a reversible or variable association may be a common result.

The utility of this effort depends not only on the ability to predict behavior, but on the ability to implement methods

[4] See for example http://www.microbiologybytes.com/video/motility.html.

that promote control of the disinfection process to a useful degree. This control appears to be advancing. Williams (2004), for example, explored methods of evaluating chlorine dosages that suggest a meaningful degree of control can be imposed on the chlorine dose in a wastewater stream in terms of measured chlorine residual. This research suggests that dosages could be stated not in terms of a wasteload-proportioned amount with a margin for variable losses, but in terms of an active residual amount. Other parameters, such as turbidity, have been tractable for years and can be measured as needed. Collectively, it is reasonable to conclude that if a model of behavior can be devised, it should be possible to actively manage sediment/disinfectant dosage simultaneously and thereby control treatment based on these parameters.

Association with sediments and other particulate matter has an impact on the way that residual bacterial biomass, which remains with the settleable material, is handled or treated. Recent publications (WERF, 2006; Higgins et al., 2006; EPA, 2006; Qi et al., 2004) have noted that residual sludge, which had a low concentration of indicator bacteria such as FC prior to dewatering, were found to have a high concentration after dewatering. An interpretation of this finding was a presumption that the bacteria were in place in the sludge and viable, but in a state that could not be cultured. This notion is not new, as it has been suggested in earlier publications (Roszak et al., 1987). Other hypotheses (found in the above references) for the reemergence of viable bacteria include a suggestion that there may be a growth-inhibiting component in the sludge that is removed by dewatering, and that there may be some repair of bacteria which had been injured by the treatment process but not totally eradicated. Growth per se seems unlikely because of the time scales involved; it appears that enumeration methods or media were not the cause of this increase in numbers. In fact, mechanical shearing of the media was not able to reproduce this effect directly. However, shearing may have an indirect impact, and one suggestion is that the shearing of the media released constituents that enabled their reappearance by stimulating reactivation. Whatever the cause of this phenomenon, the transport of indicator microorganisms and pathogens in the wastewater treatment process is a complex process not fully understood. This underlines the need to review treatment processes and disinfection alternatives at a level that includes the biological responses of the bacteria in question.

This association of bacteria with sediments has been observed in the collection system (the problem of sewer sediments is discussed in greater context in Chapter 6). Leung et al. (2005) studied a physical sewer model to evaluate the effects of detached/resuspended solids on the bacterial activity in wastewater flows. Two flow regimes were evaluated, one without sewer sediment and the other with filtered sewerage flow with sewer sediment. The first scenario evaluated the effect of solids settling and the second examined the result of purely resuspended solids. Research results noted that solids originating from the resuspended bed material exhibited higher bacterial activity than the solids originally present in the sewage stream.

Bacteria respond to more than just the physical and chemical conditions around them. They are also sensitive to other microbiological populations. Loge et al. (2001) observed bacteria losses in the activated sludge process were greater than would be expected due to endogenous decay, from which they hypothesized that some other factor such as micro predation may be reducing concentrations. They also suggested that it may be appropriate to explore other species and their behavior if the losses of indicator bacteria are to be understood.

Korich et al. (1990) provided a review contrasting different disinfectants on Cryptosporidium parvum. Their work highlighted the very different ways that microorganisms can react to disinfectants (ozone, chlorine dioxide, chlorine, and monochloramine). Cryposporidium oocysts were found, under similar conditions, to be 30 times more resistant to ozone and 14 times more resistant to chlorine dioxide than Giardia cysts.

Blum (2005) demonstrated that different biological treatment methods produce physiologically different coliform communities that vary in sensitivity to disinfection. His work also supported the notion that FC resident in chlorinated secondary treated wastewater could be resuscitated and comprised a significant fraction of the coliform community.

Darby et al. (1995) evaluated options for optimizing UV performance and compared those options with chlorination as a disinfectant. The researchers argued that UV was preferred to chlorine for a range of reasons, and that the usage of UV in the disinfection process would therefore increase in the future in contrast to chlorine. However, UV does not penetrate all large particles, and trends in energy and other factors may affect this prediction. Chlorine remains an important primary disinfectant.

Dietrich et al. (2007) modeled disinfection of particulate-embedded microorganisms in wastewater, and noted that slower reacting disinfectants (e.g., chlorine) were more efficient in disinfecting particulate-associated fractions of the microorganism population than faster reacting disinfectants (e.g., ozone) because of the reduced losses of the disinfectant in reactions extraneous to the target reaction. They indicated that longer contact times at a constant concentration than are presently generally accepted would be required to achieve disinfection of particulate bound microorganism populations. In related research, Dietrich et al. (2003) also compared the physical mechanisms of UV disinfection and chlorination, and noted that UV efficacy is probably confined to macropores, whereas chlorination can penetrate past the limits of macropores through a network of pathways to micropores that lead to dense cellular regions.

This literature suggest a reactivation of microorganisms post-disinfection are indicative of the release or emergence of fractions of population that were not actually disinfected because of location during disinfection period within protected areas of particles.

Microorganisms and Model Concept

Microorganism Behavior

The review of pathogens and indicators in wastewater treatment (Chapter 7) makes it clear that the behavior of bacteria in the context of operations is considerably more complex than simple fluid transport coupled with removal by die-off/disinfection. In fact, there are numerous bacterial characteristics and environmental factors that could have an impact on the balance of bacterial removal and survival through the treatment train. However, what is sought in this work is not a fully comprehensive model of behavior, but one which represents the potential effects of primary settling on disinfection. Disinfection by definition is a process of pathogen inactivation, but for present purposes this was equated to the ability to remove a representative indicator microorganism. Evaluation of indicator organism behavior rather than pathogens per se, is a common practice, and the viability of the notion of indicator organisms was not challenged in the present research. A model designed to examine indicator bacteria removal as a function of primary treatment and disinfection must at least deal with fluid flow, settleable material concentration and bacterial concentration; the challenge is to develop a model that represents both these factors.

Without attempting to summarize the literature review, some implications associated with developing such a model can be identified. As shown in Figure 8, even an elementary model which is intended to evaluate the mutual impact of sediment removal on microorganism inactivation, could imply multiple phases (e.g., free in the liquid, clumped in groups in the liquid, and bound to a substrate as shown).

It is immediately apparent that there would actually be more than one solid substrate phase, but for immediate purposes only a single phase will be considered. Given a single solid phase, there would be at least three relationships to build into the model, which would represent the movement between those phases. Presumably these would be reversible and in some way, gradient dependent. Added to this is the removal of active bacteria from the system by the action of the disinfectant. In deference to current findings in the literature, these are shown in Figure 8 as three reversible pathways, but for practical purposes it is possible that disinfection can be represented as one way (irreversible) processes.

Even without other complications, this leaves us faced with a model that demands development of several equilibrium or transformation relationships and associated rate constants if a simulation is to be attempted. The migration between phases must be described, as must the rate of inactivation in each phase. This is challenging, because the ability to develop rate constants for this purpose is presently limited, and at the outset of this analysis it is recognized that future experimental work will be required to develop that kind of information.

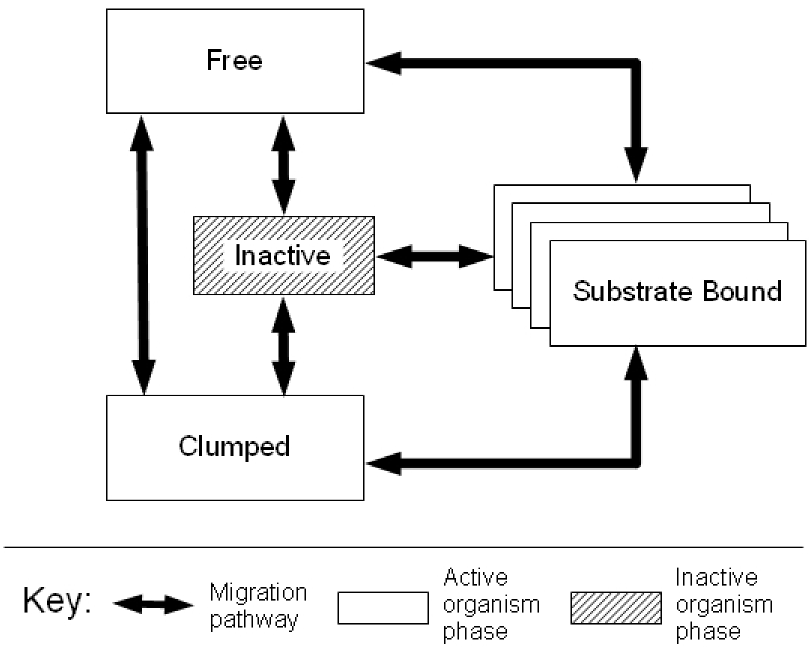

Figure 8 A concept of indicator organism mobility between phases

The phase transfer problem is not the only issue in this conceptualization, because other complications do exist. The heterogeneity of the population, variations in water chemistry, variations in substrate candidates, environmental variables (e.g., temperature), and temporal and spatial variations are all potential factors that affect rate constants and models describing microorganism fate and transport. These various component systems would be coupled and potentially non-linear, and such a model would be complex and perhaps intractable given the present state of the art. Further, indicator bacteria represent a composite phenomenon or surrogate parameter, so the identification of perhaps many dozens of parameters would be required for the successful application of such a tool. Resolving such a model if it contained significantly non-linear terms could be a substantial numerical challenge in practical contexts. This has been considered before, and there are examples of models which cope with complexity in a variety of ways (e.g., Burns, 2000). A collection of some publicly available technologies relevant to the problem at hand can be found at http://cfpub.epa.gov/crem, and guidance into some ways of evaluating model choices in general can be also be found (e.g., EPA, 2009)..

In the face of the complexity of the problem at hand, choices have to be made for a viable model to be proposed and developed within the scope and scale of the present research project. The question therefore is to consider what degree of complexity is required for the present purposes, namely to resolve the behavior of bacteria as a function of primary settling. It was decided that for the present work, the simplest model that represents the phenomena of interest was a logical next step. The set of relationships adopted was as follows:

1. A solids phase is represented as a discrete settling constituent which can be visualized as grit or larger distinct particles that are readily removed during primary settling. Finer solids materials are taken as elements of the fluid phase at this stage of model development.

2. Bacteria are represented as an undifferentiated constituent, partitioned between the solid and liquid phases by means of a reversible adsorption process.

3. Disinfectant behavior is represented using standard disinfectant dose/response relationships.

This stops one step short of the concept illustrated in Figure 8 in that bacteria in the fluid phase are considered to be undifferentiated, rather than clumped and separate, or even possibly associated in other ways. Instead, it represents a single minimum step beyond a model based solely on settling and disinfection.

It is not proven that this is the definitive next step in model development, but this is a supportable and reasonable approach. This choice was based on the data limitations that presently do not quantify or even prove the equilibrium between clumped and separate bacteria in the fluid phase, or for that matter in other intermediate forms. It is arguable whether any kinetic model of bacteria association/dissociation can be fully defended, but to make this work meaningful, at least one phase association (solid/liquid) had to be represented. By limiting the model in this way, the development of a model is mathematically straightforward, and is a direct and immediate extension of common models. The mathematical representation is readily addressed since the functional relationships necessary to achieve this emerge at once from first principles. Primary settling is a commonly understood treatment process; however, empirical support for equilibrium association/dissociation is more challenging, but the literature suggests there may be a basis for an empirically-based representation of parameters. Gannon et al. (1991a), for example, provided measured values for adsorption of *Pseudomonas fluorescens* on glass and other materials.

As well as an equilibrium relationship, a loss model is required. For purposes of this work, a concentration-dependant loss model can be considered. This is certainly not without precedent. Any competent text on unit process models contains examples of this approach (e.g., Tchobanoglous et al., 2003), and other relevant models can be cited. For example, a model which coupled sediment and bacteria by means of an irreversible absorption model was effective in its chosen physical context (Jamieson et al., 2005a). Even simpler models have also been shown to be effective, with a prime example being the application of a dynamic version of QUAL-II which treated FC as a solute subject to first order decay. This simulation produced good results simulating dynamic FC behavior in an urban stream (Rowney et al., 1982). Despite the fact that the problem is known to be extremely complicated, there is ample precedent for the use of simplifying assumptions and mathematical representations of inactivation processes. These simplifying approaches lead to a “black box” approach; however, pursuit of a more complicated equilibrium model may offer clues to WWTP upset, if these variables can be defined. This is the classic tradeoff between increased monitoring cost and perceived limited return on investment which leads to operational status quo.

Overall, this approach enables development of a model that has the necessary components and can be expected to be functional in the event that data are developed. Even so, the nature of this phenomenon is such that variability of parameters is substantial. As a result, in general applicable findings will not be immediately possible. Instead, application of the model to any particular site will require the use of testing which develops characteristic results for a particular physical/chemical set of conditions and treatment facility scale.

Treatment System Components

A second general consideration is to define the treatment system in which the above processes will be represented. A model was developed that incorporates two distinct unit process elements, as shown in Figure 9.

The two unit processes consisted of separate volumes, one for primary removal and one for disinfection, with reactions represented as they exist in the two individual units. It was intended that the system would be set up so that the net effect of removal in the two units could be evaluated. This, too, was based on the EPA interests that gave rise to this project.

Two processes implied that two further choices were to be made for simulating this process model. One was to decide on the nature of primary settling in this system, and the other was to decide on the nature of the disinfection process.

Settling was taken to be represented by discrete settling, as this is a reasonable basis for typical primary removal estimates. A number of process choices exist for disinfection. In accordance with the intended scope of this effort, the disinfecting power of the disinfectant was conceptualized based on typical behavior of chlorine in the targeted system.

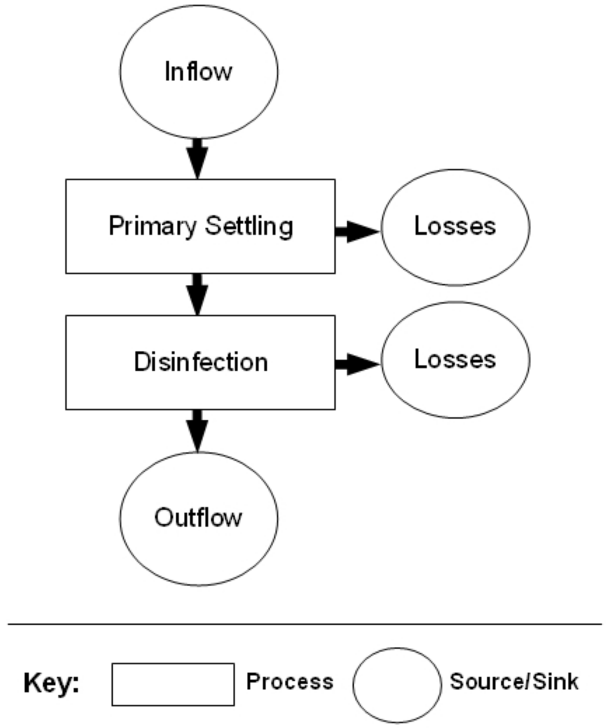

Figure 9 Basic unit processes

The net result of these choices was that two basic process representations were reasonable:

- The primary settling unit was established with sedimentation, a function of volume and overflow rate.

- The disinfection chamber was characterized as well-mixed, with disinfection a function of residence time and disinfectant concentration.

The net effect of this set of conditions is that the process model is typical of currently used tools for WWTP analysis as far as primary removal and disinfection are concerned, but adds a minimum next step in the form of the partitioning of bacteria between the solid and fluid phase. The partitioning relationship does not represent an inherent limitation in the modeling approach as far as individual microorganisms are concerned, provided that a dose/response relationship and a liquid/solid equilibrium relationship can be defined for a particular species. As a result, in the event that the model is applied in practice in subsequent investigations, it can in principle be used to represent more specific parameters of interest. The requirement in the present work was to have a capability to develop representative results rather than specific ones, but it is intended that this model can be expanded to site-specific practical applications. As data accumulates, the degree to which this is achieved can be judged.

This pragmatic approach should not be considered to constitute an assertion that there is no potential for other factors to affect results when a particular system is to be simulated. This approach is limited only to the disinfectant and sedimentation processes. Other processes, including potential predator-prey or other survival factors have the potential to affect population. To the extent that these can be represented by a gross removal process where the dominant removal is through disinfection or sedimentation, such systems could be accommodated by this model. If other factors depart materially from this set of assumptions, then results may not be approximated by this model, and a more complex model would be necessary. Further physical experimentation may be devised to develop an understanding of this potential. For the present, however, this model was applied to explore the consequences of this set of assumptions.

Model Development

The model which was devised to represent these conditions is illustrated in Figure 10[5]. The two major process compartments are the same as those identified in Figure 9, namely the primary settling unit and the disinfection unit. The concepts embodied in this model are the same as those discussed above[6]. In addition, the overall model has the following specific characteristics:

- There are two influent streams, representing wastewater and wet-weather components. Each component is defined by settleable solids content, flow rate, and bacteria content. In the instances simulated in this project, concentrations are taken as static, and flows are taken as variable, but the model could be configured to represent variations in all terms if measurements supported this.

- The two influent streams are co-mingled in the settling reactor in a direct mass balance relationship.

- Removal processes in the primary settling reactor consist of:

    - Primary settling removal of settleable solids.

    - Removal of bacteria associated with settleable solids, where bacteria are in an equilibrium reaction between the settleable solids and the fluid.

- Removal processes in the disinfectant reactor consist of:

    - Disinfectant demand as a function of solids.

    - Bacterial removal as a function of disinfectant concentration.

The technology used to simulate this set of conditions was a systems-oriented environment platform known as Simile[7]. Appendix A provides a description of the mathematical model was developed to represent this arrangement of unit processes. As noted, the model contains several compartments that represent the various unit processes, and tracks several dynamic components, including flow, sediment, bacteria and disinfectant dosage. This model is able to fully represent the dynamic behavior of a plant based on these components provided that the kinetic and rate constant data is available for each component. The modeling process is described in the following section, with estimated rate constants.

---

[5] As discussed in Appendix A, more than one model was developed. However, the model finally adopted, termed Multiphase Sediment/Bacteria Model-2 (MSBM-2), is described in this chapter as this is the model that was dealt with in detail.

[6] MSBM-2 model conceptual assumptions and limitations are described in ***Microorganism Behavior*** and ***Treatment System Components*** sections of this Chapter**.**

[7] Simile version 5.4p2 Standard, Simulistics, 2008.

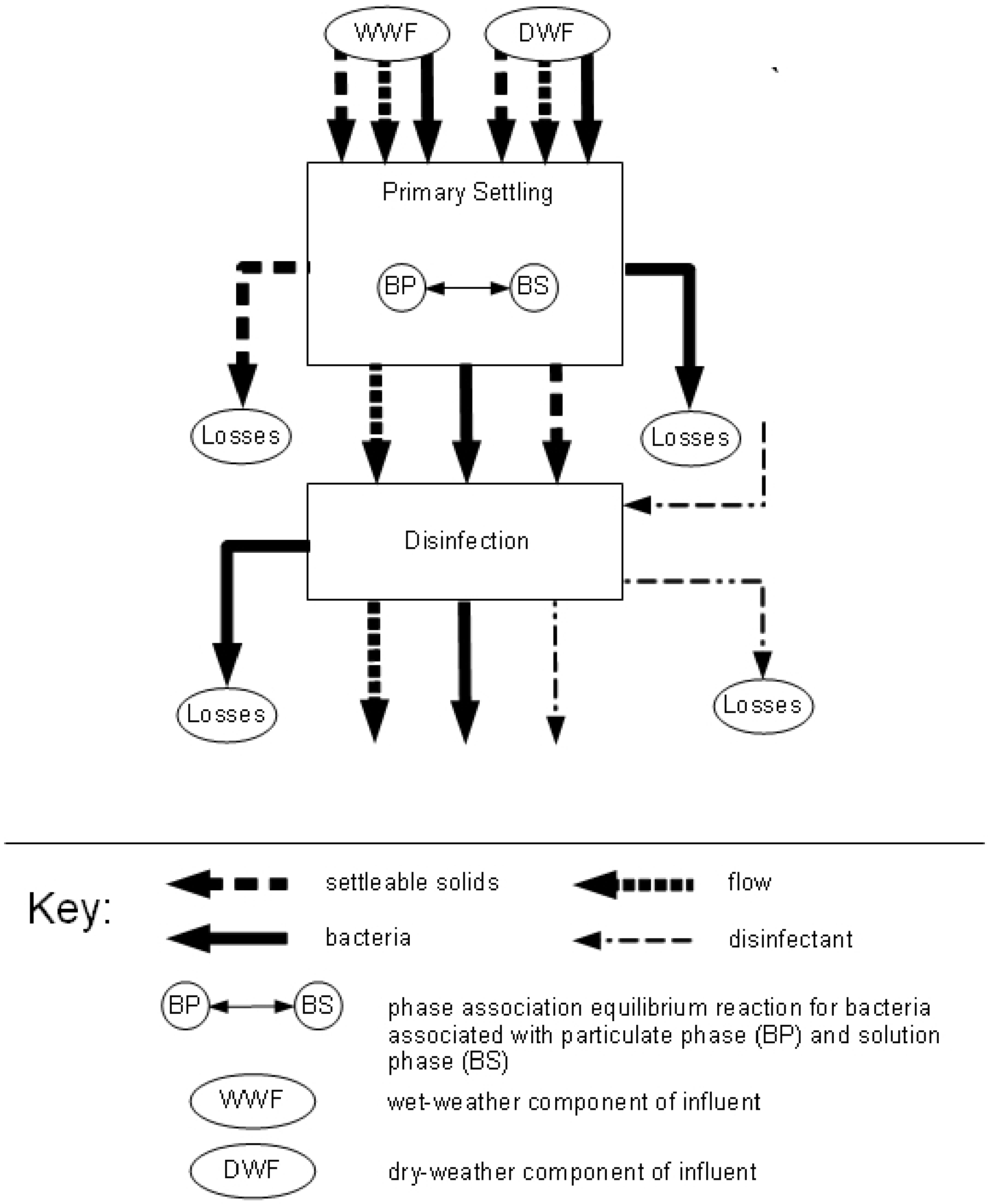

Figure 10 Multiphase Sediment/Bacteria Model- version 2 schematic

Model Implementation

Input Data

The model defined in this chapter requires a range of input parameters. As noted earlier in this chapter, some of these are reasonably well-defined in the literature while some are not. This section provides a listing of key inputs to the model as applied in this project. It is acknowledged that in some cases these values represent a starting point only, and further research may be needed to provide a more confident basis for prediction. It is also noted that, as with many environmental parameters, some numerical values will need to be established case by case and site by site even though typical values may be available at some point in the future. Therefore, the values identified below should be understood to be indicative rather than definitive.

Not all of the data potentially used by the model are defined in this work. Stormwater flows, for example, vary case by case and don't have typical values. Therefore, they were built into the tool but there has been no attempt to develop "typical" stormwater flow rates. These can be added by interested future users of the model if and when this becomes appropriate.

Wastewater Flow Rate

The diurnal variation in wastewater flow will vary according to a characteristic pattern as a population goes through its daily routine. Similarly, a weekly cycle can be detected. The nature of this variation will depend on factors including prevailing socio-economic status of the service population, seasonal effects, and sporadic conditions or other factors that affect water demand and wastewater generation. Combined sewers which are intentionally designed to accept storm flows will be much more affected than separate sanitary sewers which are subject to infiltration and inflow.

A typical weekly variation input is shown in Figure 11. This is centered on a value of 1.0 as it is multiplied by the diurnal value and the intent is to superimpose variation, not to represent an actual rate.

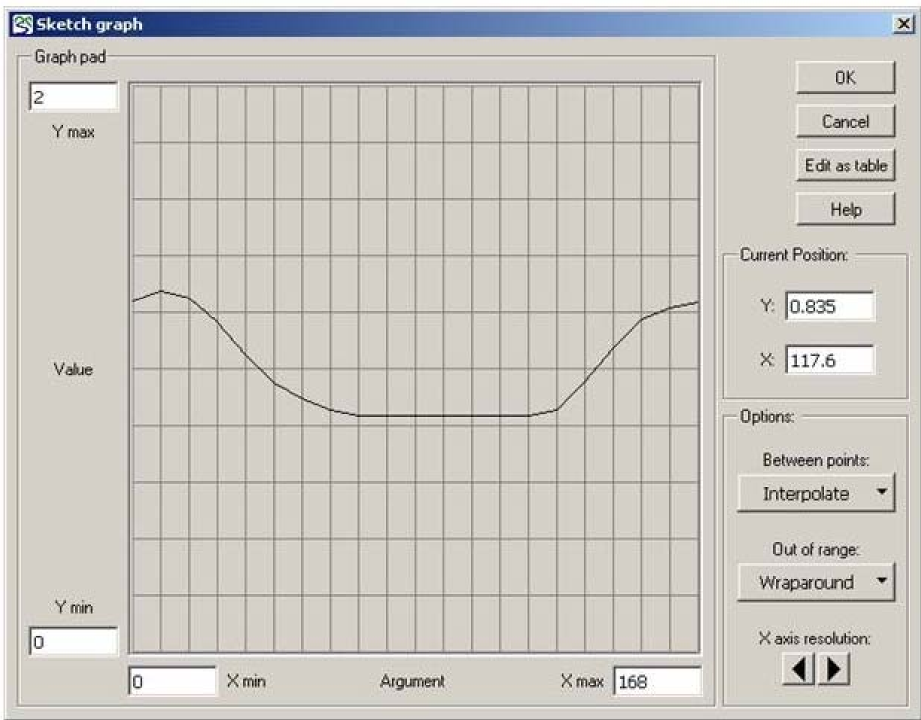

Figure 11 Characteristic of weekly flow variation input for model

Similarly, a typical daily variation input is shown in Figure 12. This is centered on a rate of 22 ft^3/d, or about 165 gal/d, which is consistent with expectations on a national basis. It is noted, though, that this rate is only a reasonable average, and that values will range quite substantially from this amount from site to site. For present purposes, however, this is a reasonable representation of typical and reasonable conditions.

As noted, both variation curves were set to automatically repeat (denoted in the model as "Wraparound"), which means that they will run in the model continuously over the full period of the model simulation. The details of these curves will not be identical to every possible actual case, but they do represent a reasonable basis to explore the sediment and bacterial demand relationships over the long-term in a typical domestic U.S. system. To illustrate this synthesis, a result of the net system behavior of the two relationships for a unit population was produced for a duration of several weeks. The individual and combined periodicity cycles are presented in Figure 13.

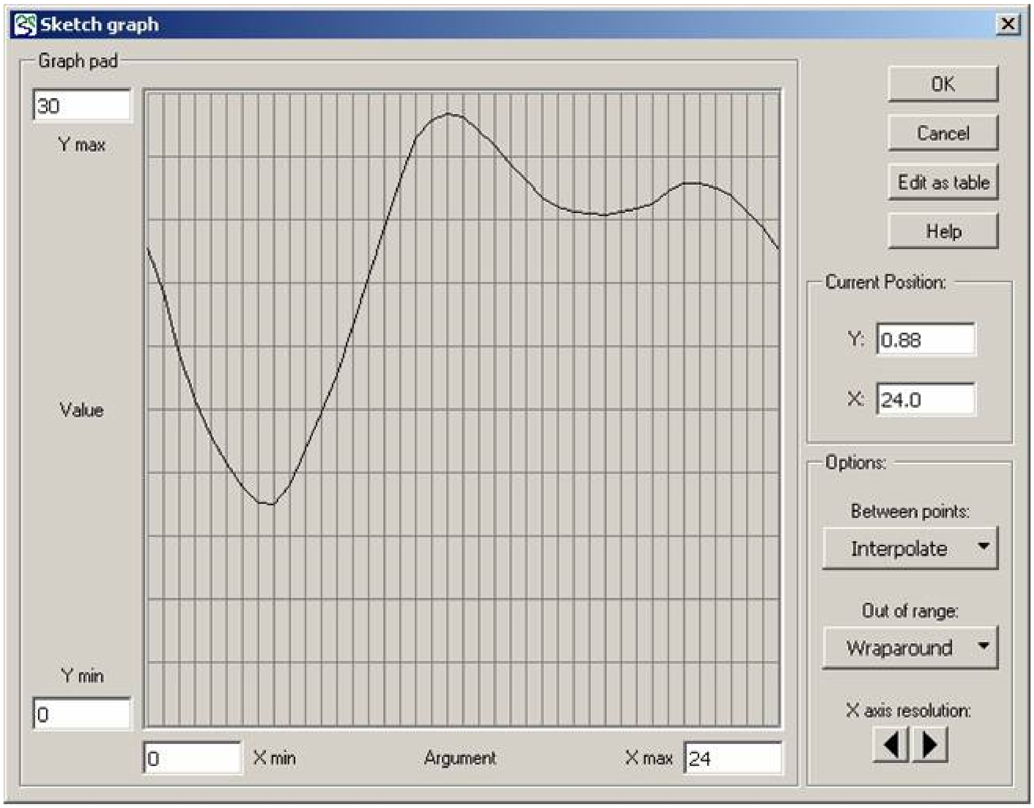

Figure 12 Characteristic of daily variation input for model

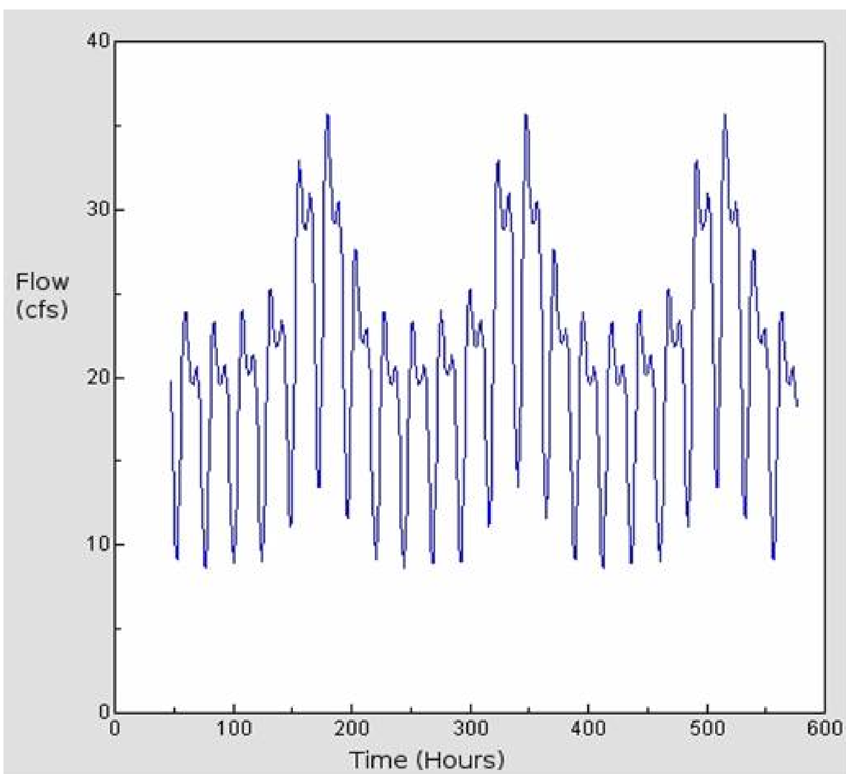

Figure 13 Characteristic of combined daily and weekly variation inputs for model

As with the individual figures, this result is only a general pattern, not one with a tight relationship to any particular location. However, it does provide a basis for an evaluation of system behavior in a dynamic state. It is noted that if necessary, other periodicities (e.g., seasonal) could be built into the model with little difficulty.

Wastewater Quality Parameters

The characteristic concentrations of suspended sediments and the representative bacterium were taken as 240 mg/l and 10^9 no/dl, respectivly[8]. Settling velocity of suspended sediments was taken as 2 cm/sec for 0.2 mm particles, but varied substantially, which is generally consistent with conditions that might be expected in a primary settling system. An adsorption rate constant of 6 x 10^{-8} m/s was used. This is consistent with some known experimental results on an inert substrate (glass) for some organisms (*Pseudomonas flourescens*) (Camper et al., 1993), but it is acknowledged that this may not be closely representative of conditions that could apply in any particular case, i.e., combination of bacterial species, fluid characteristics, and substrate in a given wastewater flow.

Model Testing

Since site-specific measurements would be required to enable estimation of removal kinetics in any particular instance, further research developing parameters for a particular case would be needed if a specific conclusion regarding removal was to be made. However, the model is still useful for exploring the general behavior of this kind of system. Several model runs were performed to evaluate some of the key variables over a range, and to determine the behavior of the system as these variables changed. Chlorine was chosen as the standard disinfection technology, though this does not preclude the use of other disinfectants for the model. If additional data were available, the model might provide increased empirical understanding of the way such a system could behave in a particular plant.

Description of Model Test Runs

The Multiphase Sediment/Bacteria Model - version 2 (MSBM-2) was constructed to enable input of parameters in any reasonable combination, and therefore allows testing of the effect of variation of parameters either independently or concurrently. For example, it is possible to vary both the rate of SS removal, k_s, and the chlorine dose, Cl_d, influence on the overall system efficiency as measured by the percentage of bacteria removed. However, in reality the effectiveness of chlorine disinfection is not independent of k_s. In the model described, both chlorine disinfection and solids removal by settling are represented as straightforward first-order processes. What is less apparent is the fact that the interaction between k_s and Cl_d directly influences treatment efficiency. Application of the MSBM-2 model, for a range of k_s and Cl_d values, provides the information necessary to develop an empirical relationship between k_s, Cl_d and treatment efficiency.

Model Results

Table 1 below provides a summary description of each MSBM-2 model run completed for this illustration. The intent of this was straightforward, namely, to generate a representative series of model results for different combinations of SS settling rates, k_s, and chlorine dosage, Cl_d. The resultant removal efficiency listed for each model run corresponds to the predicted overall bacteria reduction achieved for a 100-d simulation period. All other model parameters, either time-variable or constant, were left unchanged. As expected, treatment efficiency increases as either k_s or Cl_d rises; however, it is apparent from the results that the relative effectiveness of chlorine disinfection is influenced by the assigned value of k_s.

In order to quantify the extent to which the effectiveness of chlorine disinfection is influenced by the solids settling rate, a stepwise multiple-linear regression analysis was performed on the model results presented above in Table 1. A number of proposed combinations of predictor variables were introduced in the stepwise regression analysis including: k_s, Cl_d, the square of each of k_s and Cl_d, the product of k_s and Cl_d, and the logarithm of the product of k_s and Cl_d. This regression analysis could be refined considerably and the number of model runs used for analysis purposes could be expanded. Monte Carlo type simulation may be applied to generate a wide range of model results for random combinations of k_s and Cl_d. However, for the purposes of this illustration, the analysis described is sufficient.

[8] Terminology of no/dl is equivalent to cfu/100 ml.

Table 1 Summary of Multiphase Sediment/Bacteria Model - Version 2 Runs

SIMILE Model Run	**Solids Settling Rate k_s (d^{-1})**	**Chlorine Dose Cl_d (mg/l)**	**Bacteria Removal Efficiency (%)**
1	0.10	0.10	57.2%
2	0.50	0.10	57.6%
3	0.10	0.50	57.2%
4	0.50	0.50	59.2%
5	0.25	0.50	58.2%
6	0.01	1.00	57.4%
7	0.05	1.00	57.7%
8	0.10	1.00	58.1%
9	0.25	1.00	59.3%
10	0.50	1.00	61.1%
11	0.75	1.00	62.7%
12	2.00	1.00	68.6%
13	10.00	1.00	78.5%
14	0.50	2.00	64.5%
15	0.25	3.00	63.1%
16	0.10	3.00	59.9%
17	0.50	3.00	67.4%
18	1.00	3.00	73.4%
19	2.00	3.00	80.0%
20	2.00	5.00	66.3%
21	0.10	10.00	65.3%
22	0.25	10.00	72.5%
23	0.50	10.00	79.7%
24	1.00	10.00	86.4%
25	2.00	10.00	91.2%

The multiple-correlation coefficient was 93%. A plot of the resultant treatment efficiency surface, as defined by the regression relationship, is presented in Figure 14. This graphic is a useful visual illustration of the interaction between k_s, Cl_d and the ultimate treatment efficiency achieved.

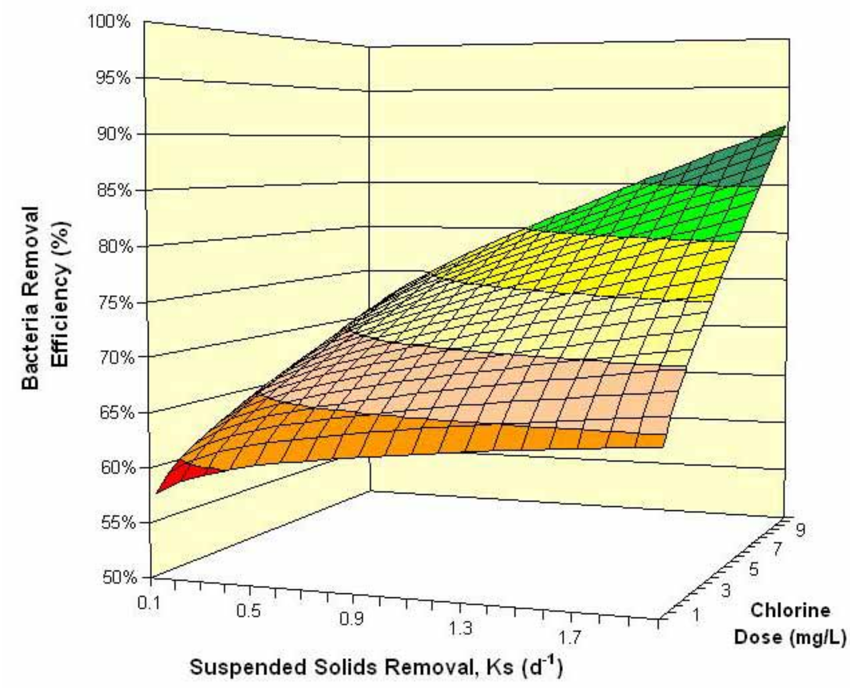

Figure 14 Removal efficiency surface plot for chlorine dose and solids removal

What is apparent from this analysis is that for any given chlorine dose in the range simulated, increasing SS removal results in greater disinfection. Similarly, for a given degree of SS removal, increasing chlorine dose results in increased disinfection. This is an expected result. Whether the forms of the surface that was developed is typical or atypical cannot be determined based on available data, but the overall results are consistent with behavior that might be expected based on what is known about the concurrent effects of sedimentation and disinfectant dose in a system of the type simulated.

Topic Area 3 Conclusions

A potentially useful multi-phase bacterial removal model, MSBM-2, which incorporated the interaction of primary treatment and disinfection, was tested for functionality using arbitrary input parameters, and performed satisfactorily for a simplified modeling approach. The model has potential applications in real-world contexts, but is presently of limited value for practical or theoretical applications because of the lack of dependable and comprehensive measurements of fundamental relationships and rate constants either in general or at a particular site. Nevertheless, this tool may have significant potential in describing further relationships between disinfection potential and solids content removal if supporting data for these correlations are eventually forthcoming.

The key missing link in data relates to bacterial behavior. The review of the literature and interviews with experts in the field (Van Weele, 2009) makes it clear that the reversibility of the bacterial attachment reaction is unproven, and that confident estimates of rate constants cannot be made without physical experimentation. Therefore, further research into this aspect of the problem is required for further progress to be made in this area.

The test case model application demonstrated that:

1. The SIMILE model platform is an appropriate numerical tool analysis of variable interactions.

2. Although limited in scope, the test case presented above underscored the importance of SS removal rates on the effectiveness of chlorine disinfection. From an operational perspective, achieving a desired level of bacteria removal or treatment efficiency requires an understanding of the extent to which bacteria are

removed through solids settling.

In terms of future research needs, the analysis described above would benefit from:

1. Experimental investigations focused on quantification of the relationships between bacteria removal and SS settling. This could suggest ways that the simple first-order relationship for SS removal applied in this MSBM-2 could be modified to account for the range of SS characteristics found in the influent waste stream.

2. Once data is available, development of more rigorous numerical experiments (MSBM-2 model runs) and formal statistical analysis to establish definitive design rules for disinfection of stormwater.

3. The variability of solids settling rates as a function of particle size is a potentially important extension of this work, and could be handled in essentially the same way that the MSBM-2 model was approached.

4. The incorporation of alternative settling behavior (other than discrete settling) would be a useful extension of this work.

5. The concurrent evaluation of multiple microorganisms (indicators and pathogens) would be useful to provide insights into the best way to apply these results in practice. If pathogenicity cannot be strongly related to indicator inactivation, it may be that the fundamental methods of determining disinfection and sedimentation are all that can be reasonably defended.

6. Given the economic significance of this area (large numbers of treatment facilities rely on sedimentation and disinfection to at least some degree nationwide) this area of research should be pursued further.

7. This model could in principle be applied in stormwater contexts where settling and disinfection are applied.

Despite the preliminary nature of the data, it seems apparent that solids settling rates and chlorine dose for disinfection purposes should be addressed in combination in order to reliably estimate disinfection efficiencies.

Although the results of this element of this project are interesting and promising, an underlying point is that there is presently little basis to defend application of advanced bacterial removal models in the absence of supporting rate and relationship data.

Chapter 8 Topic Area 4: Best Management Practice Pathogen Removal and Routing Analysis

Pathogen Indicator Routing in the Watershed

Indicator bacteria in the environment are highly variable and are produced by many sources. Sources potentially include discharges from WWTPs, WWFs, and ambient settings within a watershed. Such variability makes it difficult to interpret receiving water conditions and to associate cause and effect.

The movement and state of microorganisms through the wastewater stream was discussed in detail in Chapter 7. Significant considerations should be given when predicting indicator bacteria transport related to whether they are associated with solids or free floating and independent. The behavior of bacteria is not restricted to microscopic effects. If associated with solids, the presence of a biofilm is an additional factor of interest. Biofilms can grow on a variety of surfaces, including submerged rocks in waterways. Fleming et al. (2007) assert that bacteria attachment to the substrate, together with the associated slime developed by those bacteria, can have a significant impact on sediment topography and frictional resistance at the sediment/water interface, as well as on conductivity at greater depths.

Recent investigations relate to the association of indicator bacteria with solids. Hipsey et al. (2006) found, in an investigation of TC and *E. coli* in a drinking water reservoir, that most (>80%) of the microorganisms were associated with particles, and of these, most were associated with particles which ranged from 3.2 - 4.5 µm. They suggested this preference was a function of particle abundance and surface area relationships. They also suggested that bacterial attachment and the resulting increase in settling rate should be incorporated in models involving bacterial sedimentation.

Ambient conditions can certainly have an impact on the fate of indicator bacteria. Whitman et al. (2006) determined the relative impact of known wet-weather sources of *E. coli* on beach areas, and found that a variety of sources including land-deposited *E. coli* can affect water quality for prolonged periods. Land deposition of *E. coli* on forest soils were cited as having persistent contributions over the course of a year. The endemic nature of indicator sources and the lengthy time scales of deposition and release that are suggested by this work highlight the difficulty of evaluating indicator bacteria unless the methods are able to represent the many pathways by which these organisms can be generated and transported. Whitman et al. (2004) also investigated *E. coli* in the context of watersheds along Lake Michigan, and recommended that the entire beachshed[9], indicative of the dynamic interactions of the whole

[9] 'Beachshed' is a term denoting the land areas tributary to a beach, which is a function not just of the immediate lands above the site but of those areas that impact it through circulation of the nearby water bodies.

system, needs to be considered, including bacteria source, flux and context. It is noted that there is an influence of solar radiation on bacterial removal, and that the relationship between insolation and *E. coli* density is complicated by relative lake level, wave height, and turbidity. The numerous sources and the continuous importation and nighttime replenishment of *E. coli* were noted as well.

Struck et al. (2006) also noted the effect of solar radiation and temperature on indicator inactivation in field mesocosm studies and simulated sunlight with UV lamps on collected stormwater in bench-scale studies. Factors such as sunlight and temperature provide much of the inactivation in indicator bacteria, but other factors, including predation, sedimentation, filtration, sorption, pH, and BOD, also appear to influence indicator bacteria concentrations. Predicting predation of microorganisms may also be complicated in stormwater ponds and receiving waters due to increased nutrient discharges during WWF; Hulot et al. (2000) proposed that a non-linear food chain was a better model for macroinvertebrates in fresh water ecosystems due to nutrient enrichment.

Roslev et al. (2008) discussed the Danish shift to enterococci from *E. coli* as an indicator. This evolution was prompted by a new European Directive for bathing water, and they note that the relative lack of historical information on enterococci compared to *E. coli* in some areas may pose challenges in interpretation of conditions at a site as this regulation is implemented. They also identified in a case study that relatively similar abundances of enterococci are present in sediments and in the water column, and noted relatively greater abundance of enterococci than *E. coli*.

Pathogenic indicators from non-human sources have historically been observed and remain a problem. Ferguson et al. (2003) did a review of the literature from 1953 through 2002, and developed a synthesis of knowledge assets and gaps in the area of microorganism fate and transport associated with animal fecal deposits on the land surface. Their work provides a useful outline of many of the factors which affect transport mechanisms and also provides an understanding of the degree of uncertainty that remains a fact in this technical area. A report from the Ministry of Agriculture and Food of New Zealand (2006) provides a useful summary of animal behavior and land management practices that are most pertinent to the problem of bacteria introduced to waterways. They note the importance of vegetative patterns and consequent preferential behaviors by animals, and provide insights into management techniques that respect these factors and may mitigate the contribution of direct and indirect bacterial components to the waterway. Their context was agricultural animal sources, but the general notions of bacterial transport are certainly pertinent to this topic, and agricultural impacts on water bodies that are urban resources are certainly germane. Pachepsky et al. (2006) discuss the fate and transport of bacteria associated with manure, and among other things describe the partitioning and attachment of pathogenic and indicator organisms to solid particles in runoff, soil, and sediment, and the transport with straining or entrapment in overland flow and in streams. They note that there is a paucity of data regarding the transport of pathogenic microorganisms, and develop a useful synthesis of existing approaches to modeling some of the key relationships that apply to this phenomenon.

The heterogeneity of the underlying bacterial population further adds to this picture. Indicator bacteria are defined by a reaction to a stated test, and there can be considerable variation between species that exhibit this reaction. A recent study (Yang et al., 2004) measured about 280 *E. coli* isolates from an animal feedlot. That same study determined that motility varied very broadly within the population, and that this variability was correlated with biofilm-forming ability as a function of the selected medium. Another researcher (Molina, 2005) explored the association of enterococcus with manure, and found significant temporal and spatial variations in behavior. This work clearly demonstrated that enterococci were non-ideal indicators of cattle in that context, as apparently likely correlations were absent or limited. Other research has shown that charge differences and other cell properties can render the transport properties of indicators and pathogens quite differently (Bolster et al., 2006). Given the various factors affecting fate and mobility, it is perhaps hardly surprising that the utility of indicators is conditional; however, for pragmatic and historical reasons, indicators remain an important element in the arsenal of water quality managers.

Kim et al. (2008) experimentally determined sediment fractions and associated microorganisms in CSO, and attempted to model them. They were unable to achieve adequate results when treating the sediments as having a single settling velocity, but achieved effective results in simulating diurnal variation in SS and in indicator organisms when they partitioned the data into two sediment size fractions (fine and coarse) suggesting that sediment/indicator processes may indeed be predictable. Their work additionally raises a question as to the value of efforts to find meaningful correlations of inflow and outflow when based solely on total sediments.

Keeher (2007), reported on the results of an expert scientific workshop focused on development of criteria and noted that improved quantitative methods to evaluate indicator bacteria, including sanitary sources, could be useful. However, the industry would seem to have a long way to go in this regard. Clary et al. (2008) provide a review of the international BMP database and its content as related to indicator bacteria. In this effort, they acknowledge the high variability of the phenomenon, both in terms of statistical properties in a flow stream and in terms of the sources of the population. They attempt to draw conclusions about the impact of BMPs on indicator bacteria based on the limited available data. Among other things, they acknowledge the need for a determination of the statistical significance of differences through hypothesis testing and also note the benefits of added site-specific data. Their chosen scope, however, displays little that can shed light on the underlying removal processes within the BMP. Discussions related to this point (Jones and Clary, 2009) suggested that future plans will extend consideration of the statistics and methods of monitoring and interpreting BMP performance. Given the unknowns in particle association (e.g., Kim et al., 2008), and the high variability of the phenomena at work in this problem, it seems that continued attempts to interpret BMP performance without first understanding underlying processes or developing a statistically valid ability to discriminate cause and effect could be limited. Reference is made, for example, to recent work by Bratieres et al. (2008) who with detailed monitoring were able to demonstrate that biofilters, i.e., filter media with an overlying vegetative cover that is intended to enhance performance, can provide significant removal (e.g., 80% or better) of indicators and pathogens. That work also pointed out the significance of antecedent wet or dry conditions on performance, underlining the need to evaluate results of bacterial removal in a context not only of facility type and related factors, but also of the history of the system. Further, they note that removal of different organisms can be affected not only by the nature of the device, i.e., citing the persistence of indicators in sediments and their availability for re-suspension, but also by the behavior of the organisms, i.e., discussing the greater removal of smaller organisms in a filter if they are associated with and transported by larger particles.

Recent research indicates that the adhesion of bacteria to sediments can be influenced by concurrent exposure to other waterborne contaminants with counter intuitive results. Guber et al. (2005) found that manure in the water stream changed both the degree of attachment to a soil, with a linear isotherm characterizing *E. coli* attachment without colloids from manure, and a non-linear result otherwise, and with the degree of attachment to soil particles much greater without manure. An interesting added complication attests to the influence of water quality on attachment; Yolcubal et al. (2002) found bacteria were much more strongly associated with the fluid phase then sediments when the solution was favorable for growth, while association with the solid phase was greater when the solution was unfavorable for growth. Additionally, degree of saturation has been shown to have an impact on indicator bacteria transport through a soil medium (Powellson et al., 2001), which may prove to be relevant to adhesion to particles in receiving waters and even in case of sludge management at WWTP.

Nevertheless, there are indications that emerging methods of analysis may provide insights into the underlying cause-effect relationships that govern bacteria mobilization and control in a watershed. Alfaqih et al. (2008) used a decision analysis framework to identify *E. coli* sources in an Alabama watershed, and determined that problematic sources of the bacteria were associated with agricultural sources in the watershed. The methods by which this association was achieved appear to rest on statistical inference based on candidate cause/effect couplings. This research suggests that even if details of system processes are unavailable, large-scale trends may be tractable where indicator source control is at issue.

In a USGS report, Hyer and Moyer (2003) developed multiple linear regression models to predict FC bacteria concentrations in streams as a function of more easily measured water-quality parameters, i.e., turbidity, pH, water temperature, and DO concentration, with correlations between 64 to 88% of the observed variability in FC concentrations. Predicting FC concentrations from these parameters allows for quick and comparatively inexpensive estimates; however, empirical models values are best suited for situations requiring approximate FC concentration, only. The authors note that while turbidity appears to be an effective predictor of FC concentrations, the link between sediments, measured as turbidity, and FC is not direct; rather, the urban landscape creates conditions that favor both elevated FC concentrations and turbidity levels.

Even though, one should be cautious in the causal relationship between SS and indicator organisms, association with sediments in the natural environment is known to affect microorganisms of interest. The literature suggests that the majority of indicator bacteria are associated with sediments, by means of physical incorporation within those materials or by attachment to them, and recent research has indicated that it is possible to model the association of *E. coli* with sediments as an irreversible process in which bacteria move from the fluid phase to the solid phase (Jamieson et al., 2005a). Analysis in this case was approached by simulating sediments as an independent process, and associating removal of indicator organisms with sediment losses. The association of bacteria with sediments was based on experimentally determined partition coefficients; the irreversible nature of the reactions may be particular to the site and circumstances of that research. Other work (Jamieson et al., 2005b) indicates that resuspension during flow changes can re-introduce indicator bacteria that have been deposited in river sediments.

Modeling Approach

This project topic was structured in a way that provides some immediate results but that lends itself to further expansion and adaptation as and when further data become available. Theoretical performance of some BMPs was estimated through modeling, supported by reasonable model parameters. The intent was to develop principles of operation that can eventually be validated through more detailed analysis and potentially through field verification.

This chapter first discusses the development of the model used in this assessment, and then discusses the case studies that were tested using that model.

Model Development

The tool used to accomplish this task was a suitable watershed generation and pond routing routine (developed by the senior, leading author of this report and used in numerous research projects) in the form of a continuous model known as QUALHYMO. The model has the capability to develop removal estimates under a time series flow/concentration basis for:

- Multiple sediment fractions.
- Arbitrary first order removal kinetics (such as for a suspended, indicator bacteria).
- Advective/dispersive transport and removal.
- Arbitrary stage/volume relationship.
- Arbitrary multiple stage/discharge outlets.
- Arbitrary bypass fraction that may be lost or re-associated with the effluent stream.

The description of the model is provided in the user manual, and sections of the manual are edited and adapted for this report with the permission of the author (Rowney, 2009). The model is able to represent the theoretical behavior of a wide range of detention-based BMPs in its native form. The physical concept of the control pond used in the model is based on a volume that may have a variety of input and output sources. This reflects the intended use of the model in a planning environment comparable to that which is being considered in this project, i.e., stormwater quality control and the resulting need to allow analysis of a wide variety of possible configurations.

As shown in Figure 15, the pond is assumed to have a single inlet and a single outlet. There is provision in the model to allow simulation of a high level bypass. Alternatively, all flows can be forced through the pond. Between events, a residual volume can be permitted, or the pond can be dry. In addition, a base through-flow rate can be maintained in dry weather. It is possible to represent any gravity type pond outlet structure where backwaters are not of concern, since the outlet structure is input as a flow/stage curve. Similarly, an overflow curve can optionally be specified for the model, to represent weir flow or any other overflow condition in the event that the pond maximum storage capacity is exceeded. To reflect possible operation as a batch facility, there is a provision to incorporate a detention time in the model. Flows not exceeding the maximum storage capacity will be retained until there is a dry period longer than the stated detention time, and then released.

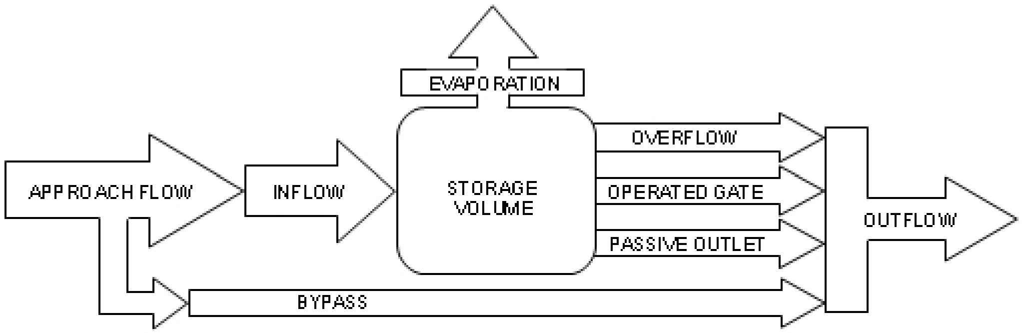

Figure 15 Typical structural control pond simulation components

The model presently contains two mechanisms for reducing the amount of pollutant in the detained water volume. These are removal by first order decay and removal by discrete settling. In both cases, removal is computed using the assumption of complete mixing, but the user is given the capability of dividing the control pond into any number of contiguous compartments. This capability allows the user to approximate the effects of either an imperfect reactor, or a plug flow reactor by using a large number of sub-compartments.

The model can route flows through the pond, or this can be provided from any suitable (continuous simulation) gauge or other simulation tool. Flows used in this way may or may not contain quality as well as quantity (flow rate) information.

Pond Flow Components

The routing of water volumes through the control pond is achieved using a routing scheme that is driven by several control curves. These are discussed below.

Inlet Control Curve

This option was not a major consideration in this project, so it was not used in this analysis. It is described for the sake of completeness. The curve allows flows approaching the pond to be split, with part of the flow bypassing the pond and part entering the pond. The purpose of the curve is to allow the user to simulate conditions where the pond is designed to have excess flows bypass the pond, or where flows less than some minimum are not controlled, or where other considerations lead to control over the approach flow. In short, on-line/off-line configurations are possible with the curve.

Volume/Stage Curve

This curve provides the storage information used in simulations. If the input is a volume/stage curve, linear interpolation or extrapolation is used to calculate pond volume at any stage. If the user inputs only an area/stage curve, a volume stage curve is calculated as:

$$V(S_1) = 0$$

$$V(S_{i+1}) = (S_{i+1} - S_i) \times \frac{A_{i+1} + A_i}{2}$$

where

S_i = stage (m) at time *i*

A_i = horizontal area (m^2) at time *i*

$V(S_i)$ = Volume (m^3) at stage S_i

Volumes are developed over the full range of depths in the pond using these relationships.

Operated Outlet Curve

This curve represents the primary outflow structure from the pond. It can be used to maintain a continuous flow through the pond or to represent a gated structure which is operated according to a design batch detention time. Selection of one option or the other is accomplished by specifying the value of detention time. The design detention time can be specified in the simulation by the user as zero or some positive value. A positive value allows simulation of batch operated ponds (as opposed to the continuous operation otherwise assumed). Batch operation is defined here as a procedure whereby volumes of water entering the pond are held for a specified time before release.

To do this, the model maintains a 'timer' which records the age (time since end of last event) of water in the pond. When the 'timer' shows all the water in the pond has been there for at least a period of *TDET* (the value of the design detention time), a release of water through the operated control curve is activated. The following operation rules are applied to govern this process:

1. Provided that the 'timer' registers greater than *TDET*, when flows enter the pond at a rate less than a specified base flow (*QBAS*), the operated outlet curve is 'opened'. Flows can therefore leave the pond at a rate determined by the operated outlet curve. Flows can also leave by the non-operated curve, as well as by overflows, if these curves are specified.

2. When flows enter the pond at a rate in excess of *QBAS*, the operated outlet curve is 'closed' and the timer is set to zero. Outflows from the pond are therefore not possible through the operated outflow curve. They can only leave the pond via the non-operated curve of overflow curve if present.

3. When flows entering the pond drop to less than *QBAS* after a period where *QBAS* was exceeded, the 'timer' is started. The operated outlet curve is left 'closed' until the ‘timer’ registers a value of greater than *TDET* as noted above.

Regardless of the status of the operated outlet curve, it is possible for flows to leave the pond through either overflow curve or the continuous outflow curve if stored volume is sufficient for flows to reach the release point on the curve, presuming these values have been specified by the user.

Continuous Outlet Curve

This is an optional secondary outflow structure, which may be omitted if desired. It permits simulation of more complex outflow conditions, or to have a small outflow released from the pond regardless of the state of the operated outflow curve. This was not used during this analysis.

Overflow/Stage Curve

This curve provides the capability for simulation of a pond overflow condition in the event that the available storage volume in the pond is exceeded. It need not be specified by the user because it can be added to the outflow curve instead, but has been provided as a capability to simplify model input under some conditions. It also enables an overflow to occur if the operated curve is closed.

Flow Routing Calculations
The model solves the continuity routing equation:

$$\frac{dV}{dt} = Qin - Qout_o - Qout_p - Q_V$$

where

V = pond volume (m^3)

Qin = total inflow rate (m^3/s)

$Qout_o$ = outflow rate through operated outlet structure (m^3/s)

$Qout_p$ = outflow rate through passive outlet structure (m^3/s)

Q_V = overflow rate (m^3/s)

The above three outflow components $Qout_1$, $Qout_2$ and Q_V can be combined in two distinct ways, depending on what the user has input. These are:

$$Q_{OUT}A = Qout_p + Q_V$$

and

$$Q_{OUT}B = Qout_p + Qout_o + Q_V$$

where

$Q_{OUT}A$ = outflow if operated gate is closed (m^3/s)

$Q_{OUT}B$ = outflow if operated gate is open (m^3/s)

These equations are true even if some of the components, i.e., $Qout_o$, $Qout_p$ and/or Q_V, are zero. The model takes advantage of this by calculating a combined outflow curve for each of these two cases, using whichever one applies at any time step depending on *TDET* and the time since the last outflow event.

The combined curves are calculated by the model and contain every stage value input by the user. The model first obtains and orders every stage supplied by the user which appears on any of the input curves (stage/area, stage/flow [operated], stage/flow [continuous], stage/volume, stage/overflow), and then drops duplicates. Flow and volume data are then found by linear interpolation of each of those curves for each stage. Finally, a total $Q_{OUT}A$ and $Q_{OUT}B$ at any stage is found by summation. The exact form of the user input data is therefore preserved in the combined outflow curves.

The solution of this equation is otherwise done in a fairly typical way. Taking a finite approximation:

$$\frac{2\times\left(V_2 - V_1\right)}{\Delta t} = (Qin_1 + Qin_2) - (Qout_1 + Qout_2)$$

where

Δt = time step (s)

V_j = volume (m^3) before (j=1) or after (j=2) time step

Qin_j = inflow rate (m^3/s) before (j=1) or after (j=2) time step

$Qout_j$ = outflow rate (m^3/s) before (j=1) or after (j=2) time step

The first term on the left hand side of the relation represents twice the rate of change in storage, and the remaining two terms on the right hand side represent twice the average inflow and average outflow respectively. This can be rearranged and expressed as:

$$\frac{V_2}{\Delta t} + \frac{Qout_2}{2} = \frac{V_1}{\Delta t} - \frac{Qout_1}{2} + \frac{Qin_1 + Qin_2}{2}$$

Since *Qout = f(V)*, then the left hand side of the above relation reduces to a unique function of the quantity of water in the pond:

$$SI = V + \frac{Qout\Delta t}{2}$$

where

SI = storage indication quantity (m^3)

Also, because outflow is a unique function of depth, i.e., it is assumed that there is no hysteresis, the SI term can be expressed as a function of outflow, *Qout*. Solution of pond outflow is then done at any time by solving for the right hand side to determine *SI* and then interpolating a curve of *SI* as a function of *Qout* that has been developed for the facility. This solution technique, referred to as the storage indication method, is popular in the simulation of reservoirs for hydrologic studies, and essentially implies that the effect of surface slope has no impact on outflows from the reservoir. For the BMPs which are being evaluated here, this is a reasonable assumption.

QUALHYMO also provides a summary of the global mass balance for flow quantity as a part of the pond calculation output statistics. This provides a check of possible continuity errors, although experience with the tool has shown that these errors are negligible.

Another factor was accommodated in this project which addressed losses from the bottom of a pond through infiltration. The above relationships can simulate this loss by a suitable choice of input conditions, but it was found that the input sets were time-consuming to develop and were also unsatisfactory because they did not reflect the head-dependant losses that are characteristic of a pond bottom infiltration. Therefore, an extended version of QUALHYMO was employed for the analysis. This version of analysis takes the above concepts a step further, but is otherwise consistent with the standard pond model. Figure 16 illustrates the nature of this simulation algorithm.

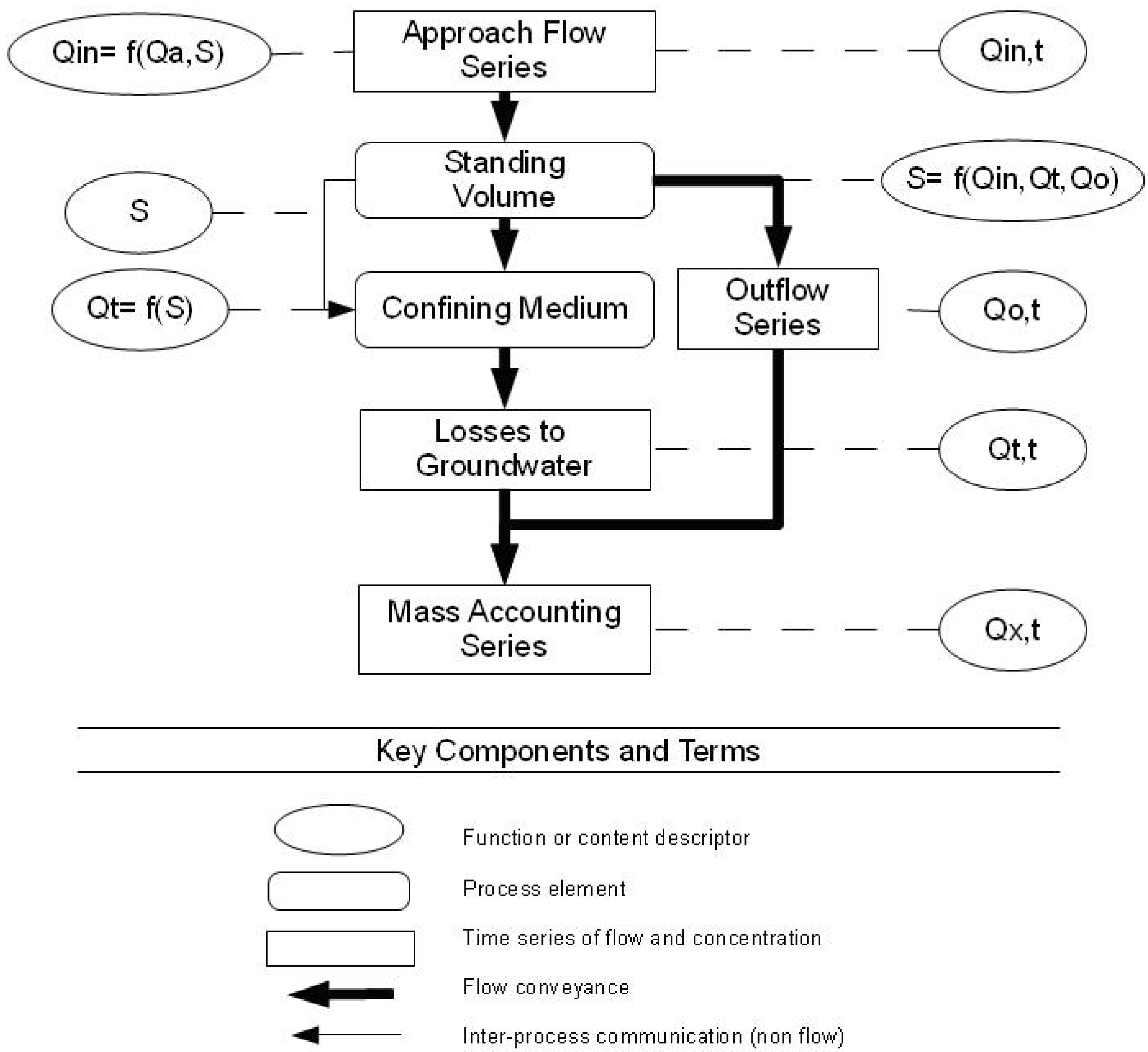

Figure 16 Key model components for pond with bottom losses

This BMP model has several features that made it effective for the present work:

- It has a head-dependant loss term, which eliminates flows and water quality constituents through the bottom of the facility.
- It is restricted to completely mixed behavior.
- It accounts for overflow, through-flow and loss terms explicitly in the model outputs, facilitating interpretation of model runs.
- An operated mode (one with a bulk detention time) was not simulated in this project.

This led to an ability to analyze structural BMPs that have passive overflow and losses through the bottom, which was a condition of general interest and consistent with the problem of interest in this project.[10] To enable a simulation with accounting for bottom losses and overflows, the model calculates inflow to the pond as:

$$SIb(S_2) = SIa(S_1) + \frac{(Qin_1 + Qin_2) \times \Delta t}{2}$$
$$SIa(S) = V(S) - \frac{(S - Oe) \times \mathrm{K} \times A(S) \times \Delta t}{2 \times Tf}$$
$$SIb(S) = V(S) + \frac{(S - Oe) \times \mathrm{K} \times A(S) \times \Delta t}{2 \times Tf}$$

where

S_j = ponding depth above datum before (j=1) and after (j=2) time step (m)

Oe = depth of outlet above datum (m)

K = hydraulic conductivity of lower confining unit (m/s)

$A(S)$ = filter area at depth S (m^2)

$V(S)$ = pond volume at depth S (m^3)

Δt = computational time step (s)

Tf = thickness of confining unit (m)

Qin_j = inflow rate before (j=1) and after (j=2) time step (m^3/s)

$SIa(S)$ = solution variable (m^3)

$SIb(S)$ = solution variable (m^3)

This equation is resolved for S_2 as a first estimate, and then S_2 is finalized as follows:

$$if\ (S_2 < Oe)\ S_2 = 0, Qt_2 = 0$$
$$if\ (S_2 > Sm)\ S_2 = Sm, Qt_2 = Qtp$$

where

Sm = maximum pond depth above datum (m)

Qt_2 = exfiltration flow rate Qt at time 2 (m^3/s)

Qtp = exfiltration flow rate at maximum pond depth (m^3/s)

To calculate outflows that do not exit through the pond bottom, a case variable is introduced. Results are dependant on the depth implied by inflow conditions, initial volume, and exfiltration rate.

[10] It is noted that this choice of conditions was not imposed by limitations of the available tool. The model can handle a wider range of BMP characteristics and these functions may be applied in subsequent research projects or practical applications if needed.

$$Qo = \begin{pmatrix} 0 & | & Oe < S_2 < Sm \\ 0 & | & S_2 \le Oe \\ Qi - Qtp & | & S_2 \ge Sm \end{pmatrix}$$

where

Qo = bypass flow rate (m^3/s)

With this determination made, it is possible to calculate volume in the pond at any time:

$$V_2 = V_1 + \frac{(Qin_1 + Qin_2)\times\Delta t}{2} - \frac{(Qt_1 + Qt_2)\times\Delta t}{2}$$

It is similarly possible to calculate flow through the bottom due to exfiltration rates (Qt in m^3/s) at the head that prevails over the time step, as follows:

$$Qt = \frac{(S - Oe)\times \mathrm{K} \times A(S)}{Tf}$$

Finally, to calculate total outflows (for mass balance accounting) a simple summation of terms is effective:

$$Qx = Qt + Qo$$

where

Qx = total outflow from the BMP (m^3/s)

Quality Analysis

Quality Routing Equations

The control pond model simulates pollutant removal by routing constituents through a series of well-mixed reactors of equal volume and depth. The user specifies the number of reactor elements, and the model determines individual element characteristics from the curves specified for the overall pond. Routing through each element is achieved by a numerical solution of a conservation equation for a completely mixed reactor, which is written here as:

$$\frac{dV \times C}{dt} = Qin \times Cin - (Qo + Qt)C - LOSSES$$

where

V = reactor volume (m^3)

C = concentration in reactor and outflow (mg/m^3)

Qin = inflow rate (m^3/s)

Cin = inflow concentration (mg/m^3)

Qo = outflow rate (m^3/s)

Qt = exfiltration rate (m^3/s)

$LOSSES$ = losses due to sedimentation or decay

with constituent units in mass (e.g., mg as shown above) or numbers (e.g., no/dl for bacteria).

In this model, a variable volume reactor is assumed, and flows and volumes at any time, *t*, are determined from the water quantity calculations. Losses may be calculated in one of two ways. For a first order decay constituent:

$$-LOSSES = -k \times C \times V$$

where

k = decay coefficient (s^{-1})

For the case of discrete settling, losses are also expressed in a first order form:

$$-LOSSES = -v_s \times C \times A$$

where

v_s = effective settling velocity (m/s)

A = horizontal projected area of pond element (m^2)

The principle difference between these two equations is that for the first order losses the coefficient k is independent of retained volume, whereas for settling the loss rate varies with depth. This is because v_s is constant for a given size fraction of sediment, while A is a function of water depth in the pond.

It is noted that the relationships used in this model result from an assumption of complete mixing as shown in Figure 17. Although the approach used is a typical method, there are also some alternatives that are commonly used. Future analyses to explore the implications of alternative loss models may be useful. The completely mixed approach used was a reasonable choice for several reasons:

- It represents hydraulic conditions consistent with the completely mixed assumptions used for the first order decay pollutant.
- The pond is likely to experience a degree of turbulence when subjected to inflows anticipated from stormwater runoff, which makes quiescent settling a questionable assumption.
- Finer fractions in particular will tend to react more to turbulence, and may be better approximated as completely mixed than in a quiescent settling situation.

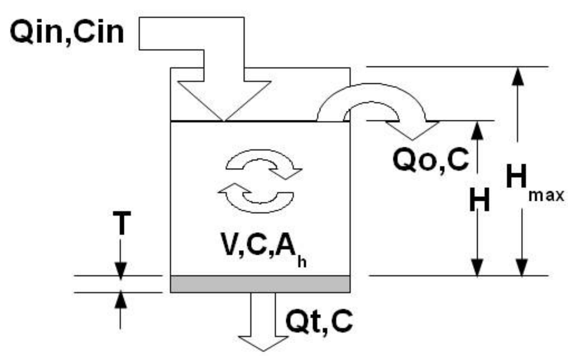

Figure 17 Pond quality routing concept (completely mixed system)

The question of what constitutes an "effective settling velocity" is difficult no matter what mechanism is postulated for simulation purposes, since the physical situation in the pond is not likely to be well represented by the equipment typically used to measure fall velocities in the lab. The vs is represented by the difference between a downward fall velocity and an upward tendency, which is a function of turbulence and concentration gradients. Other factors, such as hindering, enter the problem as well. As a result, the effective fall velocity may not only be less than the discrete settling velocity, but may vary at times. Note that the vertical mixing assumption used here will result in a removal rate which decreases as the mass of sediment in a given vertical section decreases. This assumption may be preferable to the discrete settling mechanism which provides a constant areal deposition rate from a slug of initially uniform concentration.

Solution of the Quality Routing Equations

The pond model allows losses to be calculated for intermediate conditions of mixing along the axis of the pond. This enables solutions of intermediate conditions, ranging from plug flow to completely mixed. This gives rise to the following finite form for one element:

$$2\times\left(V_2\times Cout_2 - V_1\times Cout_1\right) = (Cin_1\times Qin_1 + Cin_2\times Qin_2)\times\Delta t - (Cout_1\times Qout_1 + Cout_2\times Qout_2)\times\Delta t$$
$$- (Cout_1\times L_1 + Cout_2\times L_2)\times\Delta t$$

where

Cin_j = constituent concentration before (j=1) and after (j=2) time step (mg/m^3)

$Cout_j$ = constituent concentration before (j=1) and after (j=2) time step (mg/m^3)

V_j = element volume before (j=1) and after (j=2) time step (m^3)

L_j = loss coefficient before (j=1) and after (j=2) time step (m^3/s)

Δt = time step (s)

Terms shown as subscript '*out*' in the above equation refer to the downstream face of the element; terms shown as subscript '*in*' refer to the upstream face. Note that since a completely mixed element is assumed, concentration is the same at the interior and downstream face of the element and is denoted *Cout*. The left hand side of the equation represents changes in mass in storage in the element. Groups of terms from left to right are inflow from upstream, outflow to downstream, and removal by sedimentation or decay.

The term *L* in the above equation depends on the substance. For settleable material:

$$L = v_s \times A$$

For the first order decay pollutant:

$$L = V \times k$$

where

k = a constant parameter

The terms k and v_s are provided by the user. The element volume and area are calculated by the model each time step as (1/*NELS*) times the pond volume and area, where *NELS* is the number of sub-reactor volumes and is provided by the user. A value of *NELS* = 1 implies the whole volume of the pond is completely mixed, and a large value implies approximately plug flow.

Flow into and out of each element is distributed evenly among the elements so that overall flow rates and volumes match the information computed in the pond flow routing. It is possible, for instance, to have zero flow rate at one end of the pond, and a positive or negative flow rate at the other. An example of this would be if the outlet from the pond is closed and flows are entering. This is accounted for in distributing flows into and out of each element.

The above equation is solved in the model assuming first order pollutant decay rates (DECAYR) as:

$$C_2 = \frac{KA \times C_1 + (Cin_1 \times Qin_1 + Cin_2 \times Qin_2) \times \Delta t}{KB}$$

where

$$KA = (2 - DECAYR) \times V_1 - Q_1 \times \Delta t$$

$$KB = (2 - DECAYR) \times V_2 + Q_2 \times \Delta t$$

in which

$$DECAYR = \exp(-k \times \Delta t)$$

KA and *KB* are grouping terms

For settleable material, this equation is solved as:

$$C_2 = \frac{(KA - A \times v_s \times \Delta t) \times C_1 + (Qin_1 \times Cin_1 + Qin_2 \times Cin_2) \times \Delta t}{KB + A \times v_s \times \Delta t}$$

where

$$KA = 2 \times V_1 - Q_1 \times \Delta t$$

$$KB = 2 \times V_2 + Q_2 \times \Delta t$$

In both of the above equations, if the divisor is zero (which can only happen if the element volume and flow are both zero), C_2 is set to zero. The quality routing constituents are solved in sequence for all elements in the pond, starting at the top end. As with flow, the model produces a global mass balance for pollutant and sediment parameters. In practice, it is found that convergence errors are negligible if the input set is properly formulated.

It is acknowledged that other mixing conditions are possible. For example, short circuiting (e.g., in larger ponds with inlet and outlet near each other) can be a factor affecting mixing and losses. These more complex behaviors are not addressed in this project, and are not simulated in this version of the model.

Analysis

The model was used to explore BMP performance based on hypothetical but reasonable watershed conditions and removal rates. Two conditions were tested to evaluate detention type BMP removal performance for indicator bacteria:

- Case 1 was devised to explore the implications of a detention type BMP located at the bottom of a watershed.

- Case 2 is a more detailed set of tests examined the implications of pond exfiltration on the performance of a detention type BMP located at the bottom of a watershed.

Case 1: Theoretical Detention Pond Indicator Bacteria Removal Behavior

This case was based on results from a previous study which provided flow and indicator bacteria data. The data used in this analysis were from watershed analysis of Sawmill Creek at Ottawa, Canada, which were part of an extensive evaluation of watershed management options in the Rideau River watershed (Gore and Storrie et al., 1981). Although case by case variations will no doubt depart from what was found in the Rideau System, for purposes of this analysis this area is reasonably representative of typical mid-sized urban to suburban developments experienced in the north-eastern, mid-latitudes of North America.

Watershed parameters consistent with this case were an initial abstraction of 0.3 mm (0.01 in.) and runoff coefficient of 0.99 for impervious areas, and an initial abstraction of 3 mm (0.1 in.) and SCS curve number of 78 for pervious areas. FC pollution was simulated by assuming 20,000 no/dl concentrations in runoff from developed areas, 4,000 no/dl in undeveloped areas, and 40 no/d1 in base flow. These figures were based on results observed in monitoring and analysis undertaken locally (Rideau River Stormwater Management Study, 1981a&b). The bacterial die-off rate constant was set with a T90 (time for 90% die-off of a population) of 36 hr. This outcome was consistent with earlier work reported by Rowney et al. (1982) in the same project vicinity.

The degree of loading under undeveloped conditions was estimated, and then the impact of further urban development was assessed by simulating an increased level of development. For this example, winter months were not simulated i.e., no snowmelt was simulated. The watershed area was taken to be a catchment with a total land surface of 6,000 acres, half of which was developed. The developed area included 35% residential and 65% open land.

Figure 18 shows a typical result of the control pond analysis in terms of impact on FC. As indicated, the result is consistent with what is generally anticipated in this kind of situation. The shift from undeveloped to developed area is accompanied by an increase in the duration of elevated FC conditions. The BMP, when appropriately sized, reduces the FC conditions at the point of discharge.

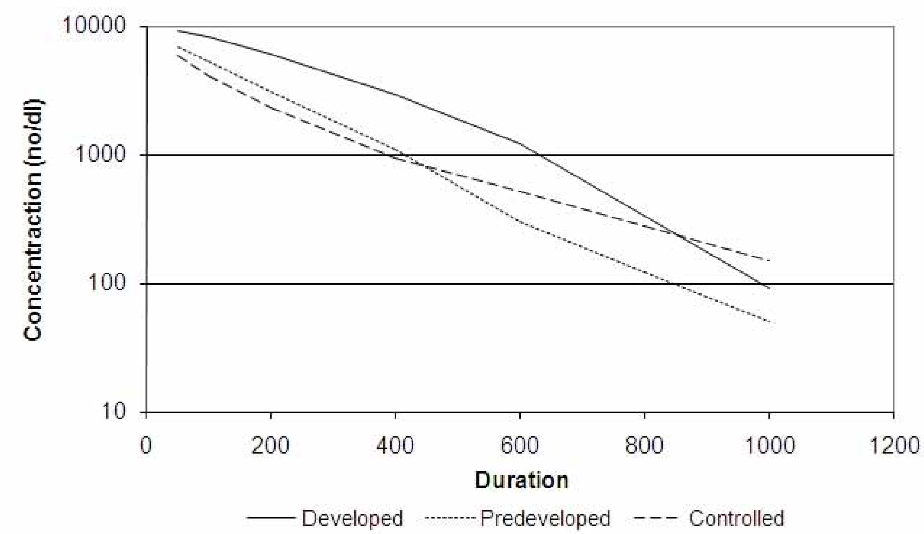

Figure 18 Typical indicator organism versus duration exceedance curves

However, the BMP performs differently at different concentrations. While conditions at high levels are improved, to the point where concentrations drop below pre-developed levels, they are predicted to exceed predeveloped at lower levels and even uncontrolled at the lowest levels presented. This result reflects the way that the indicators mix in the facility. High concentrations are reduced to moderate or low concentrations, and the discharge is spread over a longer period of time. This means that the facility tends to lead to a preponderance of moderate conditions. The facility is effective, and reduces loads over all, but the frequency/duration behavior, as a result of mixing, results in a shift towards a greater incidence of low-level conditions. The control pond selected for this example was assumed to be a single, in-line device, 2 m deep and equivalent to a 3 mm volume over the area tributary to it. This volume is not a large treatment control, and repeated simulations with other moderate sized ponds led to similar findings. The conclusion is that this shift of removal efficiency is a common finding, if measurements are provided that are sufficient to resolve the behavior of the device over all conditions.

Put differently, a conclusion that emerges from this simulation is that a significant number of measurements are required to fully determine the impact of a BMP. Modest or limited sampling will not likely be able to determine the actual performance characteristics. A substantially increased level of measurement will be required. If the example in Figure 18 is taken as an indicator of this effect, it is noted that:

- Simple random sampling will tend to show that the BMP has a neutral impact, some high and some low, compared to uncontrolled conditions, if post-event conditions are sparsely sampled.

- If sampling is done throughout a storm event including peak flow, the BMP is indeed found to be efficient in reducing indicator organism concentrations.

This bias from simple random sampling is exacerbated by the nature of the variability of the bacteria. Indicator organisms have been found to be log-normally distributed, and also have a high degree of variability.

This seems to be a significant finding in helping interpret the findings of the recent BMP database discussed by Clary et al. (2008). A BMP with the type of simulated performance exemplified in Figure 18 which predicts a significant benefit of BMP reduction in indicator organism counts, when insufficiently measured in the field may be perceived to have limited impact on water quality if the monitoring approach is not appropriate, despite the fact that the facility may actually have a significantly positive impact.

Case 2: Impact of Exfiltration on Detention Type BMP Performance

In this case, flow parameters were selected from a recent calibration where good results were obtained using the model. This effort provided a model flow series that could be used to simulate the hydraulic response of the test BMP. While the flow data from this case were useful, that case did not have adequate corresponding quality data, so indicator bacteria concentrations were taken to be the same as in Case 1.

Model Parameters

To develop some useful hydrologic model parameters, the model in this case was set up to represent a 7 acres catchment in Austin, Texas, specifically the Jollyville[11] catchment area. The rainfall series selected for this site was taken from City of Austin Records, spanning November 1995 through May 2002. Watershed parameters consistent with this case were an initial abstraction of 0.15 in. and runoff coefficient of 0.79 for impervious areas, and an initial abstraction of 0.3 in., and an SCS curve number of 67 for pervious areas. Applied to a simulation, this parameter set has good agreement with measured hydrology as represented by comparison to data sets as shown in Figure 19 through Figure 21. Figure 19 shows agreement between modeled and observed flows for over a year, while Figure 20 and Figure 21 show response to individual events. The model performance in representing hydrologic and hydraulic effects at this site was reasonable.

[11] Repurposed with the permission of Pat Hartigan, City of Austin, personal communication August 17, 2009, Figures 19-21 reproduced from City of Austin report, "City of Austin Hydrologic Model Development and Implementation", by ACR, LLC, June, 2009.

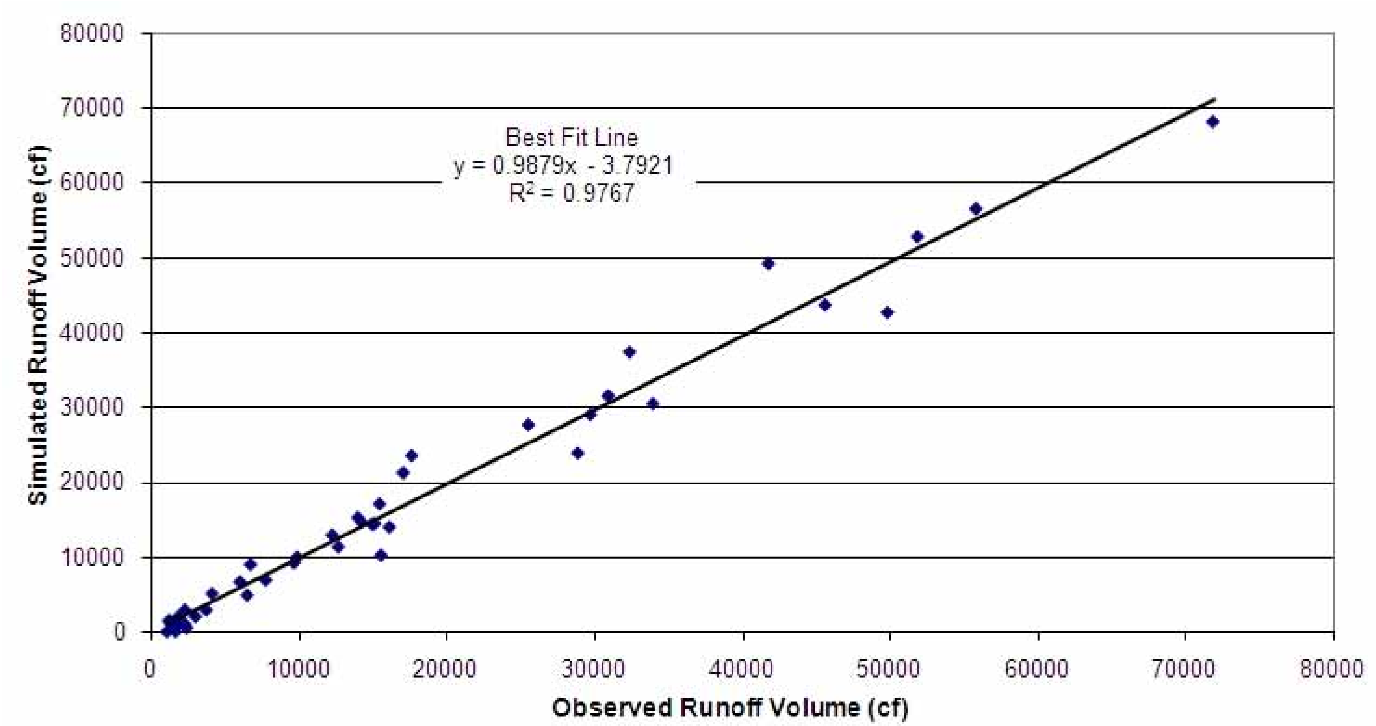

Figure 19 Model watershed adjustment based on simulated and observed runoff event volumes, Jollyville, TX, March 17, 1997 through July 17, 1998

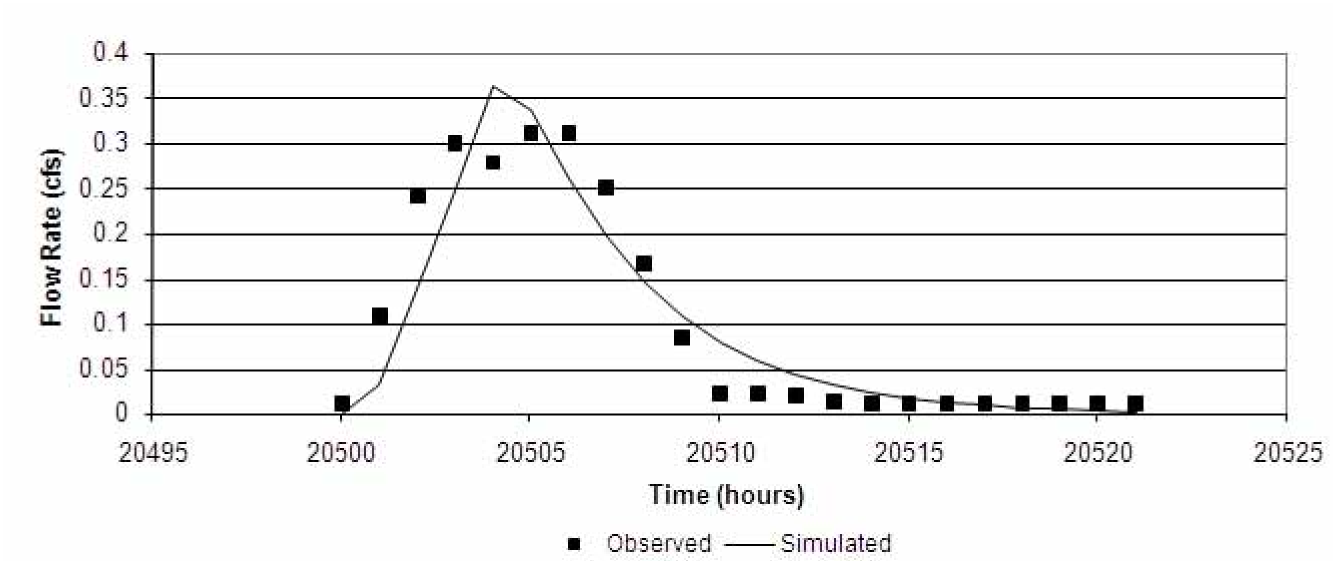

Figure 20 Observed BMP hydrograph for Jollyville, TX event May 15 - 16, 1997

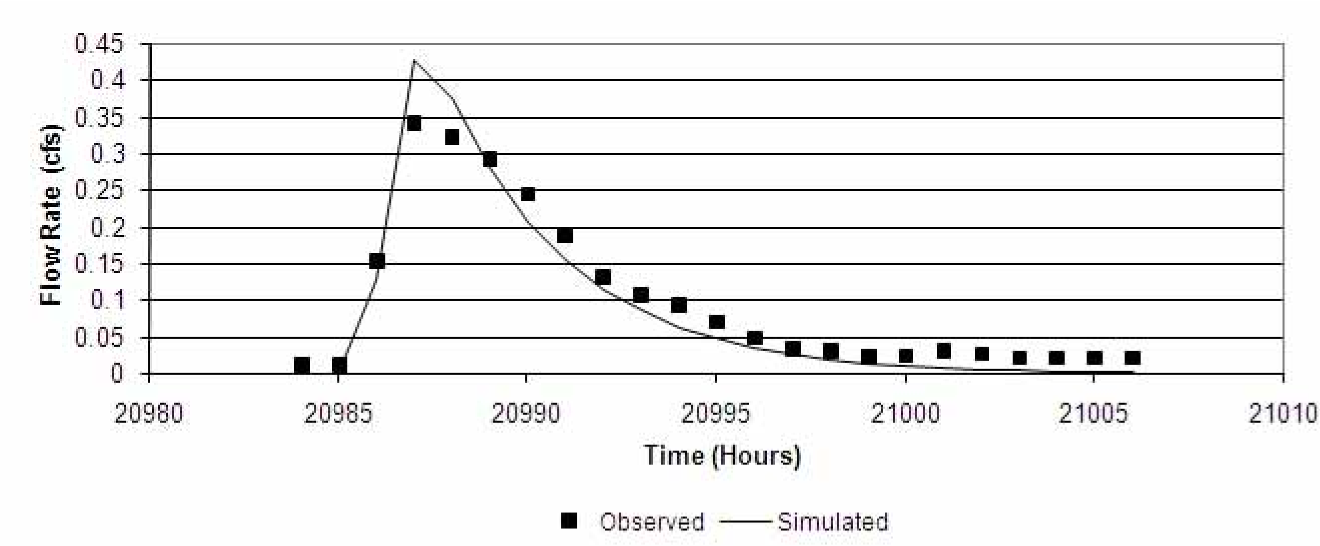

Figure 21 Observed BMP hydrograph for Jollyville, TX event June 4 - 5, 1997

BMP Performance Estimates

Because of the representative performance exhibited for Test Case 1 and 2, this model was therefore used, along with quality parameters for Case 1, to explore further possible BMP performance. The model was used to determine the efficacy of a single pond type BMP placed at the head of a watershed by evaluating the net effect of the device at the downstream end of the watershed. The end result of this evaluation is taken further to explore the impact of exfiltration on results. In addition, the development scenarios differed somewhat from what was done in Case 1. The general nature of the problem posed is as follows.

Typical Pond

If a watershed as shown in Figure 22 experiences development, the impacts and control options involve tradeoffs even in a most simple case. Consider for example a situation where a watershed has experienced development in the past, in this case the area between points B and C, but that is otherwise relatively undeveloped.

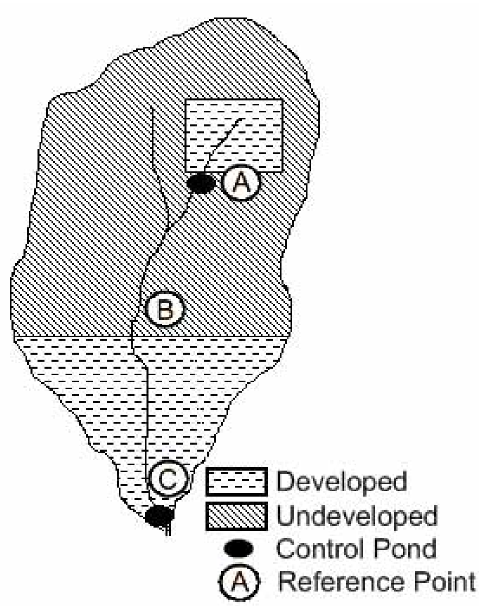

Figure 22 Test Watershed for impacts and control options of BMPs

If development occurs in the area above the watershed (e.g., at point A), then the question is how and where to provide control facilities. If the control facility for the system is at point C, a negative outcome is that there is no protection within the area, but a positive outcome is that the net effect of all contributions above that point can in principle be mitigated for all locations below that point with one BMP.

On the other hand, if a protective BMP is placed at point A, then the region immediately below point A is protected. However, uncontrolled contributions below A will have an impact on points further downstream. The positive aspect is that over some distance below A there will be protection of the system, but this is offset to some degree because both the impact and the benefits of protection of a facility at A will diminish with distance below that point. Depending on scale, the cost of a facility at A may not be recouped at all in terms of impacts below point C.

Other points could be made, but the essential question is one of BMP placement. Location and performance are tightly related. From the point of view of distributed solutions (such things as pervious paving or green roofs) the essential question of what to do at point A remains the same. To test this notion, and at the same time illustrate the use of the pathogen routing tool described above, a case study was developed and simulated. Three development cases were considered, namely: 1) the status quo (no development above point B), 2) projected new development above point A but no control pond, and 3) development above point A with a control pond just below that point. In all cases, the impact of the changes, i.e., development and BMP placement in the watershed, were considered from the perspective of a location further downstream. The total watershed area was 100 hectares (258 acres), which was mostly undeveloped except for a developed section of 3 hectares (7.4 acres). For this test case, an added development of 3 hectares (7.4 acres) was under consideration. The control pond used was sized to fit the target developing area, with a volume of 370 m^3 and a weir 3-m long set at a height of 1.5 m.

The BMP as placed has a significant control impact in its near vicinity, reducing the post developed concentrations to pre-developed levels. What is apparent is that the pond has virtually no value further downstream. As shown in the figures of exceedance curves[12] for the undeveloped case (Figure 23), developed case without pond (Figure 24) and developed with pond (Figure 25), there is virtually no detectable difference in conditions in-stream once a reasonable distance is traversed, in this case 1,000 m below where the pond was placed. This represents a realistic possibility. What is happening is that below the pond, natural sources of indicator bacteria distributed along the stream continue to contribute to the system. As distance increases, natural die-off eliminates contributions further upstream, so those amounts are no longer there. Therefore, given enough distance, the location of a pond upstream may be immaterial for management of indicator organisms downstream. Whether natural or increased by development, controlled or uncontrolled, contributions from upstream are not experienced past that distance.

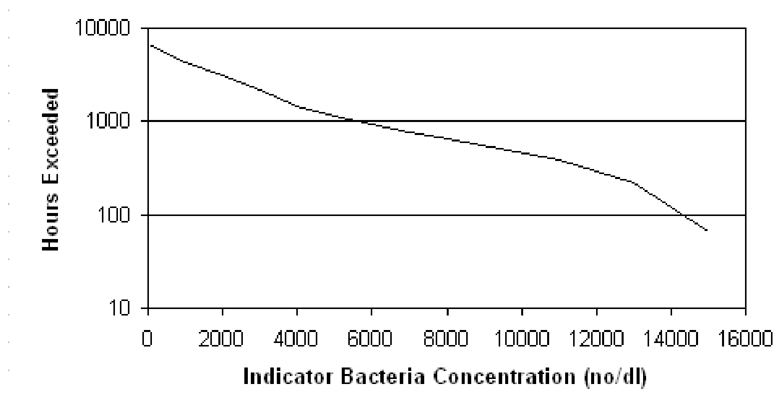

Figure 23 Exceedance curve for pre-developed case

[12] The exceedance curve shows the hours over the total time period (in this case one year) during which a particular concentration is exceeded. For example, in Figure 23 an indicator bacteria concentration of 14,500 no/dl is exceeded approximately 100 hours.

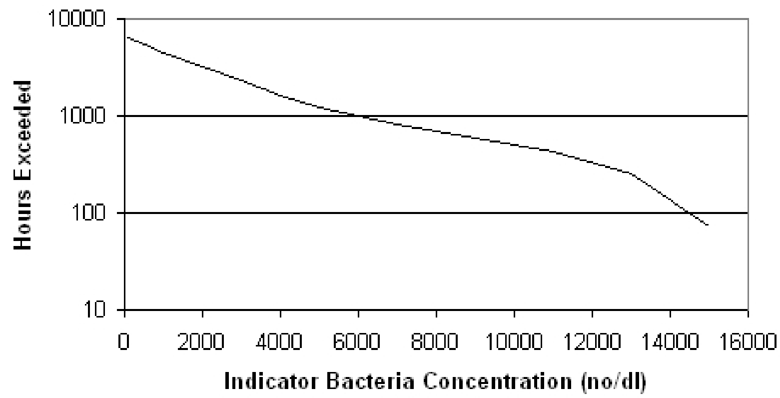

Figure 24 Exceedance curve for developed case, no pond

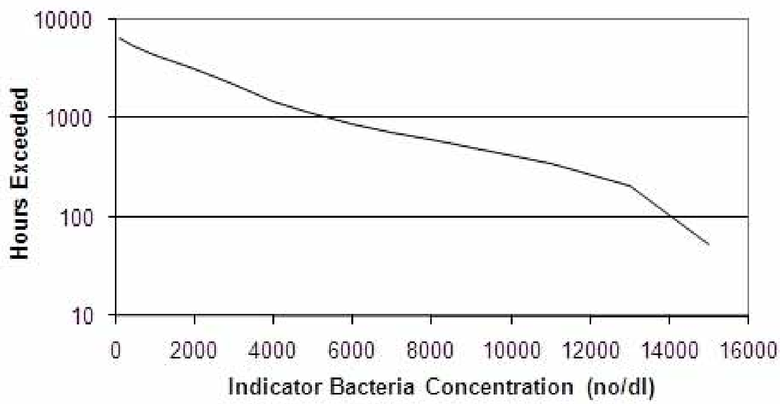

Figure 25 Exceedance curve for developed case, with pond

The test case is very simple and is not universal in detail, since as noted above it is only reflective of a specific set of numbers. These numbers are reasonable but cannot be assumed to be exactly what is experienced at any particular site; however, the principles are valid. At some distance below a site, controls within the site are (from at least this perspective) immaterial.

This is an old principle, the notion of assimilative capacity. What is not generally acknowledged in common practice as to BMP performance is the scale dependence of a solution. While water volume and rate changes can have effects that accumulate over long distances, other parameter shifts may have effects that manifest over rather different time and distance scales. The value of protection of standard ponds to resolve indicator bacteria is particularly difficult to defend on the basis of quantitative measures. Partly, this is because the current suites of commonly used indicator organisms are ubiquitous and highly variable. Even neglecting the questionable epidemiological evidence for control of indicators, it is fair to question the value of indicator control. If as in this case, a relatively small control pond will minimize flooding and erosion, but a much larger pond would be required to control indicators, it is fair to question the value of the expenditure for additional volume control beyond flood and erosion if the impact on indicators is small and not measurable downstream. Other parameters are of interest, but the major point is that scale dependence of solutions has an impact on benefits, and should presumably have an impact on selection. In short, end-of-pipe criteria alone may be of limited value for indicator organism control. Investment in larger-sized BMPs that pond water beyond the flood protection or quantity control volume should be made only with a specific understanding of the limited in-stream benefits predicted for that investment. Investment in treatment train approaches, other technologies or distributed controls may potentially minimize anthropogenic indicator discharge exceedances, but this will not resolve, nor should it, natural background indicator organism concentrations.

Pond Exfiltration

The previous example was straightforward. A more interesting case is found in a scenario specifically targeted by this task, which was to evaluate the potential efficacy of ponds as influenced by exfiltration through the bottom. In this case, the hypothesis was that pond effectiveness could be enhanced by drawing down the facility through exfiltration between events in areas where geological conditions permitted this condition.

This evaluation was done by establishing a set of conditions that contrasted a control facility performance under identical conditions except that the one pond had an impervious bottom, while the second had a pervious bottom. Conditions in the second case approximated either a highly pervious groundwater area, or an effective filter and underdrain system. The selected land areas were:

- A total watershed area of 25 hectares.
- A development area of 10% of the total area.
- A shift from 15% imperviousness to 95% imperviousness after development.

The area of interest was therefore a section of 2.5 hectares, which during development showed a significant increase in imperviousness, with a corresponding decrease in initial abstraction and increase in runoff coefficient. Otherwise, conditions were consistent with the parameters and scenarios discussed above.

The plot of flow duration curves, Figure 26, shows an increase in flow duration after development, but the changes are not major. Even so, the pond is predicted to do little more than a modest reduction in flow duration, and is not able to match the pre-developed flow/exceedance conditions. This is because the pond is an overflow type of facility, and is typically full when a new event comes along. In this situation, i.e., when the pond is full and overflows are significant, the routing benefits of a pond are minimal. This is a physically reasonable situation, because it represents a case where the intent of the pond is to achieve water quality control rather than water quantity control. There are two benefits in operation of this type of pond. One is that the flows that are captured are held for the longest possible time before a new event arrives, which provides more time for settling and indicator die-off. The second benefit is that the retained volume can reduce scouring and re-suspension as a new event sends flows into the pond.

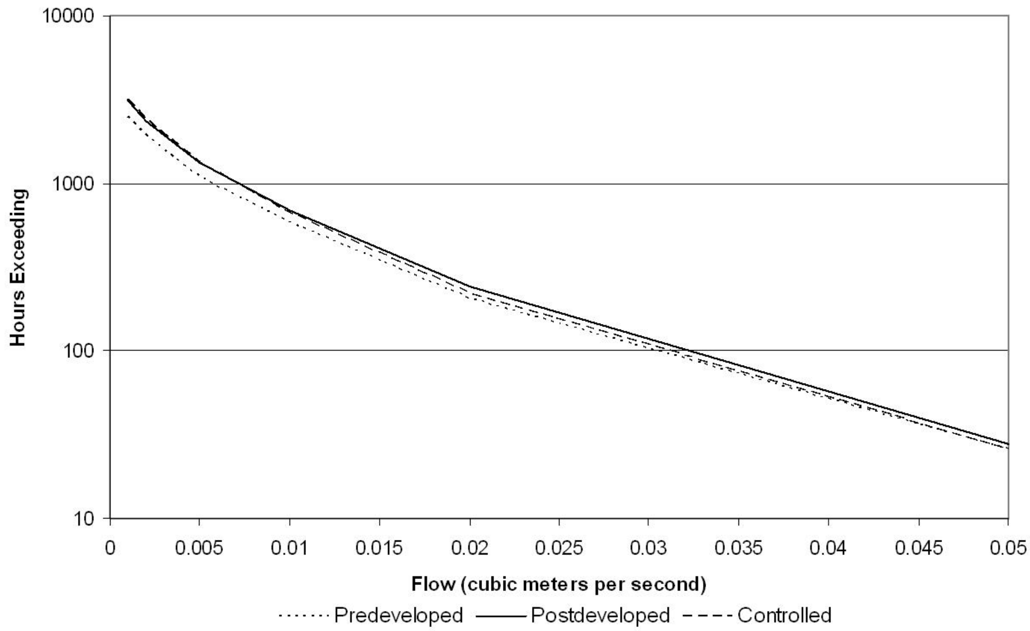

Figure 26 Flow exceedance curves, watertight pond bottom

The water quality behavior of the facility is as shown in Figure 27, and it is noted that the indicator bacteria duration curves show a different behavior from the flow curves. The observed change in quality/duration behavior as land changes from pre-developed to developed conditions is substantial. This change is consistent with expected physical behavior, because the concentrations of indicators in urban runoff can be much greater than in an undeveloped watershed, i.e., unless an agricultural land use with significant bacterial loadings is being developed, and quality can change substantially even if quantity does not.

The pond clearly does have some impact on concentrations, because for the most part the controlled curve is lower than the uncontrolled post-developed curve. It does not match the pre-developed curve, but a significant removal efficacy is predicted. This is a result of die-off as some die-off occurs in the pond between events because of the duration over which storm volumes are retained. When die-off is set to zero, the post-developed controlled curve is quite comparable to the uncontrolled curve. Without die-off, the only buffering effect of the pond is by mixing, rather than mass reduction, and this is not highly effective in reducing the impact of indicator bacteria on the system.

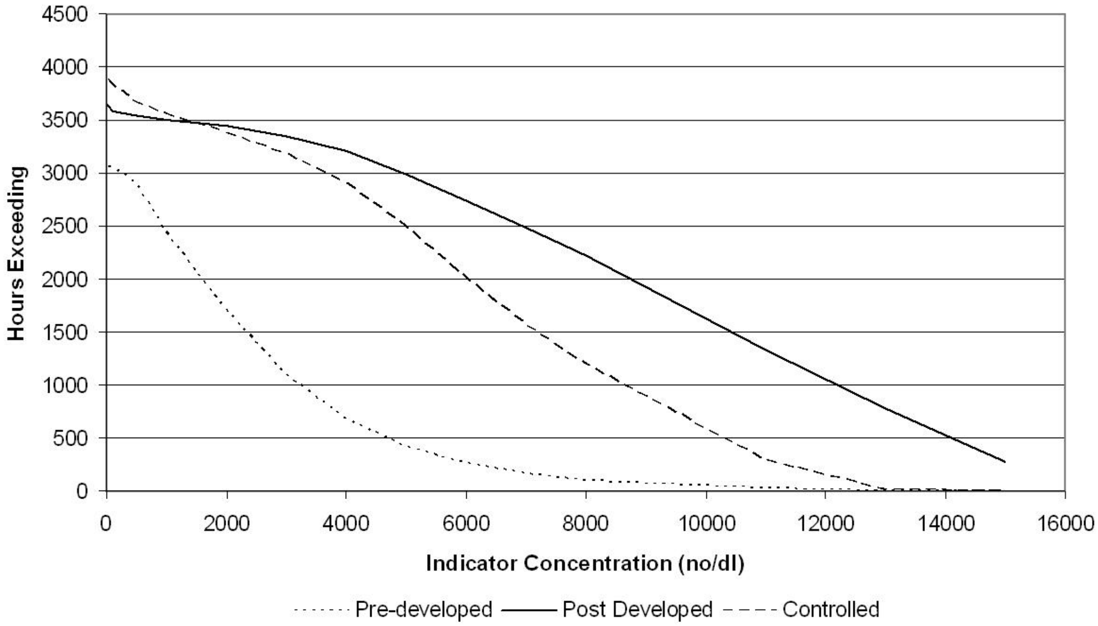

Figure 27 Indicator concentration exceedance curves, watertight pond bottom

The controlled curve also shows some interesting shifts in the nature of its impact. The unfortunate result of this situation is that the pond actually is inducing an increase in concentration/duration over part of the curve, even when compared to no control at all. This negative shift is unfortunately right where it is problematic because it is located at a very sensitive part of the curve. In the region where concentrations are in the 100 no/dl to 200 no/dl range, durations actually increase from about 3600 to 3800 hr over the simulation period (Figure 27). This change means that the system can be interpreted to be quite negatively impacted by development and that the pond actually exacerbates this result.

The physical reason for this situation is that the displacement stored runoff from one event by inflows from another subsequent event is tending to flush the retained volume into the environment. The flushed volume will be lower in concentration than its maximum, because of die-off between events, but is not being held long enough to fully inactivate or treat the organisms, so the long term result is an increase in low level indicator discharge duration. Flow exceedance curves normalize results; however, actual indicator organism concentrations are data dependent, i.e., pollutant and inter-event period dependent. The limited predictability of ponds as a management option to control indicator organisms severely limits this technology in this regard, and may help explain some of the recent findings that question the efficacy of ponds for indicator control.

A much improved result is obtained when the pond is allowed to draw down through infiltration between events. As shown in Figure 28, the drawdown causes the controlled curve to meet and in fact better the results in the uncontrolled post-developed curve to the point where it matches or is lower than the pre-developed case. The shift in volume control is not extreme, and the system behaves much as it did in the pre-developed case, but control is accomplished. What is interesting is that although the drawdown does not represent a qualitative shift in behavior from a quantity control perspective, it has major consequences for quality.

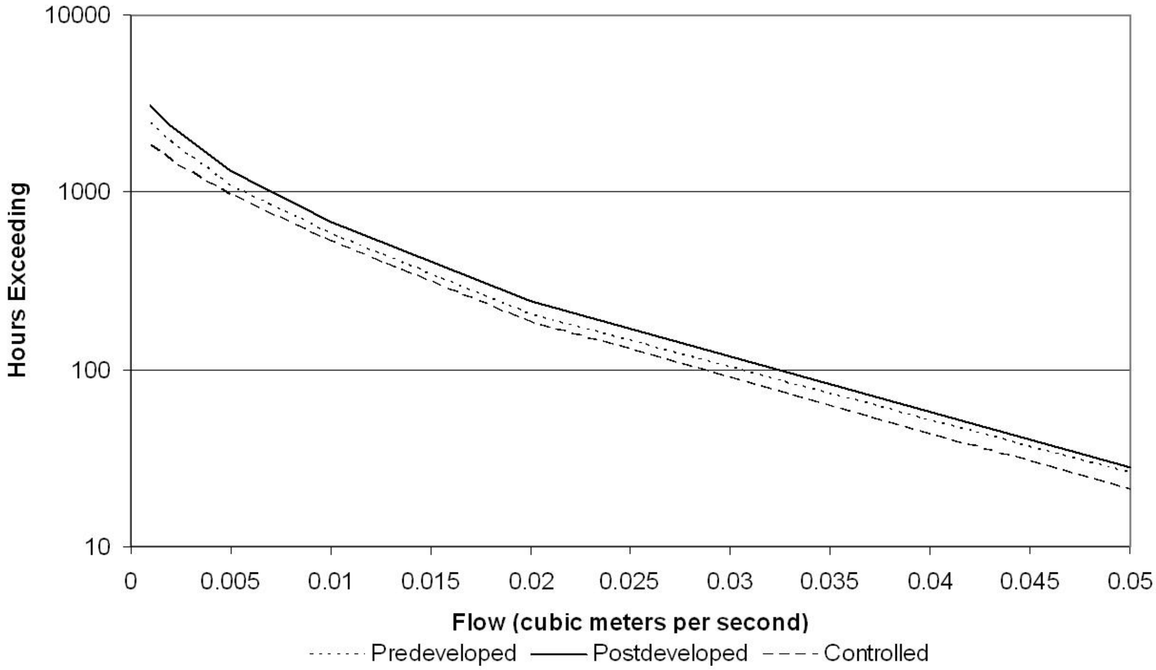

Figure 28 Flow exceedance curves, exfiltrating pond bottom

The result in Figure 29 demonstrates this case. The curve still does not perfectly match pre-developed conditions over its whole range, but part of that range is below the pre-developed curve. More importantly, the reduction is greatest in the lower end of the spectrum, where durations in the 100 no/dl to 200 no/dl range drop by about one third. The end result of this change is that the system is well controlled in the more frequent small event range, and less controlled during major events.

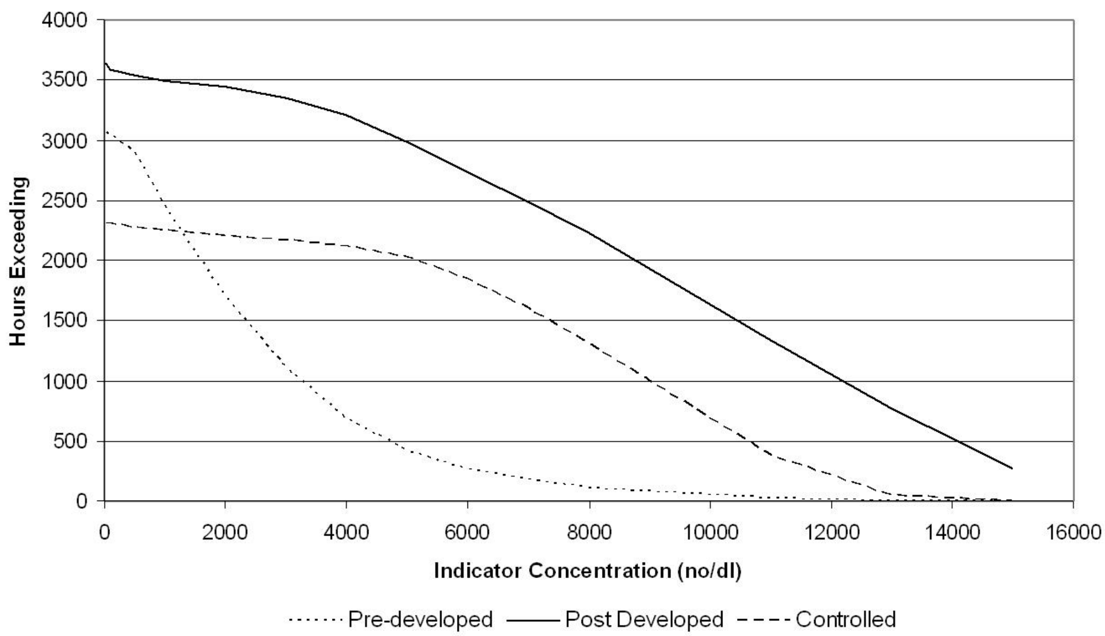

Figure 29 Indicator concentration exceedance curves, exfiltrting pond bottom

Figure 30 provides an added insight to the differences between these cases. As shown, the case with the exfiltrating (leaky) bottom demonstrates a small increase in concentrations over the higher end of the range when compared with the watertight pond. This is quite reasonable, because it reflects the difference in mixing between the two.

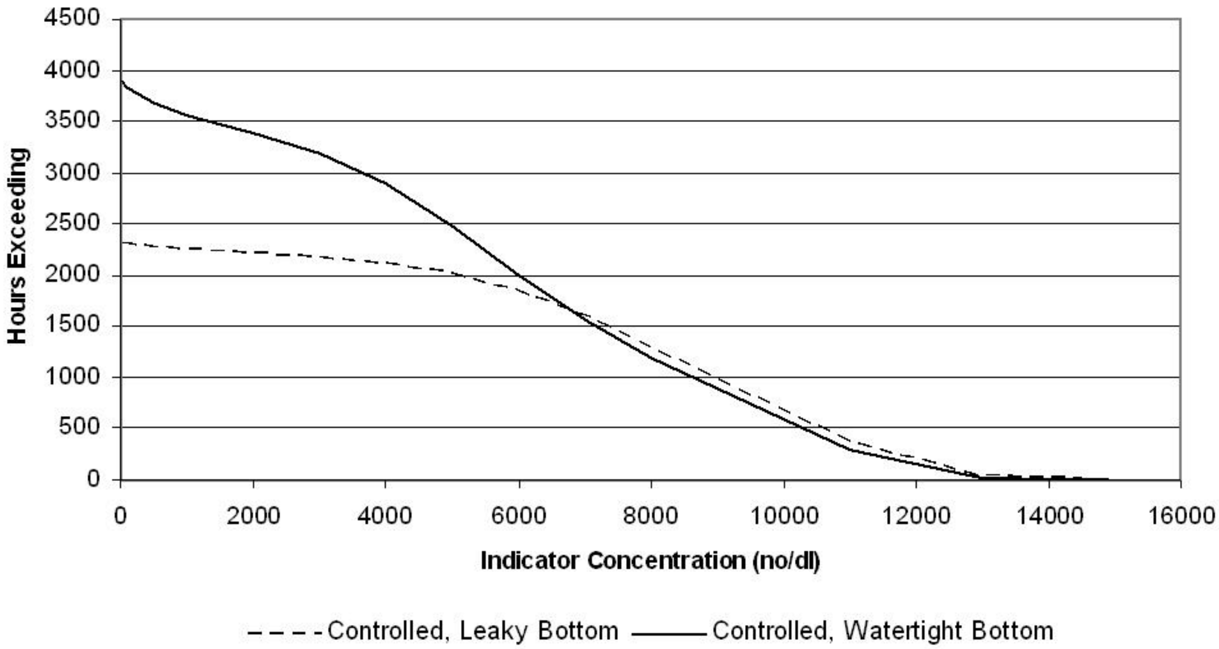

Figure 30 Comparison of leaky and watertight BMP performance

In both ponds, a major event swamps the small pond, regardless of the antecedent condition so both of the cases show significant discharges. However, the pond with the leaky bottom shows a couple of differences. There is on average less volume in the infiltrating pond, so there is less dilution. With the watertight case, there will be more volume in the pond on average when an event occurs. This means that in the watertight case there is a modest dilution of large events that overtop the system, compared to the leaky case where there is not enough water to dilute large events. The result is not of much practical significance, but it is consistent with the hydraulics and hydrology of the problem. A combined assessment of Figure 29 and Figure 30 lends credence to the LID strategy which tends to promote infiltration for the smaller storms, but also indicates this approach may be of limited benefit for indicator organism management for larger events.

Topic Area 4 Conclusions

There are a number of conclusions associated with this set of predicted outcomes. In general, future research would be of benefit in this area.

A model was developed that is able to simulate the behavior of a detention or retention pond, or a hydraulically similar BMP, which has leakage through the bottom. The model is based on an existing continuous simulation model which has the capability of simulating watershed runoff, in-stream transport and a range of BMPs. It also has the ability to directly develop exceedance curves and other performance statistics. This result appears to have the potential for wide use in U.S. water resources practices, because the increasing interest in infiltration dependant BMPs demands more attention to the impact of groundwater losses on BMP performance.

A case study has been performed which reinforces the notion that in-stream processes need to be specifically considered when the positive merits of a BMP placement are at issue. The illustrative example detailed in this chapter indicated that under some conditions, the benefit of placing a BMP to mitigate changes from a pre-developed case to a developed case may be arguable for indicator organisms. The range of scale dependencies that affect pollutants of interest is substantial. There may be benefits from placing a BMP that moderates other quantity effects though it has a negligible benefit from the perspective of one or more water quality indicators. The conclusion is that control of indicator organisms by a BMP should be evaluated from the perspective of in-stream impacts if the actual benefit of the BMP is of interest, and that in cases where competitive sources are significant and assimilative losses are substantial, the scale of benefit of the BMP may be small.

A case study has been performed which suggests that there is reason to expect that ponds which operate either as a filter or as a discharge through to groundwater through the bottom may be a more efficient alternative, particularly for smaller and more frequent events, than ponds that attempt to retain volume for as long as possible. The reason for this is that infiltration between events eliminates indicator loads (as opposed to a rapid release rate which simply discharges them) and increases the capture success of subsequent events by making more volume in the BMP available. It appears that this case study can have a significant positive impact on BMP performance.

The potential for exfiltration-dependant or exfiltration-enhanced BMPs seems clear. However, the applicability of such a device in any particular case will depend on the availability of a volumetric capacity in the soils nearby and geologic conditions. The present model enables an analysis of exfiltration-enhanced BMPs, but does not have a mechanism for simulating this capacity. Simple mounding models or base-flow algorithms are possible options, as are more complex fully featured groundwater models. Research into a comprehensive but simple way to simulate the water loss from the BMP and the accumulation in the surrounding soils would potentially lead to a model that is useful in a wider range of practical contexts.

The scale dependency of long term dynamic BMP discharges and the interaction of those discharges with ubiquitous sources of pollutants lead to a complex in-stream impact behavior. The cost implications of BMP infrastructure in the U.S. are substantial, and achieving a best return on investment in this area is important. Given the complexity of their interactions, research into the best way to plan and implement BMPs is warranted. This could be done by examining a range of parameters typically dealt with in BMPs, developing a simple way to simultaneously estimate scales of influence, and deriving placement principles that would not only ensure performance requirements in receiving waters are met but also that redundant facilities are not inadvertently implemented. A particular driver in this area is the emerging importance of LID technologies, which distribute controls and therefore increase the potential for unforeseen interactions.

Chapter 9 References

Abda, F., Azbaid A., Ensminger, D., Fischer, S., Francois, P., Schmitt, P., Pallares, A., (2008), "Ultrasonic device for real time sewage velocity and suspended particals concentration measurements", 11th International Conference on Urban Drainage.

Achleitner, S., Rauch, W., (2007), Vol. 21, Iss. 9, pp 1487-1503, "Increases of River Base Flow by Hydropower Gate Operation for Mitigation of CSO Impacts – Potential and Limitation", Water Resources Management.

Aiguier, E., Chebbo, G., Bertrand-Krajewski, J.L., Hedges, P. and Tyack, J.N., (1995), "Methods for Determining the Settling Velocity Profiles of Solids in Storm Sewage." Proceedings of IAWQ/IAHR's International Conference on Sewer Solids - Characteristics, Movement, Effects and Control, University of Abertay Dundee, Scotland, 1995.

Alfaqih, L, Pitt, R. and Hale, D., (2008), "Application of environmental decision analysis framework to identify potential E. coli sources in Lake Tuscaloosa Watershed, Alabama", 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

The American Public Works Association (APWA). 1992. "Nationwide Costs to Implement BMPs".

Ashley, R., Bertrand-Krajewski J. and Hvitved-Jacobsen T., (2005), "Sewer solids - 20 years of investigation", Water Science & Technology, Vol. 52, No. 3, p. 73-84.

Ashley, R., Crabtree, B., Fraser, A., and Hvitved-Jacobsen, T., (2003), "European Research into Sewer Sediments and Associated Pollutants and Processes", ASCE Journal of Hydraulic Engineering, Vol. 129, No. 4, p. 267-275.

Ashley, R.M., Dudley, J., Vollertsen, J., Saul, A.J., Jack, A. and Blanksby, J.R., (2002), "The effect of extended in-sewer storage on wastewater treatment plant performance", Water Science and Technology, Vol 45 No 3 pp 239–246.

Barraud, S., Gibert, J., Winiarski, T. and Bertrand Krajewski, J.-L., (2002), "Implementation of a monitoring system to measure impact of stormwater runoff infiltration", Water Science & Technology, Vol. 45, Iss. 3, pp 203–210.

Barrett, M., (2003), "Performance, Cost, and Maintenance Requirements of Austin Sand Filters", J. Water Resour. Plng. and Mgmt. Vol. 129, No. 3, pp. 234-242.

Barrett, M., (2009), Personal communication with Dr. Michael Barrett, University of Texas at Austin.

Bell, J., Serrano, R. and Schmidtbauer, T., (2005), "Evaluation of the potential regrowth of fecal coliforms bacteria in biosolids between sewage treatment plants and land application sites in Solano County, California", Biosolids Land Application Period, Biosolids Research and Education Trust Fund.

Blum, P., (2005), "Understanding Viability of Pathogens during Disinfection", Water Environment Research Foundation, Water Intelligence Online UI. 200510WF00HHE1.

Bolster, C.H., Walker, S.L., and Cook, K.L., (2006), "Comparison of Escherichia coli and Campylobacter jejuni Transport in Saturated Porous Media", Journal of Environmental Quality, Vol. 35, pp 1018-1025.

Boner, M.C., Harper, S.R., Patrick, G.C., Ghosh, D.R and Cairns, W.L., (1994), "Optimization of Vortex Separator Removal Efficiencies for CSO Treatment", Water Environment Research Foundation, Project 92-TCR-2.

Bowen, R. and Williams, T.O., (2006), "Reactivation and Regrowth of Fecal Coliforms in Anaerobically Digested Biosolids", Water Environment Federation Technical Practice Update.

Bradford, S.A. and Toride, N., (2007), "A Stochastic Model for Colloid Transport and Deposition", Journal of

Environmental Quality, Vol. 36, pp. 1346-1356.
Bradford, S.A., Yadata, F.T., and Pachepsky, Y., (2006), "Transport of Giardia and Manure Suspensions in Saturated Porous Media", Journal of Environmental Quality, Vol. 35, pp. 749-757.
Bradford, S.A., Yadata F.T. and Yan J., (2006), "Transport of Colliphage in the Presence and Absence of Manure Suspension", Journal of Environmental Quality, Vol. 35, pp. 1692-1701.
Bratieres, K., Fletcher, T.D., Deletic, A., Alcazar, L., Le Coustumer, S. and McCarthy D.T., (2008), "Removal of nutrients, heavy metals and pathogens by stormwater biofilters", 11th International Conference on Urban Drainage Edinburgh, Scotland, UK, 2008.
Brombach, H., Weiss, G. and Pisano, W.C., (2008), "Clarifier-type CSO Tanks: Hydraulic Design for Optimum Sedimentation Efficiency", 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.
Burns, L.A., (2000), "Exposure Analysis and Modeling System (EXAMS): User Manual and System Documentation", National Exposure Research Laboratory, ORD, EPA, EPA/600/R00/081, Revision A
Camp, T. R. (1946) "Sedimantation and Design of Settling Tanks" Transactions, ASCE, Vol. 111, paper no. 2285, p. 895-958.
Camper, A., (2009), Personal communication, Center for Biofilm Engineering, Montana State University.
Camper, .K., Hayes, J.T., Sturman, P.J., Jones, W.L. and Cunningham, A.B., (1993), "Effects of Motility and Adsorption Rate Coefficient on Transport of Bacteria through Saturated Porous Media", Applied and Environmental Microbiology, Vol. 59, No. 10, pp. 3455–3462.
Campisano, A., Creaco, E. and Modica, C., (2007), "Dimensionless Approach for the Design of Flushing Gates in Sewer Channels", Journal of Hydraulic Engineering, Vol. 133, Iss. 8, pp 964-972.
Characklis, G.W., (2009), Personal communication, Environmental Sciences and Engineering, University of North Carolina-Chapel Hill, 2009.
Chu, Y. Jin, Y. Baumann, T. and Yates, M.V., (2003), "Effect of Soil Properties on Saturated and Unsaturated Virus Transport Through Columns", Journal of Environmental Quality, Vol. 32, pp. 2017-2025.
Cinotto, P.J., (2005), "Occurrence of Fecal-Indicator Bacteria and Protocols for Identification of Fecal-Contamination Sources in Selected Reaches of the West Branch Brandywine Creek, Chester County, Pennsylvania", U.S. Geological Survey Scientific Investigations Report.
Clary, J., (2009), Personal communication, August 17, 2009, Jane Clary of Wright Water Engineers.
Clary, J., Jones, J., Urbonas, B., Quigley, M., Strecker, E. and Wagner, T., (2008), "Can Stormwater BMPs Remove Bacteria? New Findings from the International Stormwater BMP Database", 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.
Dalrymple, R. J., S. L. Hodd, and D. C. Morin., (1975), "Physical and Settling Characteristics of Particulates in Storm and Sanitary Wastewaters", EPA-670/2-75-011 (NTIS PB 242 001), EPA.
Darby, J., Heath, M., Jacangelo, J., Loge, F., Swaim, P., Tchobanoglous, G., (1995), "Comparison of UV Irradiation to Chlorination: Guidance for Achieving Optimal UV Performance", Water Environment Research Foundation.
Davis, A. P., Hunt, W. F., Traver, R. G., and Clar, M., (2009), "Bioretention Technology: Overview of Current Practice and Future Needs", Journal of Environmental Engineering, Vol. 135, No. 5, pages 109 -118.
Dietrich, J.P., Basagaoglu, H., Loge, F.J. and Ginn, T.R., (2003), "Preliminary assessment of transport processes influencing the penetration of chlorine into wastewater particles and the subsequent inactivation of particle-associated organisms", Water Research, Vol. 37, No. 1, pp. 139-49.
Dietrich, J.P., Loge, F.J., Ginn, T.R. and Basagaoglu, H., (2007), "Inactivation of particle-associated microorganisms in wastewater disinfection: Modeling of ozone and chlorine reactive diffusive transport in polydispersed suspensions", Water Research, Vol. 41, No. 10, pp. 2189-2201.
Driscoll, E. D., DiToro, D., Gaboury, D., and Shelly, P., (1986), "Methodology for Analysis of Detention Basins for Control of Urban Runoff Quality" Report No. EPA 440/5-87-01 (NTIS No. PB87-116562), EPA, Washington, DC.
EPA, (1983), "Results of the Nationwide Urban Runoff Program", Volume I, Final Report. NTIS PB84-185552, EPA, Water Planning Division, Washington, DC. 1983.
EPA, (1986a), Design Manual: Municipal Wastewater Disinfection, EPA/625/1-86/021. ORD, Cincinnati, OH.
EPA, (1986b), Ambient Water Quality Criteria for Bacteria – 1986, EPA-440/5-84-002. Washington, DC.
EPA, (1999), "Alternative Disinfectants and Oxidants Guidance Manual", EPA Office of Water, EPA 815-R-99-014.
EPA, (2006), "Real Time Control of Urban Drainage Networks", EPA ORD, Washington, DC, EPA/600/R-06/120.

EPA, (2007a), “Report of the Experts Scientific Workshop on Critical Research Needs for the Development of New or Revised Recreational Water Quality Criteria”, EPA Office of Water and ORD, EPA 823-R-07-006, http://www.epa.gov/waterscience/criteria/recreation/experts/index.html

EPA, (2007b), “Report of the Experts Scientific Workshop on Critical Research Needs for the Development of New or Revised Recreational Water Quality Criteria: Draft Executive Summary by Workgroup Chairs”, EPA Office of Water and ORD, EPA 823-R-07-007, http://www.epa.gov/waterscience/criteria/recreation/experts/index.html

EPA, (2009), “Guidance on the Development, Evaluation, and Application of Environmental Models”, EPA CREM, Washington, DC, EPA/100/K-09/003.

Ferguson, C., de Roda Husman, A.M., Altavilla, N. Deere, D. and Ashbolt, N., (2003), “Fate and Transport of Surface Water Pathogens in Watersheds”, Critical Reviews in Environmental Science and Technology, Vol. 33, No. 3., pp. 299 - 361.

Ferguson, C., de Roda Husman, A.M., Altavilla, N. Deere, D. and Ashbolt, N., (2003), “Fate and Transport of Surface Water Pathogens in Watersheds”, Critical Reviews in Environmental Science and Technology, Vol. 33, No. 3., pp. 299 - 361.

Feroz, M., Jensen, M. and Lindell, J.E., (2007), “The Lake Mead Intake 3 Water Tunnel and Pumping Station, Las Vegas, Nevada, USA”, Rapid Excavation Tunneling Conference Proceedings, Society for Mining, Metallurgy and Exploration.

Field, R., Clark, S., Fan, C-Y, and O’Connor, T.P., Pitt, R., (2002), “Annotated Bibliography of Urban Wet Weather Flow Literature from 1996 through 2000”, EPA ORD (http://www.epa.gov/ednnrmrl/publications/literaturereviews/literaturereview1996-2000/index.htm).

Field, R., Sullivan, D. and Tafuri, A.N., Ed., (2004), Management of Combined Sewer Overflows, CRC Press LLC, Lewis Publishers (ISBN 0-56670-636-X).

Francy, D.S., Hart, T.L., and Virosteck, C.M., (1996), “Effects of Receiving-Water Quality and Wastewater Treatment on Injury, Survival, and Regrowth of Fecal-Indicator Bacteria and Implications for Assessment of Recreational Water Quality”, U.S. Geological Survey Water-Resources Investigations, Report 96-4199.

Gannon, J.T., Manilal, V.B. and Alexander, M., (1991a), “Relationship between Cell Surface Properties and Transport of Bacteria through Soil”, Applied and Environmental Microbiology**,** Vol. 57, No. 1, pp. 190-193.

Gannon, J., Tan, U., Bavyey, P. and Alexander, M., (1991b), “Effect of Sodium Chloride on Transport of Bacteria in a Saturated Aquifer Material”, Applied and Environmental Microbiology, Vol. 57, No. 9, pp 2497–2501.

Ghayeni, S.B.S. Beatson, P.J., Schneider, R.P. and Fane, A.G., (1998), “Adhesion of waste water bacteria to reverse osmosis membranes”, Journal of Membrane Science, Vol. 138, No. 1, pp 29-42.

Gore & Storrie Ltd. and Proctor & Redfern Ltd. (1981), “Executive Summary Report on Rideau River Stormwater Management Study, Phase I”, Rideau River Stormwater Management Study, Ottawa. The Regional Municipality of Ottawa Carleton.

Gray, D., (2000), Qualitative Review of Epidemiology Studies, in Regional BEACH Program. Proceedings: U.S. Environmental Protection Agency, Office of Water, EPA 823-R-00-003, 43-45. http://www.epa.gov/waterscience/beaches/proceedings/Beach-99.pdf.

Gruning, H., and Hoppe, H., (2008), “Spectrophotometric applications in wastewater and storm water sewers”, 11th International Conference on Urban Drainage.

Guber, A.K., Shelton, D.R., and Pachepsky, Ya. A., (2005), “Effect of Manure on Eschericia coli Attachment to Soil”, Journal of Environmental Quality, Vol. 34, pp 2086-2090.

Gummerman, R.C., Culp, R.L., and Hansen, S.P., (1979), “Estimating Water Treatment Costs”, EPA, EPA600/2-79-162a.

Hatt, B.E., Fletcher, T.D. and Deletic, A., (2008), “Improving stormwater quality through biofiltration: Lessons from field studies”, 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

Hazen, A. (1904). “On Sedimentation.” Transactions, ASCE, Vol. 53.

Heaney, J.P., Sample, D. and Wright, L., (2002), “Costs of Urban Stormwater Control”, EPA, Cincinnatti, OH, EPA-600/R-02/021.

Higgins, M.J., Chen, Yen-Chih, Murthy, S.N., Hendrickson, D., Farrel, J. and Schafer, P., (2006), “Reactivation and growth of non-culturable indicator bacteria in anaerobically digested biosolids after centrifuge dewatering”, Water Research, Vol 41, No. 3.

Hipsey, M., Brookes, J., Regel, R., Antenucci, J., Burch, M., (2006), “In situ Evidence for the Association of Total Coliforms and Escherichia coli with Suspended Inorganic Particals in an Australian Reservoir”, Water, Air, & Soil Pollution, Vol. 170, Iss. 1, pp 191-209.

Hulot, F.D. Lacroix, G., Lescher-Moutoue, F., and Loreau, M., (2000), “Functional diversity governs ecosystem response to nutrient enrichment” Nature, Vol. 405, No. 6784, pp 340-343.

Hunter, A.J., Northup, D.E., Dahm, C.N., and Boston, P.J., (2003), “Persistent coliform contamination in Lechuguilla cave pools”, Journal of Cave and Karst Studies, Vol. 66, No. 3, pp 102-110.

Hyer, K. E. and Moyer, D. L., (2003), “Patterns and Sources of Fecal Coliform Bacteria in Three Streams in Virginia, 1999-2000” U.S. Department of the Interior, U.S. Geological Survey Water-Resources Investigations Report 03-4115.

Jamieson, R. Joy, D.M., Lee, H., Kostaschuk, R. and Gordon, R., (2005a), “Transport and deposition of sediment-associated *Escherichia coli* in natural streams”, Water Research. Vol. 39, No. 12, pp 2665-2675.

Jamieson, R. Joy, D.M., Lee, H., Kostaschuk, R. and Gordon, R., (2005b), “Resuspension of Sediment-Associated *Escherichia coli* in a Natural Stream”, Journal of Environmental Quality, Vol. 34, pp 581-589.

Jones, J. and Clary, J., (2009), Personnal Communication with Jon Jones and Jane Clary of Wright Water Engineers.

Keehner, D., (2007), “Report of the Experts Scientific Workshop on Critical Research Needs for the Development of New or Revised Recreational Water Quality Criteria, Draft Executive Summary by Workgroup Chairs”, EPA,Offoce of Water and ORD, Rep. No. 823-R-07-007.

Kim, W.J., Managaki, S., Furumai, H. and Nakajima, F., (2008), “Simulation of the behavior of indicator microorganisms in combined sewer system”, 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

Korich, D.G., Mead, J.R., Madore, M.S., Sinclair, N.A. and Sterling, C.R., (1990), “Effects of Ozone, Chlorine Dixoide, Chlorine and Monochloramine on Cryptosporidium parvum Oocyst Viability”, Vol. 56, No. 5, p. 1423-1428.

Kutzner, R., Brombach, H. and Geiger, W.F., (2007), “Sewer solids separation by sedimentation – the problem of modeling, validation and transferability”, Water Science & Technology Vol. 55, No. 4, pp 113–123.

Lacour, C., Joannis, C., Gromair, M.C. and Chebbo, G., (2008), “Potential of continuous turbidity measurements for improving management of pollutant flows during wet weather”, 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

Le Coustumer, S., Fletcher, T., Deletic, A. and Barraud, S., (2008), “Influence of time and design on the hydraulic performance of biofiltration systems for stormwater management”, 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

Leung, H., Chen, G., Sharma, K., (2005), “Effect of detached/re-suspended solids from sewer sediment on the sewage phase bacterial activity”, Water Science & Technology, Vol. 52, No. 3, p. 147-152.

Lewis, J.F., Hatt, R.E., Deletic, A. and Fletcher, T.D., (2008), “The impact of vegetation on the hydraulic conductivity of stormwater biofiltration systems”, 11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008.

Lin, S.D. and Lee, C.C., (2007), “Water and Wastewater Calculations Manual”, McGraw-Hill Professional.

Liu, Y., He, Y., Yang, S. and Li, Y., (2006), “The settling characteristics and mean settling velocity of granular sludge in upflow anaerobic sludge blanket (UASB)-like reactors”, Biotechnolgy Letters, Vol. 28, Iss. 20, pp 1673-1678.

Luyckx, G., Vase, G., and Berlamont, J., (2005), “Solids separation efficiency of combined sewer overflows”, Water Science and Technology, Vol. 51, No. 2, pp. 71-78.

Maus, C. and Holtrup, (2008), “Measurement method for in situ particle settling velocity”, Proceedings of the 11th International Conference on Urban Drainage, Edinburgh, Scotland, 2008.

McKay, L.D., Harton, A.D., and Wilson, G.V., (2002), “Influence of Flow Rate on Transport of Bacteriophage in Shale Saprolite”, Journal of Environmental Quality, Vol. 31, pp. 1095-1105.

McLeod, M., Aislabie, J., Smith, J., Fraser, R., Roberts, A. and Taylor, M., (2001), “Viral and Chemical Tracer Movement through Contrasting Soils”, Journal of Environmental Quality, Vol. 30, pp 2134-2140.

Ministry of Agriculture and Forestry, (2006), “Pathogen Pathways: Best Management Practices”, New Zealand, Report No. 2006/01.

Molina, M., (2005), “Temporal and Spatial Variability of Fecal Indicator Bacteria: Implications for the Application of MST Methodologies to Differentiate Sources of Fecal Contamination”, EPA National Exposure Research Laboratory, Rep. No. EPA/600/R-05/107.

Moore, J.A., Smyth, J., Baker, S., Miner, J.R., (1988), “Evaluating Coliform Concentrations in Runoff from Various Animal Waste Management Systems”, Agricultural Experiment Stations, Oregon State University, Corvallis, OR.

Mueller, J.A. and DiToro, D.M., (1977), Combined-Sewer Overflow Characteristics from Treatment Plant Data. EPA-600/2-83-049 (NTIS PB 83-224 543), EPA ORD, Cincinnati, OH.

Muyima, NYO and Ngcakani, F., (1998), “Indicator bacteria and regrowth potential of the drinking water in Alice, Eastern Cape”, Water SA, Vol. 24, No. 1.

Narkis, N., Armon, R., Offer, R., Crshansky, F., and Friedland, E., (1994), “Effect of Suspended Solids on Wastewater Disinfection Efficiency by Chlorine Dioxide”, Water Research, Vol. 29, No. 1, pp 227-236.

O'Connor, T. P., and Field, R., (2001), “U.S. EPA CSO Capstone Report: Control System Optimization,” 5th International Conference Diffuse/Nonpoint Pollution and Watershed Management, Milwaukee, June 10-15, 2001.

O’Shea, M. L. and Field, R., (1991), “Detection and disinfection of pathogens in storm-generated flows”, Canadian Journal of Microbiology, Vol. 38, No. 4, April, 1992.

Otis, R., Kreissl, J., Frederick, R., Goo, R., Casey, P. and Tonning, B., (2002), “Onsite Wastewater Treatment Systems Manual”, EPA, Office of Water and ORD, EPA/625/R-00/008.

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”, Agricultural Water Management, Vol. 86, pp. 81-92.

Perdek, J.M., Arnone, R.D., Stinson, M.K and Tuccillo, M.E., (2003), “Managing Urban Watershed Pathogen Contamination” EPA ORD Cincinnati, OH, EPA/600/R-03/111.

Perdek, J.M., and Borst, M., (2000a), “Microbial Particle Association and Combined Sewer Overflow Disinfection.” Proceedings of Disinfection 2000: Disinfection of Wastes in the New Millennium, March, 2000. Water Environment Federation, CDROM.

Perdek, J.M., and Borst, M., (2000b), “Particle Association Effects on Microbial Indicator Concentrations for CSO Disinfection.” Proceedings of 2000 Joint Conference on Water Resources Engineering and Water Resources Planning and Management, July 2000, Minneapolis, MN. American Society of Civil Engineers, CD-ROM.

Pisano, W.C., Barsanti, J., Sorensen, H. and Joyce, J., (1998), “Sewer and Tank Sediment Flushing: Case Studies”, Report No. EPA-600/2-98/157, NTIS PB99-127839INZ. EPA, Cincinnati, OH.

Powelson, D.K. and Mills, A.L., (2001), “Transport of Escherichia coli in Sand Columns with Constant and Changing Water Contents”, Journal of Environmental Quality, Vol. 30, pp. 238-245.

Qi, Y, S. Gillow, Herson, D.S. and Dentel, S.K., (2004) “Reactivation and/or growth of Fecal Coliform Bacteria During Centrifugal Dewatering of Anaerobically Digested Biosolids”, Water Science and Technology, Vol. 50, No. 9, pp. 115–120.

Rideau River Stormwater Management Study, (1981a), “Rideau River Stormwater Management Study Final Report”, Regional Municipality of Ottawa-Carleton.

Rideau River Stormwater Management Study, (1981b), “Rideau River Stormwater Management Study Report on Catchment Monitoring 1981”, Regional Municipality of Ottawa-Carleton.

Rose, J.B., Nowlin, H., Farrah, S.R., Harwood, V.J., Levine, A.D, Lukasik, J. and Menendez, P., (2004), “Reduction of Pathogens, Indicator Bacteria, and Alternative Indicators by Wastewater Treatment and Reclamation Processes.”, Water Environment Research Foundation, Report No. 00-PUM-2T.

Roslev, P., Bastholm, S., Iversen, N., (2008), “Relationship Between Fecal Indicators in Sediment and Recreational Waters in a Danish Estuary”, Water, Air, & Soil Pollution, Vol. 194, Iss. 1, pp 13-21.

Roszak, D.B. and Colwell, R.R., (1987), “Survival Strategies of Bacteria in the Natural Environment”, Microbiological Reviews. Vol. 51, No. 3, pp. 365-379.

Rowney, A.C., (2009), "QUALHYMO Technical Reference Manual 2009", ACR, LLC.

Rowney, A.C., Field, R. and O’Connor, T.P., (2008), “Emerging Implications of Balancing Disinfection and Primary Treatment as an Element in CSO Control: Model Requirements”, Proceedings of the World Environmental and Water Resources Congress, EWRI/ASCE.

Rowney, A.C, and Roesner, L.A., (1982), “Continuous Simulation of In-stream Fecal Coliform Bacteria”, Proceedings of the Water Quality Manager's Conference, Alexandria, VA, EPA.

Sahr, R.C., (2004), “Historic Cash Values”, Political Science Department, Oregon State University, Corvallis, OR, (http://puna.net.nz/archives/Social/Historic%20Cash%20Values.pdf)

Sample, D.J., Heaney, J.P., Wright, L.T., Fan, C. Lai, F. and Field, R., (Pending), “Costs of Best Management Practices and Associated Land for Urban Storm Water Control”, Journal of Water Resources Planning and Management, Vol 129, No. 1, pp. 59-68.

Scheible, K, and Moffa, P., (1999), “Combined Sewer Overflow Technology Fact Sheet Alternative Disinfection Methods”, EPA Office of Water, EPA 832-F-99-033.

Schellart, A., Veldkamp, R., Klootwijk, M., Clemens, F., Tait, S., Ashey, R., Howes, C., (2005), “Detailed observation and measurement of sewer sediment erosion under aerobic and anaerobic conditions”, Water Science & Technology, Vol. 52, Iss. 3, pp. 137-146.

Schijven,J.F, Bradford, S.A. and Yang, S., (2004), “Release of Cryptosporidium and Giardia from Dairy Cattle Manure: Physical Factors”, Journal of Environmental Quality, EPA 832-F-99-033.

Searcy, K.E., Packman, A.I., Atwill, E.R. and Harter, T., (2005), “Association of Cryptosporidium parvum with Suspended Particles: Impact on Oocyst Sedimentation”, Applied Environmental Microbiology, Vol. 71, No. 2, pp. 1072-1078.

Seggelke, K., Fuchs, L., Tränckner J., Krebs, P., (2008), “Development of an integrated RTC system for full-scale implementation”, Proceedings of the 11th International Conference on Urban Drainage, Edinburgh, Scotland, 2008.

Shepherd, W, Saul A. and Boxall, J, (2008), “Quantifying the Performance of Storm Tanks”, Proceedings of the 11th International Conference on Urban Drainage, Edinburgh, Scotland, 2008.

Stevens, R., (2006), “Fact Sheet: A WERF Study Finds Fecal Coliforms Appear to Reactivate in Centrifuge Dewatered Solids at Four of Seven Facilities Tested”, EPA Office of Water.

Stoddard, A., Harcum, J.B., Simpson, J.T., Pagenkopf, J.R. and Bastian, R.K., (2002), “Municipal Wastewater Treatment: Evaluating Improvements in National Water Quality”, John Wiley and Sons.

Struck, S.D., Selvakumar, A. and Borst, M., (2006), “Performance of Stormwater Retention Ponds and Constructed Wetlands in Reducing Microbial Concentrations,” EPA ORD Cincinnati, OH, EPA/600/R-06/102.

Tafwik, A., Klapwijk, A., van Buuren, J.C.L., El-Gohary, F. and Lettinga.G., (2004), “Physico-chemical factors affecting the E.coli removal in a rotating biological contactor (RBC) treating UASB effluent”. Water Research, Vol. 38, No. 5, pp. 1081-1088.

Tchobanoglous, G., Burton, F.L. and Stensel, H.D., (2003), “Wastewater Engineering: Treatment and Reuse (4 Ed)”, McGraw-Hill.

Tholin A.L and Keifer C.J., (1959), “Hydrology of Urban Runoff”, paper # 3061ASCE Transactions.

Thompson, S.S., Flury, M., Yates, M.V. and Jury, W.A., (1998), “Role of the Air-Water-Solid Interface in Bacteriophage Sorption Experiments”, Applied Environmental Microbiology, Vol. 64, No. 1, 304-309.

VanWeele, B., (2009a), Personnal communication with Brian VanWeele, Parsons Brinckerhoff Quade and Douglas, in which Van Weele provided this content based on work carried out by himself and J. W. Filbert.

VanWeele, B., (2009b), Personnal communication with Brian VanWeele, Parsons Brinckerhoff Quade and Douglas, in which Van Weele provided this content based on work carried out by himself and J. O'Carroll of CH2M Hill.

Vail, J.H., Morgan, R., Merino, C.R., Gonzales, F., Miller, R., and Ram, J.L., (2005), Enumeration of Waterborne Escherichia coli with Petrifilm Plates: Comparison to Standard Methods”, Journal of Environmental Quality, Vol. 32, pp. 368-373.

Vesilind, P.A. and Rooke, R.L., Ed., (2004), “Wastewater Treatment Plant Design”, IWA Publishing, ISBN: 9781843390527.

Vogel, J.R., Stoeckel, D.M., Lamendella, R., Zelt, R.B., Santo Domingo, J.W., Walker, S.R. and Oerther, D.B., (2007), “Identifying Fecal Sources in a Selected Catchment Reach Using Multiple Source-Tracking Tools”, Journal of Environmental Quality, Vol. 36, pp. 718-729.

Walker, D., Golden, J., Bingham, D. and Driscoll, E., (1993), “Combined Sewer Overflow Control”, EPA ORD, Cincinnnati, OH, EPA-625/R-93-0007 (NTIS PB 93-144 649).

Wang, L.K., Y. Hung, and N. Shammas, Ed., (2005), “Physicochemical Treatment Processes”, Handbook of Environmental Engineering Volume 3, The Humana Press, Totowa, NJ ISBN: 1-58829-165-0.

Whitman, R.L., Nevers, M.B. and Byappanathalli, M.N., (2006), “Examination of the Watershed-Wide Distribution of Escherichia coli along Southern Lake Michigan: and Integrated Approach”, Applied and Environmental

Microbiology, Vol. 72, No. 11, p. 7301-7310.
Whitman, R.L., Nevers, M.B., Korinek, G.C., (2004), “Solar and Temporal Effects on Escherichia coli Concentration at a Lake Michigan Swimming Beach”, Applied and Environmental Microbiology, Vol. 70, p. 4276-4285.
Williams, Damon S. Associates, LLC., (2004), “Online Monitoring of Wastewater Effluent Chlorination using Oxidation Reduction Potential (ORP) vs. Residual Chlorine Measurement”, Water Environment Research Foundation, Vol. 72, No .11, pp 7301-7310.
Yang, Hsiao-Hui, Vinopal, R.T., Grasso, D. and Smets, B.F., (2004), “High Diversity among Environmental Escherichia coli Isolates from a Bovine Feedlot”, American Society of Microbiology, Vol. 70, No. 3, pp. 1528-1536.
Yolcubal, I., Pierce, S.A., Maier, R.M. and Brusseau, M.L., (2002), “Biodegradation during Contaminant Transport in Porous Media: V. The Influence of Growth and Cell Elution on Microbial Distribution”, Journal of Environmental Quality, Vol. 31, pp. 1824-1839.
Zhuang, J., and Jin, Y., (2003), “Virus Retention and Transport as Influenced by Different Forums of Soil Organic Matter”, Journal of Environmental Quality, Vol. 32, pp. 816-823.

Links as of the time of preparation of this report:

http://www.ecojustice.ca/publications/reports/the-green-infrastructure-report/attachment
http://www.microbiologybytes.com/video/motility.html.
http://www.ysi.com/extranet/EPGKL.nsf/447554deba0f52f2852569f500696b21/90a0378150c2d2dd85256a1f0073f295/$FILE/069300B.pdf

Software (other than office tools such as spreadsheets and word processors) used in the conduct of this work:

Simile version 5.4p2 Standard, Simulistics, (http://www.simulistics.com/)
QUALHYMO 2007, Rowney (http://qualhymo.watertoolset.com)

Appendix A Multiphase Sediment/Bacteria Model Development

Model Development

The modeling tool used to construct the multiphase sediment/bacteria model (MSBM) was a simulation environment known as Simile[13,14] The environment is designed to enable rapid building and application of models that includes the representation of unit processes of the type defined for this project.

Simile Definitions

The complete documentation for Smile is available in the documentation provided with the tool, and will not be described here. A few basic terms, however, are provided to aid readers not conversant with Simile in understanding some of the diagrams in this Appendix. The basic scope of elements represented in a Simile model includes the elements indicated in Figure 31.

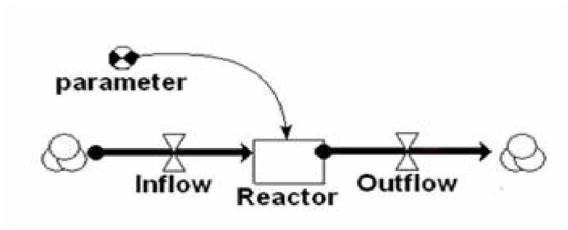

Figure 31 The principal Simile icons

- The major computational unit is the reactor, represented by a box.
- Inflows and outflows to the reactor are represented by bold arrows with clouds at the ends.
- Parameters are represented by a small circle with two blackened segments.
- A thin arrow connects a parameter to a reactor, an inflow or an outflow and the parameter value is thereby known to that element.

---

[13] This tool is available from Simulistics, Inc. (http://www.simulistics.com/)

[14] Although the authors find this to be a useful tool and a preferred option for this kind of simulation task, it is not recommended for particular use or otherwise generally endorsed by the authors. It was selected for this research because it was known to the authors to be convenient for the purposes of this project and to be technically competent to do the job at hand. The software used was purchased at standard commercial rates from the vendor under a standard license agreement, and no preferential purchase or support mechanisms were offered to or accepted by the authors.

These elements can be connected in logical ways with few restrictions and can therefore represent complex systems. Each element is capable of holding a value or equations, and the computational capabilities of Simile are such that the tool will resolve all equations and values according to the ways they are connected. The model is robust and dependable, and primarily designed for use in research. It facilitates research of this type in that it is able to represent phenomena and enable computations without requiring that the researcher write the masses of peripheral data entry and solution code necessary to do this. In the problem at hand, the reactor elements readily represent the unit processes being analyzed. The inflow and outflow elements can represent wastewater flows, sediments or bacteria passing through those reactors. Other information, such as rate constants and fall velocities, can be defined with the parameter elements. The specific formulation of the model used in this work is described below.

Model Formulation

The model was developed in two versions, referred to as MSBM-1 and MSBM-2. The overall structure of the first tool, MSBM-1, is shown in Figure 32. There are components that represent flow, suspended sediment, and disinfection processes, as discussed in the model conceptual development. Also represented are processes that represent bacteria carried through the model in solution or in association with particles. MSBM-1 terms are defined below.

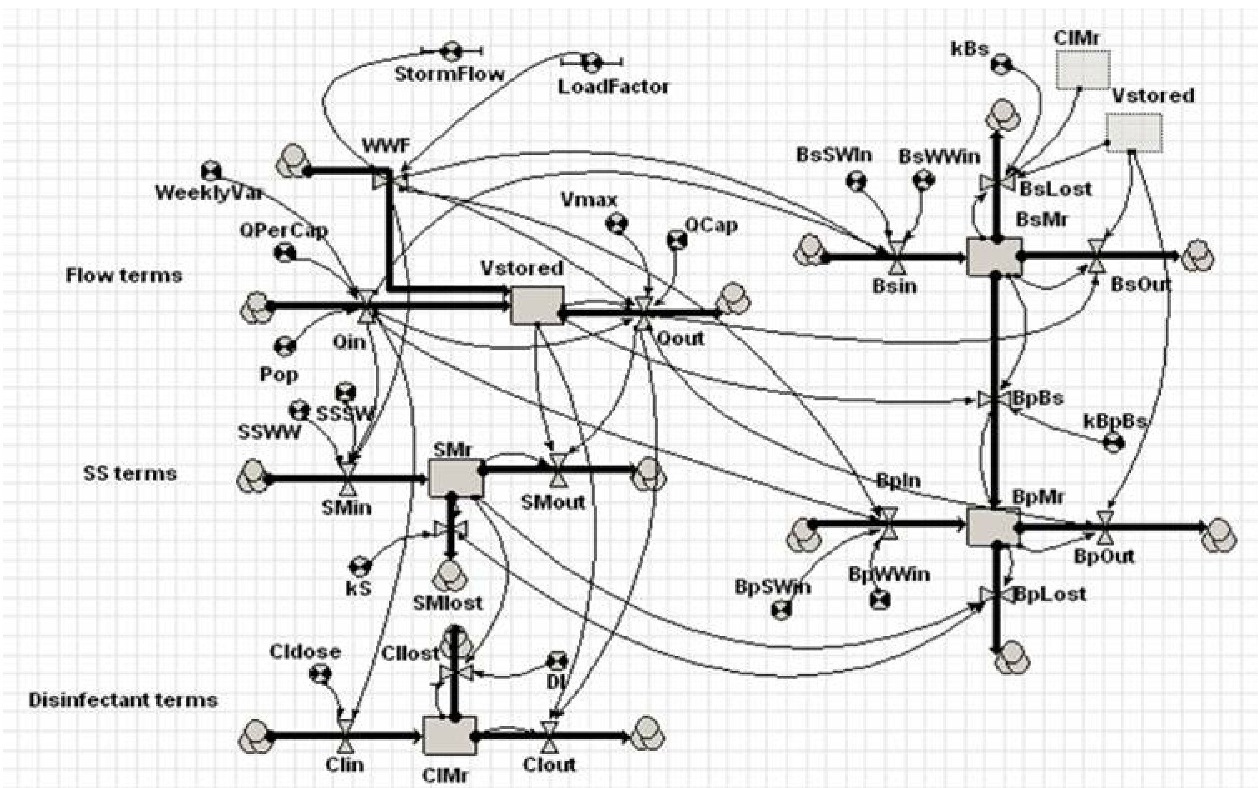

Figure 32 Multiphase Sediment/Bacteria Model - version 1 schematic

Flow Component

The flow component consists of a forcing function that includes provision for diurnal variation, and for weekly variation. The key terms are:

WeeklyVar -	A weekly variation of wastewater flow, centered on 1.0 and ranging between minimum and maximum weekly values, daily.
QPerCap -	Per capita unit flow, centered on 1.0 and ranging between minimum and maximum daily values, hourly.
Pop -	Total population generating wastewater.

Qin -	Total inflow rate = WeeklyVar*QPercap*Pop
StormFlow -	The rate of flow resulting from wet-weather conditions
LoadFactor -	A utility factor, used to reflect intervention that reduces the StormFlow contribution.
WWF -	Wet weather flow = StormFlow*LoadFactor
VStored -	Volume maintained in the reactor = VStored(0) + Σ (Qin+WWF) - Σ (Qout)
QCap -	Maximum outflow rate from reactor
Qout -	Total outflow rate = if Vstored<Vmax then Vstored/Vmax*QCap else Qin+WWF

Sediment Component

The sediment component has terms that reflect the addition and removal of suspended sediments from the system. It is noted that in this configuration, the reactor behaves as a well-mixed reactor in regards to outflow. Variations on this model use the product of settling velocity and area to calculate sediment removal. The key terms in this version are:

SSWW -	Suspended sediment concentration in wastewater
SSSW -	Suspended sediment concentration in stormwater
SMin -	Suspended sediment entering reactor = SSWW*Qin+WWF*SSSW
SMr -	Sediment mass in the reactor = SMr(0) + Σ (SMin) - Σ (SMout)
kS -	Sediment loss rate constant
SMlost	Sediment losses in the reactor = SMr*kS
SMout -	Suspended sediment leaving the reactor SMr/Vstored*Qout

Disinfectant Component

The disinfectant component reflects the available disinfectant in the fluid stream. The key terms in this component are:

Cldose -	Chlorine dose added to the reactor, expressed as a concentration
Clin -	Chlorine entering the reactor = Qin*Cldose
Dl -	Chlorine lost per unit mass of sediment in the reactor
Cllost -	Chlorine lost to sediment demand = Dl*ClMr*SMr
ClMr -	Chlorine mass in the reactor = ClMr(0) + Σ (Clin) - Σ (Clout) - Σ (Cllost)
Clout -	Chlorine leaving the reactor = ClMr/Vstored*Qout

Bacteria in Solution Component

This is one of the major outcomes of the calculations, accounting for bacteria that pass though the system in solution. The key terms in this component are:

BsSWin -	Bacteria concentration in stormwater entering the reactor
BsWWin -	Bacteria concentration in wastewater entering the reactor
Bsin -	Bacteria entering the reactor in solution = BsSWIn*WWF+BsWWin*Qin
kBs -	Bacteria lost rate constant for disinfection agent.
BsLost -	Bacteria in solution lost to disinfection = kBs*(ClMr/Vstored)*(BsMr/Vstored)
kBpBs -	Bacteria particulate/solute phase partition coefficient
BpBs -	Bacteria adsorption migration rate = Vstored*(BsMr-BpMr)*kBpBs
BsMr -	Bacteria in the reactor = BsMr(0) + Σ (Bslin) - Σ (BsOut) - Σ (BsLost) - Σ (BpBs)
BsOut -	Bacteria in solution leaving the reactor = Qout*BsMr/Vstored

Bacteria Bound to Particles Component

This is the other major result of the calculations, accounting for bacteria that pass though the system in association with particles. The key terms in this component are:

BpSWin -	Bacteria concentration in stormwater-born particulates entering the reactor
BpWWin -	Bacteria concentration in wastewater-born particulates entering the reactor
Bpin -	Bacteria entering the reactor in solution = BpSWin+WWF+Qin*BpWWin
BpLost -	Bacteria in particulates lost to sedimentation = BpMr/SMr*SMlost
BpMr -	Bacteria in the reactor = BpMr(0) + Σ (Bplin) - Σ (BpOut) - Σ (BpLost) + Σ (BpBs)
BpOut -	Bacteria in solution leaving the reactor = Qout*BpMr/Vstored

While the MSBM-1 version was workable, it suffered from a limiting data deficiency that was anticipated in development but became critical during testing. It had a component that explicitly represented bacterial treatment losses associated with variable chlorine does. This is mathematically tractable and attractive in that it enables a deeper quantitative interpretation of the relationship between the dose, sedimentation and bacterial output. Unfortunately, the available data for dose/bacteria response were not found to support a meaningful reaction of this type. This model was therefore dropped from further consideration, and is documented primarily to provide a possible step forward in model development if and when meaningful data of this type becomes available.

A second model, MSBM-2 was therefore developed. This version is focused on an examination of chlorine demand and input to the chlorination stream. To facilitate this, the volumes involved are explicitly separated into a primary settling unit and a disinfection unit to more readily enable evaluation of sizing alternatives. This model was the focus of the remainder of this effort, as shown in Figure 33. MSMB-2 terms are defined below.

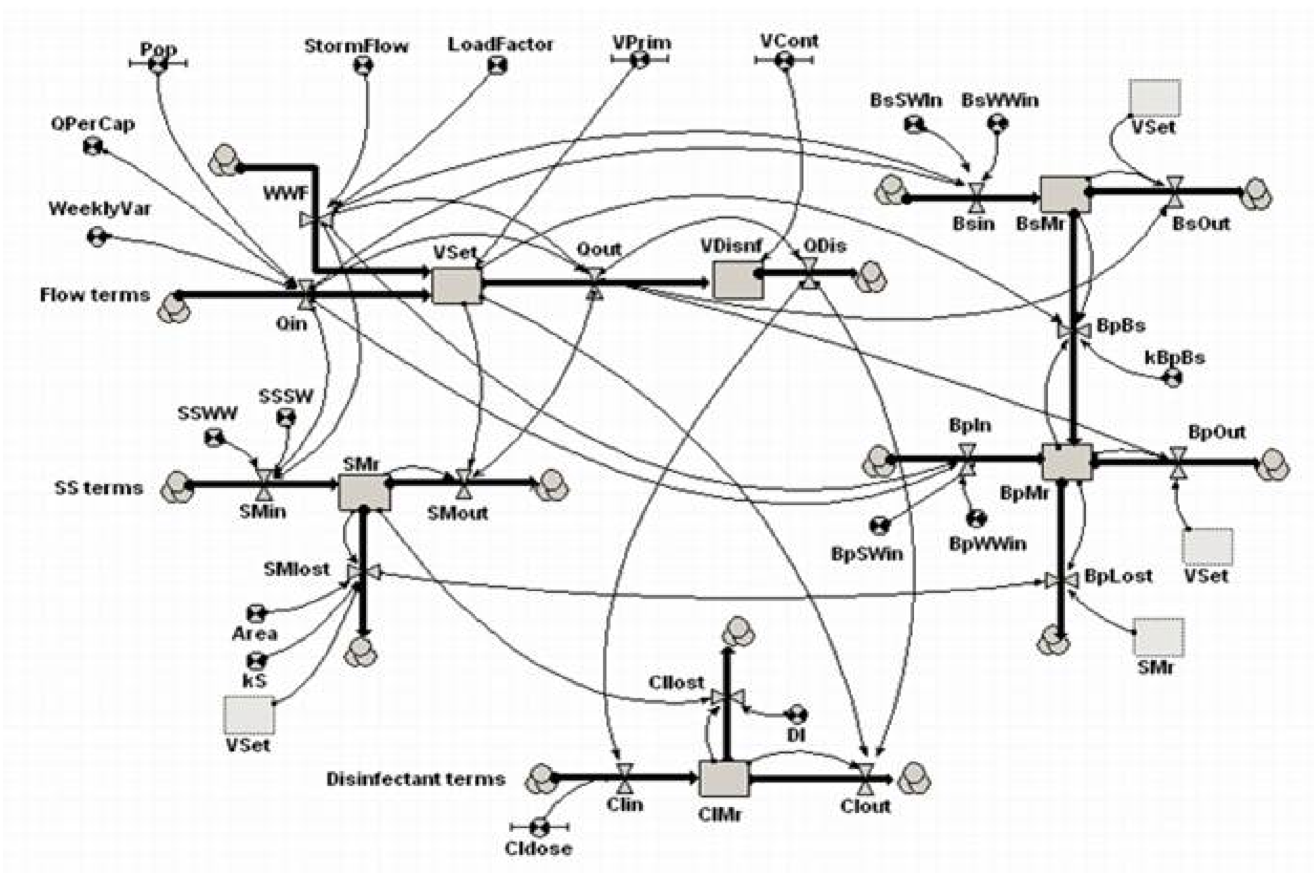

Figure 33 Multiphase Sediment/Bacteria Model - version 2 schematic

Flow Component

The flow component consists of a forcing function that includes provision for diurnal variation, and for weekly variation. MSBM-2 differs from MSBM-1 in that volumes are constant, and the disinfection reactor is explicitly separate from the primary settling reactor to facilitate model use. The key terms are:

WeeklyVar -	A weekly variation of wastewater flow, centered on 1.0 and ranging between minimum and maximum weekly values, daily
QPerCap -	Per capita unit flow, centered on 1.0 and ranging between minimum and maximum daily values, hourly
Pop -	Total population generating wastewater
Qin -	Total inflow rate = WeeklyVar*QPercap*Pop
StormFlow -	The rate of flow resulting from wet-weather conditions
LoadFactor -	A utility factor, used to reflect intervention that reduces the StormFlow contribution
WWF -	Wet weather flow = StormFlow*LoadFactor
VPrim -	Volume maintained in the primary settling reactor
VSet -	Volume maintained in the primary settling reactor = VPrim
VCont -	Volume maintained in the chlorination reactor
VDisinf -	Volume maintained in the chlorination reactor = VCont
Qout -	Total outflow rate = Qin + WWF
QDis -	Discharge flow rate = Qout

Sediment Component

This is similar to MSBM-1. The sediment component has terms that reflect the addition and removal of suspended sediments from the system. It is noted that in this configuration, the reactor behaves as a well-mixed reactor in regards to outflow. Variations on this model use the product of settling velocity and area to calculate sediment removal. The key terms in this version are:

SSWW -	Suspended sediment concentration in wastewater
SSSW -	Suspended sediment concentration in stormwater
SMin -	Suspended sediment entering reactor = SSWW*Qin+WWF*SSSW
SMr -	Sediment mass in the reactor = SMr(0) + Σ (SMin) - Σ (SMout)
kS -	Sediment loss rate constant
SMlost	Sediment losses in the reactor = SMr*kS
SMout -	Suspended sediment leaving the reactor SMr/Vstored*Qout

Disinfectant Component

This also is similar to MSBM-1. The disinfectant component reflects the available disinfectant in the fluid stream. The key terms in this component are:

Cldose -	Chlorine dose added to the reactor, expressed as a concentration
Clin -	Chlorine entering the reactor = Qin*Cldose
Dl -	Chlorine lost per unit mass of sediment in the reactor
Cllost -	Chlorine lost to sediment demand = Dl*ClMr*SMr
ClMr -	Chlorine mass in the reactor = ClMr(0) + Σ (Clin) - Σ (Clout) - Σ (Cllost)
Clout -	Chlorine leaving the reactor = ClMr/Vstored*Qout

Bacteria in Solution Component

This is an aspect where MSBM-2 differs from MSBM-1, although it remains a major outcome of the calculations, accounting for bacteria that pass though the system in solution. The key terms in this component are:

BsSWin -	Bacteria concentration in stormwater entering the reactor
BsWWin -	Bacteria concentration in wastewater entering the reactor
Bsin -	Bacteria entering the reactor in solution = BsSWIn*WWF+BsWWin*Qin
kBpBs -	Bacteria particulate/solute phase partition coefficient

BpBs - Bacteria adsorption migration rate = Vstored*(BsMr-BpMr)*kBpBs
BsMr - Bacteria in the reactor = BsMr(0) + Σ (Bslin) - Σ (BsOut) - Σ (BpBs)
BsOut - Bacteria in solution leaving the reactor = Qout*BsMr/Vstored

Bacteria Bound to Particles Component

This is essentially identical to MSBM-1, and is another underlying defining characteristic of these models, accounting for bacteria that pass though the system in association with particles. The key terms in this component are:

BpSWin - Bacteria concentration in stormwater-born particulates entering the reactor
BpWWin - Bacteria concentration in wastewater-born particulates entering the reactor
Bpin - Bacteria entering the reactor in solution = BpSWin+WWF+Qin*BpWWin
BpLost - Bacteria in particulates lost to sedimentation = BpMr/SMr*SMlost
BpMr - Bacteria in the reactor = BpMr(0) + Σ (Bplin) - Σ (BpOut) + Σ (BpBs)
BpOut - Bacteria in solution leaving the reactor = Qout*BpMr/Vstored