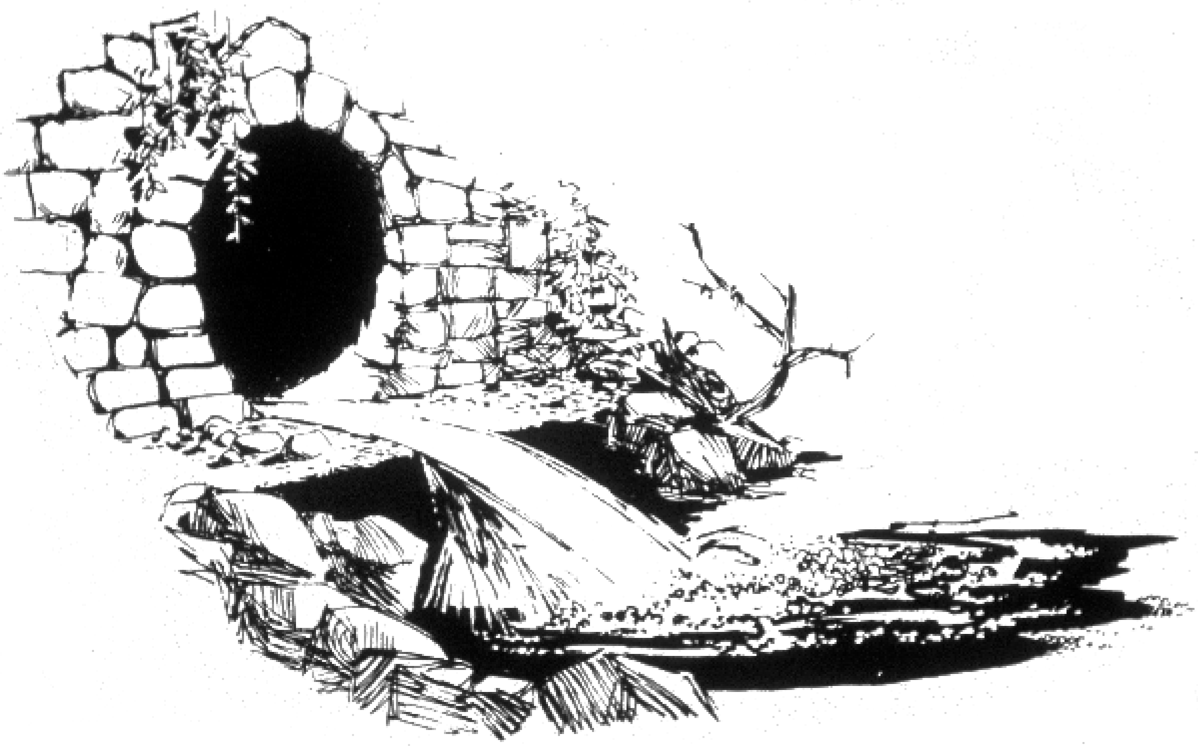
United States
Environmental Protection
Agency

Office of Research
and Development
Washington, DC 20460

EPA/600/R-06/033
July 2006

BMP Modeling Concepts and Simulation



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BMP Modeling Concepts and Simulation

by

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In support of:

EPA Contract No. 68-C-01-020
University of Colorado at Boulder

Project Officer

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Notice

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Sally Gutierrez, Director.
National Risk Management Research Laboratory

Abstract

In order to minimize impacts of urban nonpoint source pollution and associated costs of control (storage and treatment) associated with wet-weather flows (WWFs), stormwater runoff volumes and pollutant loads must be reduced. A number of control strategies and so-called "best management practices" (BMPs) are being used to mitigate runoff volumes and associated nonpoint source (diffuse) pollution due to WWFs and include ponds, bioretention facilities, infiltration trenches, grass swales, filter strips, dry wells, and cisterns. Another control option is popularly termed "low impact development" (LID) – or hydrologic source control – and strives to retain a site's pre-development hydrologic regime, reducing WWF and the associated nonpoint source pollution and treatment needs.

Methodologies are needed to evaluate these BMPs, their effectiveness in attenuating flow and pollutants, and for optimizing their cost/performance since most models only partially simulate BMP processes. Enhanced simulation capabilities will help planners derive the least-cost combination for effectively treating WWFs. There is currently a confusing array of options for analyzing hydrologic regimes and planning for LID. Integrating available BMP and LID processes into one model is highly desirable.

This work analyzes several current modeling methods to evaluate BMP performance with the intention of facilitating the integration of improved BMP modeling methods into the U.S. Environmental Protection Agency (EPA) Storm Water Management Model (SWMM). Several other models are examined as part of this study. Options for enhancement of SWMM's LID simulation capabilities are also presented. Two extensive case studies in Portland, Oregon help to clarify current SWMM capabilities and needs for enhancement. The effort documented in this report is linked to a parallel effort at the University of Colorado related to optimization strategies for WWF control.

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List of Acronyms and Abbreviations

AGNPS	Agricultural Nonpoint Source Model
AMC	antecedent moisture condition
APWA	American Public Works Association
ASCE	American Society of Civil Engineers
BES	Bureau of Environmental Services
BMP	best management practice
BOD	biochemical oxygen demand
BOD_u	ultimate carbonaceous oxygen demand
C	carbon
CalTrans	California Department of Transportation
CDF	cumulative distribution function
cfs	cubic feet per second
CBOD	carbonaceous biochemical oxygen demand
CFSTR	continuous-flow, stirred-tank reactor
COD	chemical oxygen demand
CRCCH	Cooperative Research Centre for Catchment Hydrology
CSO	combined sewer overflow
CU	University of Colorado
CV	coefficient of variation
DCIA	directly connected impervious area
DMSTA	Dynamic Model for Stormwater Treatment Areas
DO	dissolved oxygen
DOC	dissolved organic carbon
DON	dissolved organic nitrogen
DP	dissolved phosphorus
DTM	digital terrain model
EPA	U.S. Environmental Protection Agency
ED	extended detention
EMC	event mean concentration
EPM	effluent probability method
ER	efficiency ratio

ET	evapotranspiration
FPC	fundamental process category
FWS	free water surface
G-A	Green-Ampt
GIS	geographic information system
GUI	graphical user interface
HRT	hydraulic residence time
HSPF	Hydrologic Simulation Program – Fortran
LID	low-impact development
MS4	municipal separate storm sewer system
MSL	mean sea level
MUSIC	Model for Urban Stormwater Improvement Conceptualization
N	nitrogen
NCOB	nitrogen, carbon, DO, bacteria (in WETLAND model)
NO_3-N	nitrogen as nitrate
NH_3	ammonia
NOD	nitrogenous oxygen demand
NRCS	Natural Resources Conservation Service
NRMRL	National Risk Management Research Laboratory
NURP	Nationwide Urban Runoff Program
O&M	operation and maintenance
OSU	Oregon State University
P	phosphorus
P8	Program for Predicting Polluted Particle Passage through Pits, Ponds and Puddles
PCSWMM	Personal Computer Storm Water Management Model
PFR	plug flow reactor
POC	particulate organic carbon
PPCC	probability plot correlation coefficient
PREWET	Pollutant Removal Estimates for Wetlands
PVC	polyvinyl chloride
REMM	Riparian Ecosystem Management Model
RTC	real time control
RTD	residence time distribution
S-I	storage-indication
SCS	Soil Conservation Service
SLAMM	Source Loading and Management Model
SS	suspended solids
SSF	sub-surface flow
SOD	sediment oxygen demand
SSO	sanitary sewer overflow
S/T	storage/treatment
STA	stormwater treatment area
SW	surface water
SWMM	Storm Water Management Model
TDS	total dissolved solids
TIS	tanks in series
TKN	total Kjeldahl nitrogen
TMDL	total maximum daily load
TN	total nitrogen
TOC	total organic carbon
TP	total phosphorus

TSS	total suspended solids
USDA	United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
USTM	Universal Stormwater Treatment Model
UWRRC	Urban Water Resources Research Council
VAFSWM	Virginia Field Scale Wetland Model Program
WASP	Water Quality Analysis Simulation Program
WEF	Water Environment Federation
WETLAND	Wetland Water Balance and Nutrient Dynamics Model
WMM	Watershed Management Model
WWC	wet-weather control
WWF	wet-weather flow

Acknowledgments

The author acknowledges the assistance of EPA Contract No. 68-C-01-020 in support of this research and the assistance of Project Officer Dr. Fu-hsiung (Dennis) Lai. The opinions expressed herein do not necessarily reflect those of the EPA.

Ms. LaMarr Clannon (who changed her name upon matrimony), a graduate research assistant employed on this project, performed most of the Portland LID modeling work described in this report and initiated the comparative model evaluations. Another graduate research assistant, Mr. Matt Stouder, performed most of the Portland BMP modeling work and provided additional evaluation of alternative models. The Oregon State University team is grateful to Dr. James Heaney and his colleagues at the University of Colorado for their partnership in this EPA project. Chapter 2 was written in cooperation with this group. The report content was improved by the comments of two external reviewers.

The project team is indebted to Mr. Joe Hoffman of the Portland Bureau of Environmental Services (BES) for his considerable help in providing and explaining the study area data. Information on the subcatchments in Johnson Creek as well as subcatchment data for the Lexington Hills BMP area came from Mr. Tom Liptan and Mr. Tim Kurtz at the BES. Mr. Kurtz provided much additional help, and many thanks are due to him for his help in this regard.

During the course of this EPA study, the OSU investigators also participated in National Cooperative Highway Research Program Project 25-20(01) related to evaluation of BMPs for highway applications and in Water Environment Research Foundation Project 02-SW-1 related to more general evaluation of urban BMPs. Efforts on both of these projects contributed to an improved understanding of BMP evaluation, especially that it related to analysis of event mean concentration data.

Project Publications

Publications resulting wholly or in part from Task B project activities (documented in this report) are listed by authors below:

Cannon, L. (2002). *Urban BMPs and their Modeling Formulations*, Master of Science Project Report, Dept. of Civil, Construction, and Environmental Engineering, Oregon State University, Corvallis, August.

Huber, W.C. (2001). Wet-weather Treatment Process Simulation Using SWMM. *Proc. Third International Conference on Watershed Management*. National Taiwan University, Taipei, Taiwan, pp.253-264.

Huber, W.C. and Cannon, L. (2002). "Modeling Non-Directly Connected Impervious Areas in Dense Neighborhoods," In *Global Solutions for Urban Drainage*, Proc. Ninth International Conference on Urban Drainage, E.W. Strecker and W.C. Huber, eds., Portland, OR, American Society of Civil Engineers, Reston, VA, CD-ROM, September (2002b).

Huber, W.C., Lai, F., Clannon, L. and Stouder, M. (2004). "Modeling Concepts for BMP/LID Simulation," *Best Management Practices (BMP) Technology Symposium: Current and Future Directions*, World Water and Environmental Resources Congress, Salt Lake, City, UT, Environmental and Water Resources Institute, American Society of Civil Engineers, Reston, VA, June, 11 pp.

Stouder, M. (2003). *Simulation Methods for Wetland/Pond BMPs*, Master of Science Project Report, Dept. of Civil, Construction, and Environmental Engineering, Oregon State University, Corvallis, June.

1 INTRODUCTION

1.1 THE NEEDS

Pollution problems stemming from combined sewer overflows (CSOs), sanitary sewer overflows (SSOs), and stormwater discharges are extensive throughout the nation, with the Northeast, Midwest, and Far West being the principal areas of concentration. Nationwide, approximately 1,100 municipalities have combined sewers, 85% of which are in eleven states serving 43 million people; there are over 15,000 overflow points within these systems. SSOs occur in more than 1,000 municipalities, and stormwater discharges occur in as many as 1.2 million municipal, industrial, commercial, institutional and retail sources (EPA NRMRL 1996).

National cost estimates have been developed to control contamination from these three sources of wet weather flow (WWF). According to the most recent 1996 EPA Clean Water Needs Survey (http://www.epa.gov/owmitnet/mtb/cwns/1996rtc/toc.htm), projected costs for CSO pollution abatement totaled $44.7 billion, and stormwater contributes $7.4 billion out of total clean water needs estimate of $139.5 billion (Table 1-1). SSO costs are included in categories I, III and IV in the table and "EPA believes that the needs estimates in these categories related to SSOs underestimate the total costs associated with preventing SSOs" (http://www.epa.gov/owmitnet/mtb/cwns/1996rtc/toc.htm). Indeed the EPA Research Plan for wet-weather flows (EPA NRMRL 1996) estimates SSO costs in the "tens of billions." The document also cites an American Public Works Association (APWA) study indicating that costs of controlling stormwater pollution are much higher, at more than $400 billion for capital investment and capitalized costs of $540 billion for operation and maintenance (O&M) of stormwater control facilities, in order to meet water quality standards for stormwater discharges. These large capital and O&M costs pose severe financial difficulties to cities and municipalities throughout the nation.

During wet weather periods, urban sewer systems often become overloaded. This can be alleviated, for example, by providing additional storage in the system or by providing additional treatment, either downstream at the treatment facility or upstream within the watershed. Since WWF impacts and controls are complex and the costs of abatement alternatives are enormously large, there are tremendous opportunities for significant cost savings as a whole if an "optimal" cost-effective combination of storage and treatment alternatives can be objectively formulated. EPA is developing a multi-year research program aimed at devising tools that can be used to evaluate sewerage systems and determine the optimal combination of WWF control alternatives for the most cost-effective operation of the system.

A related need deals with EPA's total maximum daily load (TMDL) program, by which pollutant loads are to be reduced on a watershed basis, with a goal to meeting water quality standards in receiving water systems. Management of stormwater quality is usually performed through a combination of so-called "best management practices" (BMPs) and a form of hydrologic source control popularly known as "low-impact development" (LID). Reliable simulation of BMPs and LIDs is needed for the modeling efforts that usually are a part of TMDL development.

Table 1-1. Needs for publicly owned wastewater treatment facilities and other eligibilities, January 1996 dollars. (http://www.epa.gov/owmitnet/mtb/cwns/1996rtc/toc.htm)

Needs Category		**Total Needs, Billion $**
Title II Eligible Projects		
I	Secondary Treatment	26.5
II	Advanced Treatment	17.5
IIIA	Infiltration/Inflow Correction	3.3
IIIB	Sewer Replacement/Rehabilitation	7.0
IVA	New Collector Sewers	10.8
IVB	New Interceptor Sewers	10.8
V	Combined Sewer Overflows	44.7
VI	Stormwater*	7.4
	Total Categories I-VI	128.0
Other Eligible Projects (Sections 319 and 320)		
VIIA-C	Nonpoint Source (agriculture and silviculture only)*	9.4
VIID	Urban Runoff	1.0
VIIE-G	Ground Water, Estuaries, Wetlands	1.1
	Total Category VII	11.5
Grand Total		139.5

*Modeled needs only. Estimated Category VI needs documented by the States are $3.2 billion.
Estimated Category VIIA-C needs documented by the States are $0.5 billion.
Costs for operation and maintenance are not eligible for federal funding and therefore are not included.

Municipal separate storm sewer system (MS4) owners and operators need to identify effective BMPs for improving stormwater runoff water quality; owners and managers of other highly-impervious land (e.g., highways and industries) have similar needs. Evaluation of performance involves the use of computer models and tools, and empirical relationships describing a quantitative estimate of pollution removed by BMPs. This information will help planners derive an effective combination of control strategies for WWFs. Because of the current state of the practice, however, very few sound scientific data are available for making decisions about which structural and non-structural management practices function most effectively under what conditions, and, within a specific category of BMPs, to what degree design and environmental variables affect BMP efficiency.

1.2 STATEMENT OF TASK

Models for simulation of BMP and LID options must consider antecedent runoff conditions for both a single rainfall event and continuous rainfall and simulate the physical processes of rainfall, evapotranspiration, soil infiltration to groundwater, and overland sheet-flow processes from individual lots to streets, then from streets to sewers and eventually to their outfalls. The EPA Stormwater Management Model (SWMM) is a state-of-the-art urban runoff process model and may be used as the process model of choice (Huber and Dickinson 1988, Roesner et al. 1988). It uses well-known hydrologic and hydraulic concepts to simulate the urban watershed. Moreover, the software itself has undergone an evolutionary upgrade to a user-friendly, object-oriented version called SWMM5 (http://www.epa.gov/ednnrmrl/models/swmm/index.htm). This program, which includes a graphical user interface (GUI), was prepared within the EPA itself and provides the framework for enhancements necessary to better model WWF control alternatives, especially BMP and LID options. SWMM5 enhancement needs are addressed herein through a goal of better tools for quantifying the effectiveness of various WWF control alternatives. The effectiveness of each control alternative constitutes essential input to analysis of management alternatives.

This report is half of a larger project, “Optimization of Urban Sewer Systems during Wet-Weather Periods” (EPA Contract No. 68-C-01-020), to the University of Colorado (CU) and a subcontract to Oregon State University (OSU) dealing with the related issues of *optimization* and simulation of wet-weather controls. Simulation can provide performance estimates of controls such as BMPs and LIDs that can be coupled to analytical tools to optimize life-cycle costs of capital and O&M investments as well as overall performance, e.g., as measured by effluent characteristics, amount of runoff treated, etc. The issue of performance measures will be discussed in detail in subsequent chapters.

A companion report (Heaney and Lee 2005) deals with the optimization efforts. This report deals with the simulation component of the project. It is based in part upon unpublished progress reports during the course of the project as well as OSU master’s project reports by Cannon (2002) and Stouder (2003).

1.3 FUNDAMENTAL PROCESS CATEGORIES AND URBAN BMPS

A fundamental process is essentially the same as a unit process in environmental engineering (Metcalf and Eddy 2003). There is no single universal list of fundamental process categories (FPCs); Minton (2002) provides a useful taxonomy, oriented toward stormwater treatment. Common structural BMPs are grouped by nine FPCs in Table 1-2, but alternatives are possible, and one BMP may fall under different FPCs.

Similarly, there are many types of BMPs. Modelers interested in describing BMP effectiveness may wish to model BMP types (typical swale or pond), or instead model the fundamental processes that occur in a BMP (sedimentation, infiltration, etc.). For purposes of this investigation, representative structural stormwater BMPs are listed in Table 1-3, adapted from a taxonomy prepared by the Minnesota Pollution Control Agency (http://www.pca.state.mn.us/water/pubs/sw-bmpmanual.html). This list is one of many from similar sources that could be used.

If modeling BMP by type, a method for determining performance parameters must be described. This may be done by investigating BMP effectiveness as reported, for instance, in the EPA/ASCE BMP Database (http://www.bmpdatabase.org/) and applying similar effectiveness measures to the model. This leads to many questions about measuring and reporting BMP performance. Modeling BMPs by simulation of FPCs requires extensive data on stormwater treatability as well as site and design descriptions, and raises issues about data availability. As will be described in this report, most BMPs are modeled by a heuristic combination of FPC simulation and empirical performance measures.

Table 1-2. Structural BMPs categorized by fundamental unit processes. Adapted from: BMP manual http://www.pca.state.mn.us/water/pubs/swm

Process	**Definition**	**BMP**
Sedimentation	Gravitational settling of suspended particles from the water column. It can be a major mechanism of pollutant removal in BMPs because pollutants often adsorb to particulate matter, especially clay or organic soils. Detention systems intercept runoff for gradual release later. Most are designed to empty between events and treat water quantity rather than quality. These systems can provide limited settling, which can often be resuspended with a subsequent event.	Dry pond, wet pond, other basin, small storage devices, wetland, underground pipes, vaults, tanks
Flotation	Separation of particulates with a specific gravity less than water. Trash, Styrofoam, oil and hydrocarbons can be removed from BMPs designed with an area for these to accumulate.	Oil-water separators, density separators, dissolved-air flotation
Filtration	Filtration devices remove particulates by passing water through a porous medium like sand, gravel, soil, peat, compost, or combinations thereof.	Trash racks, bar racks, screens, sand filters, compost filters, vegetation and soil, may be part of filtration systems. Modular or drop in filter systems
Infiltration	Infiltration systems capture runoff and provide a means of infiltration into the ground. Infiltration is the most effective means of controlling storm water runoff because it reduces the volume discharged to receiving waters. Filtration by soil removes TSS and associated pollutants, and dissolved nutrients are removed by adsorption.	Infiltration basins, porous pavement, infiltration trenches or wells, ponds, constructed wetlands
Adsorption	Contaminants are bound (to clay particles or macrophytic vegetation). Adsorption is not a common mechanism used in stormwater BMPs, although it can occur in infiltration systems with clayey soils, in organic filters, or in wetland systems.	Infiltration systems with clay soils, adsorption to macrophytes in constructed wetlands, compost filters

Table 1-2 (concluded)

Biological Uptake and Conversion	Biological uptake is an important nutrient control mechanism in BMPs treating urban runoff that typically contains high levels of nutrients. This occurs as aquatic plants and microorganisms utilize nutrients for growth. Maintenance and harvest of these plants is essential, otherwise as plants die and decay, they re-release nutrients. Biological conversion happens when microorganisms and bacteria break down organic contaminants into less harmful compounds.	Pond, bioswale, wetland
Chemical Treatment	Chemicals such as alum are added to promote flocculation and settling. Chlorine disinfection is sometimes used to treat combined sewer overflows.	Precipitation, flocculation, disinfection
Degradation (volatilization, hydrolysis, photolysis)	Degradation happens in open pool BMPs where contaminants volatilize, hydrolyze or photolyze.	Pond, wetland, open pool BMPs
Hydrodynamic Separation	Varies by device.	Swirl concentrators, secondary current devices, oil-water separators
Combination (Retention)	Retention systems hold and treat runoff until displaced by another volume of water. These can be very effective in treating both quality and quantity of runoff. Several processes can occur, such as sedimentation, filtration, infiltration, biological uptake, conversion and degradation.	Wetlands, bioswales

Table 1-3. Representative structural stormwater BMPs.*

BMP	Definition
Bioretention Facilities (constructed wetlands, wetland basins and wetland channels)	Similar to other facilities (pond, basin or channel) with more than 50% of its surface or bottom covered by emergent wetland vegetation. A wetland channel is a channel designed to flow very slowly, probably less than 2 ft/s at the 2-year flood peak flow rate. It has, or is designed to develop, dense wetland vegetation on its bottom.
Dry Wells	Dry wells are drilled often through impervious layers to reach lower pervious layers, filled with porous media designed to percolate surface water to groundwater.
Filter Strips	Grass filter strips, sometimes called bio-filters, bioswales, or buffer strips, are vegetated areas designed to accept sheet flow provided by flow spreaders, which accept flow from an upstream development. Vegetation may take the form of grasses, meadows, forests, etc. The primary mechanisms for pollutant removal are filtration, infiltration, and settling.
Grass Swales	A swale, sometimes also called a bio-filter or bioswale, is a shallow grass-lined channel with little bottom width, designed for shallow flow near the source of storm runoff.
Ponds	Retention ponds are also commonly known as “wet ponds” because they have a permanent pool of water, unlike detention basins, which dry out between storms. The permanent pool of water is replaced in part or in total by stormwater during a storm event. The design is such that any runoff captured during a storm event is released over time. The hydraulic residence time (HRT) for the permanent pool over time can provide biological treatment. A dry-weather base flow, pond liner and/or high groundwater table are required to maintain the permanent pool.
Cisterns	Rain barrels or cisterns act as storage devices and retain a portion of runoff for later use, such as irrigation or use as grey water.
Infiltration Trenches	Percolation or infiltration trenches can generally be described as ditches filled with porous media designed to encourage rapid percolation of runoff to the groundwater. An infiltration basin can capture a given stormwater runoff volume and infiltrate it into the ground, transferring this volume from surface flow to groundwater flow.
Extended Detention (ED)	ED dry basins are designed to completely empty at some time after stormwater runoff ends. These are adaptations of the detention basins used for flood control. The primary difference is in outlet design; the extended detention basin uses a much smaller outlet that extends the detention time for more frequent events in order to facilitate pollutant removal. The term “dry” implies that there is no significant permanent water pool between storm runoff events. Multiple uses (e.g., recreation) are possible for land occupied by ED facilities.

Table 1-3. (concluded)

Porous Pavement	Modular-block porous pavement is composed of perforated concrete slab units underlain with gravel. The surface perforations are filled with coarse sand or sandy turf. It is used in low traffic areas to accommodate vehicles while facilitating stormwater runoff at the source. The units should be placed in a concrete grid which restricts horizontal movement of infiltrated water through the underlying gravels.
	Poured-in-place porous concrete or asphalt is generally placed over a substantial layer of granular base. The pavement is similar to conventional materials, except for the elimination of sand and fines from the mix. If infiltration to ground water is not desired, a liner may be used below the porous media along with a perforated pipe and a flow regulator to slowly drain the water stored in the media over a 6 to 12 hour period.
Hydrodynamic Device	These devices are BMPs such as oil-water separators, sand interceptors, swirl-type concentrators, sedimentation vaults, and other prefabricated and package-type treatment devices.

*BMP definitions are from the ASCE BMP database: website, www.bmpdatabase.org

1.4 CURRENT BMP APPROACHES

Traditional stormwater controls have strongly emphasized large spatial scales for flood and water-quality analyses. Runoff volume is usually the most important hydrologic parameter in water quality studies, while peak flow rate and time of concentration are usually the most important hydrologic parameters for flooding and drainage studies. The relationships between these different hydrologic parameters and storm parameters are significantly different for different classes of rains. Runoff models for water quality investigations should therefore have additional capabilities beyond those of runoff models for flooding and drainage investigations.

Common, small storms (also termed "micro-storms" in the literature) are responsible for most of the annual urban runoff discharge quantities throughout North America (Heaney et al. 1977, EPA 1983, Pitt 1987, WEF and ASCE 1998, Wright et al. 2000). However, some existing urban runoff models originate from drainage and flooding evaluation procedures that emphasize very large design storms (several inches in depth). These large storms only contribute small portions of the annual average discharges. Several authors have suggested that stormwater quality can be managed by treating the flows associated with small storms (Heaney et al. 1977, EPA 1983, WEF and ASCE 1998, Wright et al. 2000, Pitt and Voorhees 2000, Sample et al. 2001). These storms occur many times a year and are responsible for the majority of the pollutant discharges. Runoff from about 70 to 80% comprises these frequent discharges of the annual precipitation onto urban areas, the effects of which are mostly chronic in nature (such as contaminated sediment and frequent high flow rates). Pitt and Voorhees (2000) use the breakdown bulleted below for various storms. It should be noted that relationships between depth and cumulative percentages vary regionally (Heaney et al. 1977, WEF and ASCE 1998).

- Frequent storms having relatively low pollutant discharges are associated with depths of less than 0.5 in. (12 mm). These are key events in which runoff-associated water quality violations, such as for bacteria, are of concern. In most areas, runoff from these rain storms should be totally captured and either re-used for on-site beneficial uses or infiltrated in upland areas. For most areas, the runoff from these rains is relatively easy to remove from the surface drainage system.

- Rains between 0.5 and 1.5 in. (12 and 38 mm) are responsible for about 75% of the nonpoint source pollutant discharges and are key rains in terms of addressing mass pollutant discharges. The small rains in this category can also be removed from the drainage system and the runoff re-used on site for beneficial means or infiltrated to replenish the lost groundwater infiltration associated with urbanization. The runoff from the larger rains should be treated to prevent pollutant discharges from entering the receiving waters.

- Rains greater than 1.5 in. (38 mm) are associated with drainage design and are only responsible for relatively small portions of the annual pollutant discharges. Typical storm drainage design events fall in the upper portion of this category. Extensive pollution control designed for these events would be very costly, especially considering the relatively small portion of the annual runoff associated with the events. However, discharge rate reductions are important to reduce habitat problems in the receiving waters. The infiltration and other treatment controls used to handle the smaller storms in the above categories would have some benefit in reducing pollutant discharges during these larger, rarer storms.

- In addition, extremely large rains also occur infrequently that exceed the capacity of the drainage system and cause local flooding. Two of these extreme events were monitored in Milwaukee during the Nationwide Urban Runoff Program, NURP (EPA 1983). Such storms, while very destructive, are sufficiently rare that the resulting environmental problems do not justify the massive stormwater quality controls that would be necessary for their reduction. The problem during these events is massive property damage and possible loss of life. These rains typically greatly exceed the capacities of the storm drainage systems, causing extensive flooding.

This interest in smaller storms requires modeling approaches that can properly account for losses (hydrologic abstractions such as infiltration, depression storage, and evapotranspiration) for low rainfall depths. SWMM is one such model with this capability. However, as noted elsewhere in this report, even though SWMM provides the option for vadose zone simulation, interaction with subsurface hydrologic processes could be improved with respect to the overall water balance. Alternatively, better opportunities to interface with subsurface models could be provided.

The micro-scale (spatially) is defined as a fundamental hydrologic unit, that is, a sub-parcel of an individual tract of land. A sub-parcel could be a roof, sidewalk, grass lawn, driveway, garden, or other landscape area. LID, also known as hydrological source control, is based on reducing hydrologic impacts and incorporating micro-scale BMPs throughout the subcatchment (Prince George’s County 2000). Reducing runoff by incorporating storage and infiltration onsite improves the quality and reduces the quantity of runoff produced by a developed site.

The LID philosophy is that runoff controls based on the micro-scale may be an important component of a stormwater management plan, and stormwater control at the micro-scale may effectively mitigate the effect of urban areas within the long-term hydrologic cycle. While not a panacea (Strecker 2001), this volume-based approach to management of runoff has great potential to reduce the runoff volume, sediment loads, and floatables that can reach receiving waters. By reducing runoff (and hence the associated pollutants) municipalities stand to substantially reduce the estimated billions of dollars associated with pollution abatement from WWF.

One question is whether the existing stormwater models designed for drainage and flood control can be adapted for evaluating these small-storm events at the sub-parcel scale. Flood design is usually based on a single design event. However, there are no accepted criteria for selecting a design small-storm. Rather, the appropriate strategy should incorporate continuous simulation over multiple years so that the integrated behavior of the system can be evaluated over all of the storms that can be the dominant source of water onto pervious areas. SWMM is one such model with this capability; others are discussed in Chapter 5.

1.5 SUMMARY OF CURRENT SWMM BMP SIMULATION CAPABILITIES

The heart of the research described herein has been to document SWMM capabilities needs with regard to BMP and LID simulation. The current SWMM (version 4.4h) as well as previous versions are each divided into four primary computational "blocks" or "modules": Runoff (converting rainfall to runoff and generating nonpoint source runoff quality), Transport (kinematic wave flow routing and water quality routing through conveyances and storage), Extran (dynamic wave flow routing), and Storage/Treatment (S/T) (treatment and storage devices). The object-oriented SWMM5 will not refer to "blocks" but rather to hydrologic, hydraulic, and other descriptive "objects" such as watersheds, channels, storages, land use, etc. To set the stage for the report that follows, capabilities and limitations of the current (SWMM4.4h and SWMM5) model have been identified and are summarized in Table 1-4 (Huber 2001b). Reference is made in Table 1-4 to the four SWMM blocks just described.

The implications of Table 1-4 are:

1. Storage is well simulated in any of the four current SWMM blocks, although the Runoff Block offers the least flexibility.
2. The S/T Block offers the most flexibility in terms of mimicking conventional and high-rate treatment devices, e.g., for combined sewers.
3. Removal fractions (applied to incoming loads) may be used for Runoff Block overland flow segments and Transport Block channel/pipes.
4. First-order decay may be applied in the Runoff, Transport, and S/T Blocks.
5. Settling velocities may be used in the Transport Block to simulate sedimentation (but no related effects, such as build-up of solids or resuspension).
6. Particle size ranges (or settling velocity ranges) can be tracked through the current Runoff and Transport Blocks as separate constituents. However, there is no linkage with the S/T Block, which provides the most sophisticated sedimentation routines, including application of Camp's (1946) sedimentation theory to up to five settling velocity ranges.
7. Overland flow rerouting options inherent in LID can be simulated in the Runoff Block, although one subcatchment per surface type might be necessary. SWMM can be applied at the parcel (individual lot) level on the basis of the modeling shown in this report.
8. Infiltration from channels is not simulated, except artificially, e.g., by an imposed "negative hydrograph" or through monthly evaporation.
9. Biological interactions (e.g., in bioswales, wetlands) may be simulated only through first-order decay (Runoff, Transport, S/T) and/or removal equations in the S/T Block.
10. There are few mechanistic, fundamental treatment processes included in the SWMM model, except for sedimentation in the S/T Block.

Table 1-4. Wet-weather controls and SWMM suitability (after Huber 2001b).

WWC Option	SWMM Suitability
Storage and Associated Treatment, e.g., Ponds	Simulation of most storage options, with several options for hydraulic controls. Treatment by removal equations, first-order decay, or sedimentation. Removal in Transport Block channel/pipes by first-order decay or sedimentation (constant settling velocity) or removal fraction applied to incoming loads.
Screening and Filtration	Simulation by removal equations.
Chemical Treatment, Chlorination	Simulation by removal equations.
Wetlands, Bioretention	Simulation to the extent that wetland or biological channel behaves like a storage device.
Source Controls	Simulated by reduced loadings.
Street Cleaning	Simulated directly through removal fractions or by reduced loadings.
Cisterns	Can divert water to storage but cannot arbitrarily retrieve it.
Dry Wells	Can divert water to well, but it then goes "out of simulation" unless tracked using groundwater options.
Overland Flow, Swales, Infiltration, Porous Pavement, Filter Strips	Simulation in Runoff Block only. Optional quality removal by first-order decay or removal fraction applied to incoming load. Infiltrating water carries pollutants with it.
Infiltration Trench	Can simulate infiltration from overland flow "trench" in Runoff.
Maintenance, e.g., Sewer Flushing	Simulation by reduced loading, with simplistic option for sediment scour/deposition in Transport Block.
Illicit Connection Removal	Simulation through modification of drainage system connectivity.
Inlet Constrictions	Hydraulic control, simulated in Extran Block.
Real-Time Control	Orifice and weir settings in Extran can be controlled as function of time and/or head changes.

Within this report, modeling concepts and, where appropriate, mathematical formulations are identified for viable BMP/LID alternatives that have been identified as missing from SWMM (Table 1-4). The effort to prepare this information is based on the assumption that such formulations are, in fact, available, which has not always been born out. Generally, BMP alternatives include (Table 1-3): storage (ponds, dry detention), bioretention facilities, infiltration trenches, grassed swales, filter strips, dry wells, cisterns, and hydrodynamic devices. Some theoretical and/or heuristic processes/equations that are based on state-of-the-engineering knowledge and information are proposed for possible inclusion in SWMM – probably in version 5 (SWMM5). The use of some BMPs to attenuate stormwater in upland locations as LID options, including rain gardens and roof-top green areas, is also gaining much attention. Necessary modeling parameters, e.g., soil characteristics and antecedent moisture content, which reflect local site-specific conditions, are identified in the discussion and in the examples of Chapters 14 and 15. SWMM modeling limitations for various BMP/LID alternatives are summarized in Chapter 16. The areas that have been addressed include:

- Spatial resolution to allow application of micro-management of flows in a residential lot (Section 4.9).
- Overland flow movement in pervious and impervious surfaces within a residential lot, and from lots to street gutters, swales, buffer strips, channels, and sewers (Sections 4.7 and 4.8).

- Subsurface flow movement of rainwater infiltration through unsaturated (aeration) zone and interaction of surface and ground waters (Chapters 8 – 11).
- Routing and attenuation of pollutants in overland and subsurface flow movement considering subsurface adsorption, absorption, and dispersion processes (needs identified in Chapter 16).
- Routing of flows and pollutants from lots to swales/street gutters/inlets/sewers (Sections 4.7 and 4.8).
- Hydraulic efficiency of storage-routing for pollutant removal (Sections 6.5 and 7.8).
- Runoff/storage/infiltration/treatment BMP/LID process designs (Chapters 4, 6-8).

Generally, the first part of this report deals with BMPs and their simulation options, followed by Chapters 14 and 15, which are devoted to two detailed case studies. *Note: generic reference to "SWMM capabilities" will refer to version 4.4h (April 2002) since SWMM5, which was officially released in October 2004, will continue to be enhanced in the future.*

2 STUDY AREA OPTIONS

2.1 OBJECTIVES

BMP and LID simulation may be more easily grasped in the context of examples. A component of this project is to provide locations for which case studies may be possible. Case studies with enough intra-event data for BMP simulation are just emerging (July 2003) from examination of the EPA/ASCE BMP Database (http://www.bmpdatabase.org/), and no attempt has been made to simulate only "real" BMPs for this reason. However, regarding LID, are there hypothetical residential *lots* (either conceptual or an actual residential lot) for which alternative BMP/LID stormwater management techniques could be demonstrated? The SWMM model could then be applied to these hypothetical lots to demonstrate how alternative scenarios would affect the model's WWF control simulation capabilities. Case studies based on these locations should include typical urban land uses and include a mix of combined and separate sewer systems. The locations should have detailed data on soils, land use, precipitation, infrastructure components, etc., and ideally should have been modeled (perhaps by the agency supplying the data) using SWMM or a similar program. Cost and performance data are also important – if unlikely to be available.

Five locations were identified during the course of this study that might be suitable for LID evaluation and/or simulation. They are summarized briefly below.

2.2 PORTLAND, OREGON

Portland, Oregon is a city containing many CSO areas with serious problems in terms of flooded basements and overflows to receiving waters. The Portland Bureau of Environmental Services (BES) is using a wide variety of BMPs and has been a leader in applying simulation models to their problems. Their CSO control program is embedded in a watershed plan for the entire area. Good cost data and excellent internet links for accessing data are available (http://www.cleanrivers-pdx.org/). Portland can be studied at micro and macro scales. Additional information is deferred to the extensive case studies presented in Chapters 14 and 15.

2.3 VALLEJO, CALIFORNIA

Vallejo, California is an SSO area with serious basement flooding and overflow problems. An excellent database on flows, sewer infrastructure, failure rates, control costs, and SWMM modeling results is available. A good general description of the Vallejo Sanitation and Flood Control District SSO elimination program can be found on their consultant's web site, http://www.carollo.com/vsfcd/. Technical information and data may be found in Carollo Engineers and CH2M Hill (2000). Results of risk optimization for Vallejo can be found in Wright et al. (2001a, b).

2.4 HAPPY ACRES

A hypothetical 105-ac study area called Happy Acres has been used extensively for research and teaching at the University of Colorado. This study area is comprised of low and medium density residential land use, a shopping center, and a school. It has been used as a single study area to evaluate water supply, wastewater, and stormwater options. Optimized designs have been done for the water, wastewater, and stormwater infrastructure. ArcView files have been generated to summarize the geographic information system (GIS) information. An Access database has also been developed for each parcel, and a facilities database is being developed. Results of analyses have been published in previous reports and papers dealing with costs (Heaney et al. 1999a, Fan et al. 2000, Sample et al. 2001), GIS (Heaney et al. 1999b, Sample et al. 2001), and optimization (Heaney et al. 1999c,d; Heaney et al. 1999e). A good benchmark of alternative designs is available for Happy Acres based on its extensive use in both research and teaching.

2.5 WONDERLAND CREEK IN BOULDER, COLORADO

Wonderland Creek in Boulder is a 14-ac residential neighborhood that has been evaluated in great detail regarding the nature of the imperviousness and its effect on runoff. Various levels of spatial detail may be obtained from the GIS coverage for this study area. SWMM has been set up and run for this neighborhood using 1-, 15-, and 60-minute rainfall data for Boulder (Lee 2003). The results of an accurate determination of the nature of imperviousness in urban areas indicate that existing estimates can be very inaccurate. The error in estimating imperviousness causes major changes in the accuracy of the predicted hydrographs (Lee 2003).

2.6 FAIR OAKS ESTATES IN CAROL STREAM, ILLINOIS

The Fair Oaks storm sewer and detention pond is used as a design example in the second edition of the *Hydrology Handbook* (ASCE 1996). Fair Oaks Estate is a subdivision of Carol Stream, Illinois, a suburb west of Chicago. The total area of the subdivision, which includes 13 lots, is 13.4 ac. Details of the detention pond design are shown in Burke (1979), Burke and Gray (1979), Burke and Burke (1994), and ASCE (1996). The design was done in the 1970s and the system has been in operation since that time. It is a conventional design and appears to be working fine after 20 years of operation. In his original MS thesis, Burke (1979) evaluated the impact of using the Rational Method (Dooge 1973) vs. the ILLUDAS (Terstriep and Stall 1974) model to solve the Fair Oaks problems. The use of ILLUDAS showed the hydraulic inadequacy of some of the pipes selected based on the Rational Method. Similar comparisons were done for other hydrologic and hydraulic models.

2.7 SUMMARY

Two locations in Portland were selected for SWMM simulations in this study for several reasons:

- Proximity to OSU: the project team performed most of the SWMM simulations.
- Existence of good flow monitoring data, rainfall data, and catchment characterization data for multiple sites.
- Willingness on the part of the Portland BES to share data and simulation results.
- Lack of some key information from the other sites, plus the fact that Happy Acres is hypothetical.

Using the two Portland locations, SWMM simulation examples are presented in Chapter 14 for LID and Chapter 15 for a pond.

3 BMP EVALUATION OPTIONS

3.1 BMP PERFORMANCE EVALUATION

If a model such as SWMM is used to simulate BMP performance, on what basis will the BMP's effectiveness be measured? A simulation model has the capability to produce simulated influent and effluent concentrations for thousands of storm events. Statistical methods used to evaluate monitored BMPs might also be used to evaluate simulated BMPs. If so, what conclusions can be drawn from evaluation efforts to date of monitored BMPs? This chapter discusses options for BMP evaluation based on measured concentrations and flows. Some of these options could also be applied to simulation model output.

A fundamental statistic of a quality monitoring program is the event mean concentration, or EMC. Although it is defined as the total constituent mass for an event divided by the total flow volume for the event, it is usually computed by preparing one flow-weighted composite sample from the several quality samples taken during a storm, and sending the one sample to the laboratory for analysis. The BMP "performance" is then usually – but not always – computed on the basis of the relationship between the influent and effluent EMCs, hopefully for many monitored storm events. In essence, the BMP is considered to be a "black box," to which several statistical and mathematical procedures might be applied to deduce the "transfer function" (Chapra 1997) that relates input to output. The most widely used statistical methods will be listed in the next section.

The issues involved in selecting methods for quantifying BMP efficiency, performance, and effectiveness are complex. It is also important to appreciate that the reliability and performance of many of these controls have not been well established. Accurate reporting of BMP effectiveness is important to modelers wishing to calibrate to actual practices.

The EPA/ASCE BMP Database (http://www.bmpdatabase.org/) currently characterizes BMP performance as EMC (i.e., a single representative concentration). Although some groups believe that a better method for measuring performance is amount of flow treated and effluent quality, (Strecker et al. 2001), an outlet EMC value is probably the simplest single measure of BMP performance.

While analyzing the EPA/ASCE BMP Database, Strecker et al. (2001) observed a trend indicating higher removal efficiency in BMPs with higher influent concentrations. Strecker et al. (2001) aver that several BMPs can be characterized by the effluent EMC distribution, that is, the effluent probability method (EPM), rather than a heuristic or process model of the system. However, current data available are insufficient to tie BMP design to effluent quality using the EPM, and it would be difficult to model BMPs in series (treatment train) with this method. As the EPA/ASCE BMP Database evolves, the EPM method (discussed below) may be enhanced.

Before discussing some EMC methods in more detail, it is important to note that *quality* is not the only performance measure. Strecker et al. (2001) proposed that there should be a threefold metric for BMP characterization:

- How much catchment runoff is averted by a control measure (hydrologic source control), e.g., by infiltration or evapotranspiration?
- Of the runoff that enters the BMP, how much is treated and how much is bypassed or routed through the BMP at rates exceeding treatment rates?
- What are the quality characteristics of the treated BMP effluent?

A fourth metric can be added:

- How much downstream flow management is provided by the BMP?

Metrics one, two, and four are hydrologic and depend strongly on regional patterns of storm events and dry-weather intervals (and of course, local catchment's characteristics such as soils). It will be seen in this report that SWMM is capable of good short-term and long-term hydrologic performance characterization – except for infiltration from channels. SWMM can also produce the event EMCs that can be used to characterize effluent quality. Finally, the viability of BMPs such as constructed wetlands and biofilters depends in part upon the nature of the entering stormwater. Whether or not the BMP can support healthy and diverse vegetation may depend on issues related to nutrient dynamics and subsurface water quality, which SWMM may or may not be able to address. The bulk of this report identifies ways in which SWMM deficiencies can be overcome in order to enhance the model's already powerful capabilities.

3.2 IDENTIFICATION OF COMMON EMC-BASED METHODS

The difficulty in selecting measures of efficiency stems from the desire to compare a wide range of BMPs and the large number of methods currently in use. There is much variation and disagreement in the literature about what measure of efficiency is best applied in specific situations; however, it is generally accepted that the EMC and long-term loading provide the best data for observing the effects of the BMP on acute and chronic pollution, respectively (GeoSyntec et al. 2002).

Ten methods for evaluating BMP performance using EMC data are summarized by GeoSyntec et al. (2002) in documentation for the ASCE/EPA BMP Database. These methods are used when the input database consists of event mean influent and effluent concentrations.

1. Efficiency ratio
2. Summation of loads
3. Regression of loads
4. Event mean concentration
5. Efficiency of individual storm
6. "Irreducible concentration" and "achievable efficiency"
7. Percent removal relative to water quality standards
8. Lines of comparative performance
9. Multi-variate and nonlinear models
10. Effluent probability method

All of these methods appear to be statistical models that are appropriate to use when the only input data that are available are storm event mean influent and effluent concentrations. They do not provide a process level characterization of performance. Four methods are described below, however, the ASCE study team (Strecker et al., 2001) does not recommend any of the first three (EMC, efficiency ratio, regression). In fact, the team recommends that a better measure of BMP performance is the amount of runoff prevented, the amount of flow treated (and bypassed) and the effluent quality of the treated flow. They have observed that BMPs tend to produce a fairly narrow range in effluent quality, and, therefore, percent removal is merely a function of how polluted the inflow is. Characterization of effluent quality is one feature of the EPM, also discussed below, and recommended by the team.

Additional work to standardize BMP monitoring protocols and calculations is needed to make monitoring data comparable from site to site (GeoSyntec et al. 2002). In addition to the EPA/ASCE BMP Database reports, documents that summarize BMP efficiency information include the National Pollutant Removal Performance Database (Brown and Schueler 1997), the Terrene Institute's report *The Use of Wetlands for Controlling Stormwater Pollution* (Strecker at al. 1992), and the recent text by Minton (2002).

3.3 EVENT MEAN CONCENTRATION

The term event mean concentration (EMC) is a statistical parameter used to represent the flow-proportional average concentration of a given parameter during a storm event. It is defined as the total constituent mass divided by the total runoff volume, although it is usually computed in practice by compositing multiple samples on the basis of the flow rate at the time the sample was taken, prior to sending the samples to the laboratory. The single flow-weighted composite then provides EMCs upon chemical analysis. When combined with flow measurements, the EMC can be used to estimate the pollutant loading from a given storm. Under most circumstances, the EMC provides the most useful means for quantifying the level of pollution resulting from a runoff event. Collection of EMC data has been the primary focus of the EPA/ASCE BMP database project.

The EMC for an individual event or set of field measurements, where discrete samples have been collected, is defined as:

$$EMC = \frac{\sum_{j=1}^{n} V_j C_j}{\sum_{j=1}^{n} V_j} \qquad (3\text{-}1)$$

where

V_j = volume of flow during the period j,

C_j = average concentration associated with the period and volume V_j, and

n = total number of measurements taken during event.

Currently, SWMM can easily produce a constant concentration (EMC value) in the Runoff Block and as the output of a treatment device in the S/T Block. However, storage and quality control devices in Runoff and Transport cannot easily (i.e., without odd data manipulations) produce a constant outflow concentration different from the inflow concentration. That is, it is not easy to have a constant inflow concentration of, say, 20 mg/L and a constant outflow concentration of, say, 8 mg/L from a Runoff or Transport flow element.

3.4 EFFICIENCY RATIO

The efficiency ratio (ER) is most often used to report BMP efficiency. While this may be appropriate for determining the reduction in pollution for an event, it may not indicate performance of a BMP over time, or for events of varying intensity and volume (GeoSyntec et al. 2002). Another reason is because the ER approach does not consider the statistical significance of the result. Most researchers assume that ER has the meaning of "percent removal." While this method is not recommended for monitoring the effectiveness of BMPs, because of the data availability and the wide spread analysis with this method, ER values are often reported in the literature. If modeling with a "black box" approach, average EMC and ER values are simple to implement, e.g., in spreadsheets. The efficiency ratio is defined in terms of the average EMC of pollutants over some time period:

$$\mathrm{ER} = 1 - \frac{\text{average outlet EMC}}{\text{average inlet EMC}} = \frac{\text{average inlet EMC - average outlet EMC}}{\text{average inlet EMC}} \qquad (3\text{-}2)$$

This method (GeoSyntec et al. 2002):

- Weights EMCs from all storms equally regardless of relative magnitude of storm. For example, a high concentration/high volume event has equal weight in the average EMC as a low concentration/low volume event.
- Minimizes the potential impacts of "smaller/cleaner" storm events on actual performance calculations. For example, in a storm-by-storm efficiency approach, a low removal value for such an event is weighted equally to a larger value.
- Allows for the use of data where portions of the inflow or outflow data are missing, based on the assumption that the inclusion of the missing data points would not significantly affect the calculated average EMC.

Comments:

- Many studies use the ER method to characterize performance, but it fails to take into account some of the complexities of BMP design, especially media filters and other BMPs that treat to relatively constant levels and are independent of inflow concentrations.
- This method also assumes that if all storms at the site had been monitored, the average inlet and outlet EMCs would be similar to those that were monitored.
- Under all circumstances this method should be supplemented with an appropriate non-parametric (or, if applicable, a parametric) statistical test indicating whether the differences in mean EMCs are statistically significant (it is better to show the actual level of significance found, rather than just noting whether the result was significant, assuming a 0.05 level).

Currently, SWMM can use an efficiency ratio (based on inflow *concentration*) to simulate BMP performance in the S/T Block. The Runoff and Transport Blocks allow application of a removal fraction to the incoming *load*, which is similar, but not the same. (It would be the same if the outflow equaled the inflow at the particular time step.)

3.5 REGRESSION MODELS

Regression methods may also be used to evaluate BMPs. These include regression of outlet load vs. inlet load ("load regression") and regression of ER values vs. inflow concentration. The former method has been used with some success to characterize performance (GeoSyntec et al. 2002), although there are many assumptions involved. The current version of SWMM provides this capability in the S/T Block through the use of its "universal removal equation" (Section 4.3). However, regression of ER values against inflow concentration produces spurious correlation (Benson 1965) since it amounts to regression of EMC_{out}/EMC_{in} vs. EMC_{in}. A seemingly significant regression is inevitable (Benson 1965). That is, the dependent variable includes as a divisor the independent variable EMC_{in}. The resulting relationship implies that removal efficiencies depend on influent concentration, but such a relationship may or may not exist. It always indicates a spurious relationship between percent removal and influent quality, and can lead to misguided "lines of comparative performance" (GeoSyntec et al. 2002, Minton 2002). Regression models are useful in doing preliminary investigations into cause-effect relationships, but they should be restricted to functional forms of how the BMP should work.

3.6 EFFLUENT PROBABILITY METHOD

The effluent probability method (EPM) is straightforward and provides a clear, but qualitative picture of BMP effectiveness. The EPM consists of a lognormal probability plot (although any distribution could be used for which probability paper exists, including normal) of EMC vs. either probability of occurrence or percent exceedance (equivalent to the cumulative distribution function, or CDF). Probability plots are among the most useful pieces of information that can result from a BMP evaluation study (Burton and Pitt 2001). The authors of the EPA/ASCE BMP Database *Monitoring Manual* strongly recommend that the stormwater industry accept this approach as a standard "rating curve" for BMP evaluation studies, as it provides a visual representation of the frequency distribution of both influent and effluent quality (GeoSyntec et al. 2002).

Lognormal plots are ordinarily used because the lognormal distribution has been found to be a good fit to most stormwater EMC data (USEPA 1983, Driscoll 1986, Van Buren et al. 1997). One advantage of normality, either of the logs or of the untransformed data, is that parametric statistical tests can be applied, such as the t-test, chi-square test, and analysis of variance. Statistical tests used to compare the difference between data sets typically require normality among the data sets, and some also require the sets to have the equal variances. Lognormal (and normal) probability plots can be used for qualitative guidance, since the same slope of two such curves indicates the same variance of the data.

The most basic test for normality is whether or not the data plot is a straight line on normal probability paper (or vs. equivalent values of the standard normal variate, z) (Burton and Pitt 2001, Bedient and Huber 2002). Tests for normality itself include tests directly related to probability plots, such as the probability plot correlation coefficient (PPCC) (Vogel 1986) and the Shapiro-Wilk test (Helsel and Hirsch 1992). Tests not related to probability plots include the Kolomogorov-Smirnov test and the chi-square test (Benjamin and Cornell 1970, Helsel and Hirsch 1992). However, the latter are less powerful, in a statistical sense, than tests that use probability plots, not to mention that the plots themselves yield great qualitative information, as discussed below. It is of interest that if ranked data are plotted against standard normal variates ("z-values") obtained from the inverse of the plotting position probability (Bedient and Huber 2002), using Excel, the linear fit of data (untransformed or logarithms) obtained using Excel's "trend line" option provides the required PPCC. The PPCC can then be tested for statistical significance (Vogel 1986, Helsel and Hirsch 1992). The critical values of the correlation coefficient account for the inherent correlation between two ranked data sets (order statistics). If normality, and in some cases equal variance, is ensured among the respective data sets, parametric tests can be employed to test the difference between the means/medians of the data sets.

Parametric and nonparametric statistical tests should be conducted after the probability plots are generated to indicate whether perceived differences in influent and effluent mean EMCs are statistically significant (it is preferable to provide the level of significance, instead of just noting whether the result was significant, such as at a 95% confidence level). Helsel and Hirsch (1992) provide an excellent primer on such methods, with applications to water resources and water quality. Many parametric and non-parametric tests are included in standard statistical software.

Regarding the effluent probability plots, there are limited quantitative assumptions that can be made simply on the basis of the plots themselves. As the data rarely enter into the plots as matched pairs (if the quantiles *did* occur as storm-event pairs, it would be sheer coincidence), there are limited inferences that can be made regarding BMP effectiveness over a particular concentration range. That is, one *cannot* make such inferences simply on a comparison of quantiles. Only when the data sets are entered as matched pairs, as for instance, in a simple scatter plot of EMC_{out} vs. EMC_{in}, can a quantitative assessment of the removal over specified concentration ranges be conducted. Other authors (Burton and Pitt 2001, p. 584) have discussed the EPM plots, drawing such inferences as "shows that SS are highly removed over influent concentrations ranging from 20 to over 1,000 mg/L." Unless the data are entered as matched pairs, such statements cannot be justified. For example, there may be one particular storm event in which no removal was observed (the influent and effluent EMC values are the same), but by plotting quantiles, the outcome of any single storm event may not be readily observed. There may be some effluent EMCs that are greater than the effluent data points for storm events, even though the generated trend line for the influent data set is still higher than the trend line generated for the effluent data set. Therefore, removal is not always guaranteed.

However, the relative range of both influent and effluent quality can be determined based upon the concentration range between given percentile values. In addition, the normality and equal variance among the data sets can be qualitatively observed, although any inferences about variance must be confirmed through quantitative statistical testing. Separation between the trend lines, an indication of removal, may be tested parametrically with a t-test if both lines are lognormal, or by using the non-parametric Kruskal-Wallis or Hodges-Lehmann tests, for example (Helsel and Hirsch 1992). Examples (Brown 2003) of effluent probability plots are shown as Figures 3-1 to 3-3, using data for an extended dry detention facility (Seton Pond) in Austin, Texas (Keblin et al. 1997).

The effluent probability plot for TSS at the Seton Pond facility is shown in Figure 3-1. From the figure, one can gage the relative range of influent and effluent EMCs, as signified by the concentration range between the 10th and 90th percentile. The slopes of the influent and effluent trend lines are similar, thus indicating similar variance between the logs of the two data sets. Both slopes appear somewhat flat, indicative of a relatively low coefficient of variation. But of considerable interest, contrary to what might be expected from a visual evaluation, the logs of the effluent TSS EMCs fail two normality tests (Brown 2003). Hence, the parametric t-test cannot be used to compare means. Finally, by comparing the data points plotted, it appears the lowest influent data point is still greater than the highest effluent point. Therefore, this BMP is projected to achieve removal, based upon the relative distance between the influent and effluent trend lines and the fact that there is no overlap among influent and effluent data points. This may be confirmed through the non-parametric Kruskal-Wallis test.

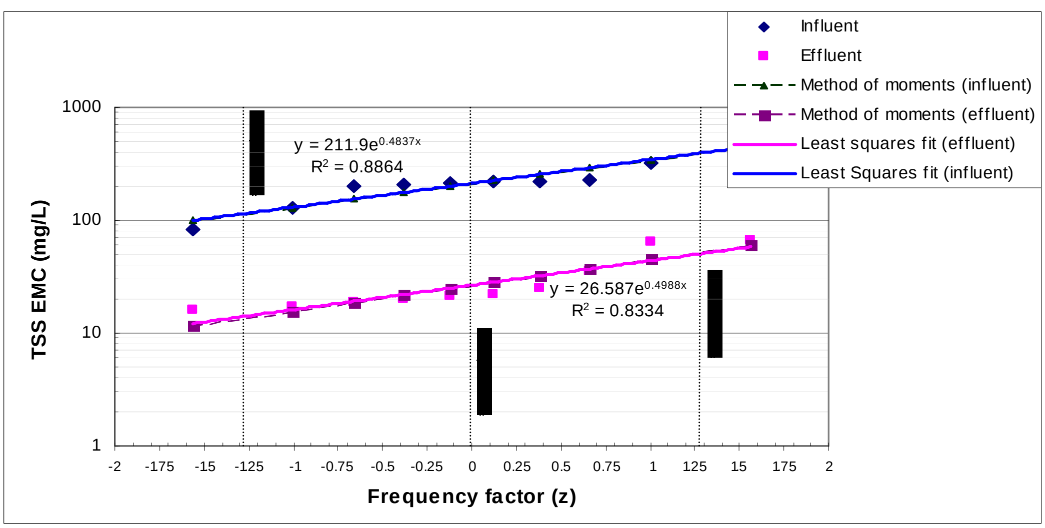

Figure 3-1. Probability plot for TSS removal at the Seton Pond facility, Austin, Texas (Brown 2003).

In comparison, Figure 3-2 shows the effluent probability plot for nitrate at the Seton Pond facility. Both the influent and effluent EMCs are confirmed to be lognormal (Brown 2003). The range of influent and effluent concentrations can still be observed, but the range of effluent concentrations actually exceeds the range of influent concentrations so little can be said about the system performance. The slopes of the influent and effluent generated trend lines are quite dissimilar, indicating differing variance among the logs of the nitrate data sets, and as trend lines intersect, there is little to be said about the removal at any range. With the trend line intersection, it is easily observed that the lowest influent data point is much lower than the highest effluent data point. Therefore this BMP is not predicted to achieve removal of nitrate, based upon the overlap and intersection of data points and generated trend lines. Indeed, the parametric t-test indicates that equality of the means of the logs cannot be rejected as does the non-parametric Kruskal-Wallis test (Brown 2003).

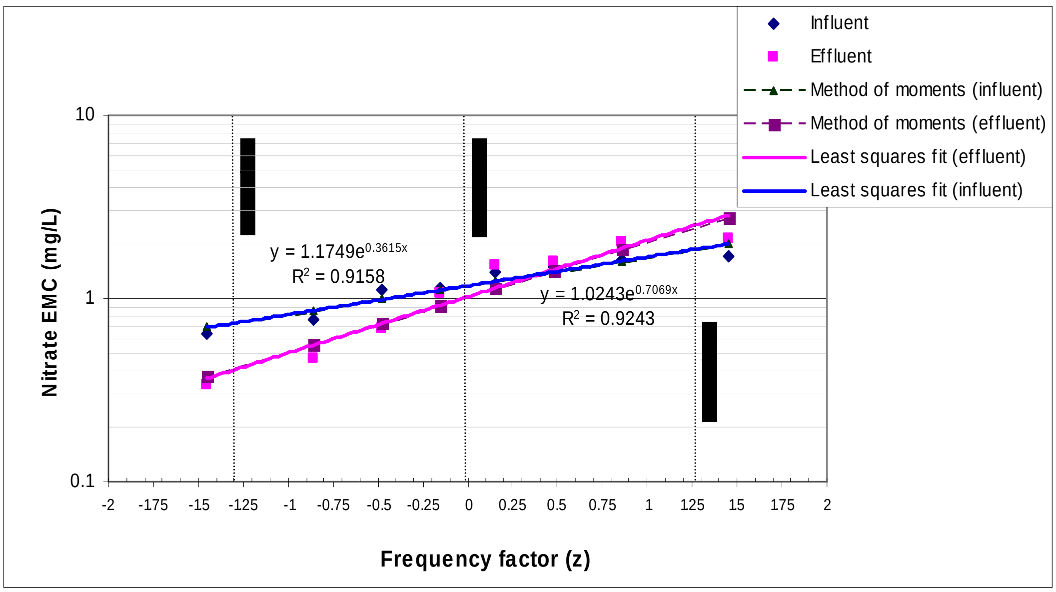

Figure 3-2. Probability plot for nitrate removal at the Seton Pond facility, Austin, Texas (Brown 2003).

The final example in Figure 3-3 shows the effluent probability plot for total zinc at the Seton Pond facility. The lognormality of both the influent and effluent EMCs may be confirmed by statistical tests. Like the previous plots, the range of influent and effluent concentrations is easily observed, with only a minimal overlap between influent and effluent EMCs. The slopes of the trend lines appear dissimilar, indicating differing variance among the data sets. But appearances can be deceiving! The hypothesis of equality of variance between influent and effluent EMCs is not rejected by three statistical tests (Brown 2003). The magnitude of the effluent slope is greater than the magnitude of the influent slope; therefore, the effluent data set is predicted to have a larger coefficient of variation (because the effluent mean of logs is also lower than the influent mean of logs). Comparing the data points themselves, it appears that the lowest influent data point is slightly lower than the highest effluent data point. Therefore, removal for this BMP can only be ensured through appropriate statistical examination, to determine whether a significant difference does exist between the two data sets, as it can not readily be shown through the plots. In fact, removal is confirmed by both the parametric t-test and non-parametric Kruskal-Wallis test (Brown 2003).

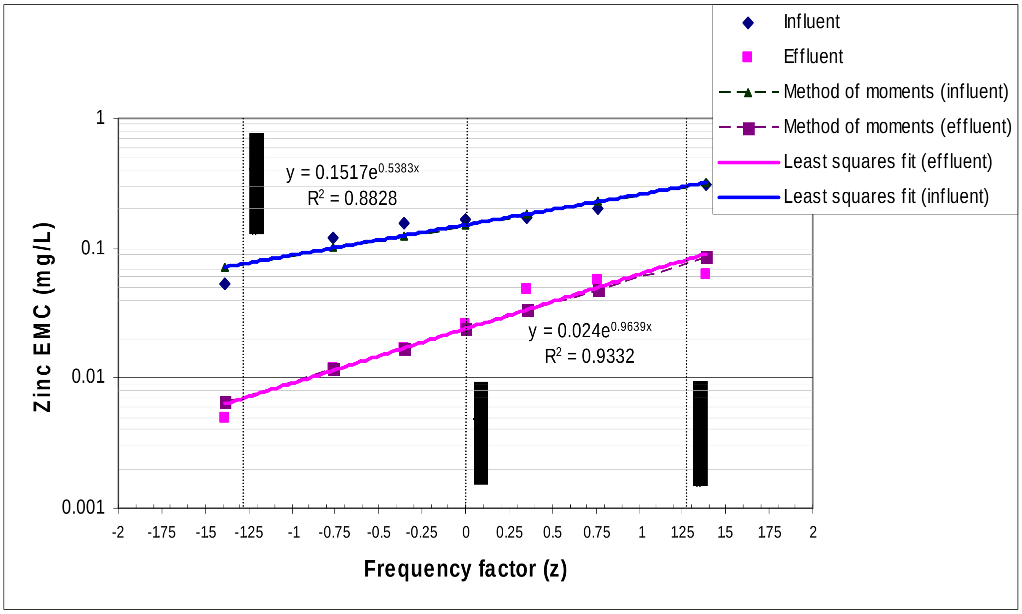

Figure 3-3. Probability plot for total zinc removal at the Seton Pond facility, Austin, Texas (Brown 2003).

In many ways, the qualitative inferences from the EPM may also be obtained from other descriptive statistics, such as box plots, as well as quantitatively through the parametric t-test and non-parametric comparisons of medians. However, the EPM has the advantage of illustrating the lognormal (or other distribution) fit of the data, rather than simply certain quantiles, as with box plots. The primary problem with the EPM is that certain quantitative assumptions, such as removal and performance at or around a certain concentration value, cannot be made unless data points are entered as matched pairs (e.g., as in scatter plots of effluent vs. influent EMC). This discrepancy was noted in the CalTrans BMP study and assessment (CalTrans 2003), indicating that interpretation of these plots should be performed in conjunction with related analyses, such as scatter plots. Another concern raised with the EPM is its ability to provide sufficient information regarding BMP selection. In areas requiring a set removal percentage, use of the EPM may not adequately portray whether a BMP is capable of meeting that performance standard (CalTrans 2003). However, the EPM retains the advantage of being able to deal easily with an unequal number of influent and effluent EMC data points.

Currently, SWMM is not capable of simulating a prescribed lognormal (or any) frequency distribution, either for influent concentrations or for BMP outflows. That is, it is not possible to enter parameters of a lognormal distribution as a Runoff Block option instead of, say, a buildup-washoff formulation. In principle, such a distribution could be observed over the period of a long-term continuous simulation from constituent quality generated by buildup-washoff or another mechanism (Huber et al. 1987), but the need here is to specify a distribution *a priori*. Equally useful would be a method to simulate the water quality of runoff on the basis of a specified frequency distribution. This would be an option to be used instead of buildup-washoff, constant concentration, etc. Likewise, the frequency distribution of BMP effluent EMCs could be prescribed, on the basis of observed data. However, it should be noted that the SWMM Statistics Block is capable of analyzing any flow or quality time series that can be placed on an "interface file" as the output of a SWMM Block, to generate frequency distributions and identify lognormal parameters through the method of moments. This could easily be enhanced through the SWMM5 GUI.

3.7 SIMULATION MODELS

Simulation models allow evaluation of long-term rainfall to assess volumes of stormwater treated by BMPs and how much is bypassed, or processed at rates that result in non-effective treatment. Thus, in principle, a simulation model can provide all the information needed to evaluate a BMP that would be prohibitively expensive to collect in a monitoring program – if the model is to be believed. This includes information such as much of the rainfall record is treated or controlled. The dynamics of the filling and emptying of these BMPs is vital to understanding treatment efficiencies and runoff removal rates. The bulk of this report will be devoted to evaluating the effectiveness of SWMM in this regard and examining alternative simulation algorithms from other models.

In the event that a model such as SWMM has not been specified *a priori*, searching for a model that characterizes BMP effectiveness is not simple. The lack of consistent and concise evaluation of modeling abilities makes choosing a model quite complex and time consuming. Just as BMP effectiveness monitoring is currently being standardized, so is model evaluation. Model descriptions may be found in texts and reports (e.g., Singh 1995, Field et al. 2001, Debo and Reese 2002, Field and Sullivan 2003, Field et al. 2004), but anything printed may soon lose currency. One such listing is shown in Table 3-1. Web pages, such as a circa-1998 site provided by the Great Lakes Sediment Management Program (http://www.glc.org/tributary/pdf/allweb32.pdf), can provide lengthier and timelier model information – if they are updated!

Some models listed in Table 3-1 do not perform quality calculations; they are included in this listing because 1) they are commonly used in urban areas of the U.S. and Canada, and 2) quality loads are often computed after an accurate hydrologic and hydraulic model is run, simply by multiplying flows by EMCs. Additional, less common models evaluated as part of this study are listed in Chapter 5.

Table 3-1. Primary urban hydrologic, hydraulic and water quality models commonly used in the United States.
Source: W.C. Huber, class notes. Updated March 25, 2003.

Model	Agency/Source	Primarily Hydrology/ Hydraulics	Continuous Simulation or Storm Event	Complete Dynamic Flow Routing?	Quality Simulation?	Graphical User Interface[j]
DR3M-QUAL[a]	USGS	Hydrology	CS/SE	No	Yes	ANNIE[a,d]
FEQ[a,b]	USGS	Hydraulics	SE	Yes	No	No
HEC-1/HMS[c]	HEC/Vendors	Hydrology	SE	No	No	Yes
HEC-2/RAS[c]	HEC/Vendors	Hydraul. (backwater)	Steady state	No	No	Yes
HSPF[a,d]	EPA	Hydrology	CS/SE	No	Yes	ANNIE[a,d], EPA BASINS
InfoWorks CS[e]	HR Wallingford in UK, Montgom. Watson in US	Hydrology/Hydraulics	CS/SE	Yes	Yes	Yes
MIKE 11[f]	Danish Hydraulics Inst.	Hydraulics (open channels)	SE	Yes	Yes	Yes
MOUSE[f]	Danish Hydraulics Inst.	Hydrology/Hydraulics	CS/SE	Yes	Yes	Yes
P8[g]	Wm. W. Walker, Jr.	Hydrology	CS/SE	No	Yes	Menu
Santa Barbara	Vendors	Hydrology	SE	No	No	3rd party
SCS[h]	NRCS/Vendors	Hydrology	SE	No	No	3rd party
*SewerCAT[i]	Reid Crowther Consult.	Hydraulics	SE	Yes	No	Yes
SLAMM[l]	R. Pitt	Hydrology	CS	No	Yes	No
*STORM[c]	HEC/Vendors	Hydrology	CS	No	Yes	No
SWMM[d,i,k]	EPA/OSU	Hydrology/Hydraulics	CS/SE	Yes	Yes	3rd party
UNET[c]	HEC/Vendors	Hydraulics	SE	Yes	No	HECDSS

Web addresses for models begin with prefix: http://

a. h2o.usgs.gov/software/surface_water.html
b. www.dilurb.er.usgs.gov/proj/feq/
c. www.hec.usace.army.mil/
d. www.epa.gov/ceampubl/softwdos.htm
e. www.hrwallingford.co.uk/
f. www.dhi.dk
g. www.wwwalker.net/p8/
h. www.ncg.nrcs.usda.gov/tech_tools.html
i. www.ccee.orst.edu/swmm (not currently available, contact Dr. C. Vitasovic at DHI)
j. 3p = 3rd party (e.g., an outside vendor)
k. www.epa.gov/ednnrmrl/swmm/index.htm (for SWMM5)
l. www.eng.ua.edu/~rpitt/SLAMMDETPOND/WinSlamm/WINSLAMM.shtml

*May not be currently available.

4 CURRENT SWMM SIMULATION CAPABILITIES

4.1 THE MODEL

The EPA SWMM is a dynamic rainfall-runoff simulation model, primarily but not exclusively for urban areas, for single-event or long-term (continuous) simulation. Flow routing is performed for surface and sub-surface conveyance and groundwater systems, including the option of fully dynamic hydraulic routing in the Extran Block. Nonpoint source runoff quality and routing may also be simulated, as well as storage, treatment and other BMPs.

Version 4 of the SWMM evolution is currently (March 2004) at version 4.4h (Huber and Dickinson 1988, Roesner et al. 1988). The most current release of version 4.4h can be downloaded from the Oregon State University website (www.ccee.orst.edu/swmm). Simulation of storage and treatment was generalized in Version III of the model (Nix et al. 1978) so that algorithms were not specific to individual devices. SWMM has been through many modifications since Version II, and since the first version 4 release in 1988, it now includes several options that enhance the model's ability to simulate BMPs and general control options for management of stormwater and combined sewers (Huber 1996, Huber 2001b). Version 4.4h will gradually be replaced by SWMM5, developed by the EPA, as that object-oriented model and its GUI gain functionality (http://www.epa.gov/ednnrmrl/swmm/index.htm). However, most of the BMP and LID simulation capabilities described for version 4.4h also apply to SWMM5.

SWMM capabilities regarding simulation of wet-weather control (WWC) alternatives have been discussed in Chapter 1 and summarized in Table 1-4. The following text discusses significant control options and consequent SWMM simulation capabilities.

4.2 STORAGE

Storage in an urban catchment may occur on the ground surface, in the drainage system, and in specific storage devices (e.g., ponds, tanks, secondary flow removal devices). Pollutant removal occurs primarily through sedimentation and decay. SWMM is most effective at simulating storage-type BMPs because such devices have been extensively studied and information about them is widely available. Flow routing through storage is easily performed using a variety of methods, and multiple outlets (and bypass) may readily be simulated. SWMM also has the ability to simulate hydraulic controls and time-dependant regulators (such as weir and orifice settings that depend on stages in specified locations), although these options are primarily located within the Extran block, which does not model water quality. Storage options are available in the Runoff, Transport, Extran and Storage/Treatment (S/T) Blocks.

S/T quality removal may be based on a first-order process, for which simple finite difference formulations exist for well-mixed (i.e., continuous-flow, stirred-tank reactors or CFSTRs) or plug-flow systems. (However, see discussion of possible errors in the manner in which the S/T Block simulates well-mixed, first-order decay, in Section 4.3 below.) A second mechanism consists of a generalized removal equation designed such that most empirical or other results can be mimicked (Section 4.3). For example, removal of a given pollutant can be computed as a function of concentration of any pollutant, removal fraction of any other pollutant, and/or detention time. A third removal option is by sedimentation using Camp's (1946) theory for quiescent conditions, modified for turbulence by Chen (1975). For this formulation up to five settling velocity ranges must be defined and then routed through a progression of up to five S/T units. While this third option is ideal for simulation of S/T units in series (e.g., a treatment train), unfortunately, such pollutant characterization is not generated elsewhere (upstream) in the model. That is, *treatability* data are typically not generated by the model upstream of the S/T Block and cannot be linked – unless the user specifically simulates discrete particle size ranges as separate constituents in upstream blocks, which certainly is an option.

Removal by first-order decay may also be simulated in the Transport, and Runoff Blocks, assuming complete mixing (in accordance with their quality routing methods) within conveyance and storage elements. Constituents in Transport and Runoff Block channels, pipes, and storages are also subject to removal based on constant removal fractions applied to incoming loads and settling with a constant settling velocity (Huber 2001b).

Interactions among water quality constituents are simulated minimally in the S/T Block (e.g., removal of one pollutant can depend upon removal of another, in the manner of sorption). The Transport Block has a relatively untested capability to simulate linked BOD-NOD-DO dynamics, essentially using a modified Streeter-Phelps analysis (Huber 2001b) in the manner of WASP (Wool et al. 2001). There is only limited capability in SWMM to simulate combined physical-biological removal processes in wetlands and bioswales except to the extent that such removal can be characterized by the processes to be described.

To summarize, fundamental process categories of sedimentation, biological removal, sorption, filtration, flotation, chemical treatment, high-rate biological treatment, degradation, hydrodynamic separation may all be *mimicked* by the S/T Block as long as removal characterization relations are available. However, the only FPCs simulated explicitly in storage are first-order decay (e.g., for degradation) and sedimentation.

4.3 S/T GENERALIZED REMOVAL EQUATION

Because the generalized removal equation or "universal removal equation" or "one-equation-fits-all" may be used for several purposes, additional information is provided here.

The "universal removal equation" within the S/T Block is

$$R = \left(a_{12} e^{a_1 x_1} x_2^{a_2} + a_{13} e^{a_3 x_3} x_4^{a_4} + a_{14} e^{a_5 x_5} x_6^{a_6} + a_{15} e^{(a_7 x_7 + a_8 x_8)} x_9^{a_9} x_{10}^{a_{10}} x_{11}^{a_{11}}\right)^{a_{16}} \qquad (4\text{-}1)$$

where

x_i = removal equation variables (model state variables),

a_i = coefficients,

R = removal fraction, $0 \le R \le RMX \le 1.0$, and

RMX = maximum removal.

The removal fraction is applied to the load (mass) in a plug (for plug flow), at each time step. The removal fraction can thus be made a function of selected state variables, x_i. Each removal equation variable (selected state variable), x_i, may represent one of several variables available in the program at each time step, such as an inflow concentration or residence time, described further below. With these variables and the coefficients, a_i, the user can develop the desired removal equation.

An example of the application of Equation 4-1 for plug flow is provided by the common exponential removal equation characteristic of tank reactors and ponds observed in EPA Nationwide Urban Runoff Program (NRUP) studies (USEPA 1983). For suspended solids (SS) this might be:

$$R_{SS} = R_{max}\left(1 - e^{-kt_d}\right) \qquad (4\text{-}2)$$

where
R_{SS} = suspended solids removal fraction, $0 \le R_{SS} \le R_{max} \le RMX$,
R_{max} = maximum removal fraction in fitted equation,
RMX = maximum removal, an S/T input variable, ≤ 1.0.,
t_d = detention time (sec), and
k = first order decay coefficient (1/sec).

This equation can be constructed from Equation 4-1 by setting $a_{12} = R_{max,}$ $a_{13} = -R_{max,}$ $a_3 = -k$, $a_{16} = 1.0$, and letting x_3 = detention time, t_d (by setting INPUT(I,IP,3) = 1 on data group G2 in SWMM S/T). All other coefficients, a_j would equal zero. RMX would not be necessary, as R_{max} limits the value of R.

Well-mixed storage units are simulated by a finite-difference solution to the conservation of mass equation for a CFSTR (Chapra 1997):

$$\frac{dVC}{dt} = I\,C^{I} - QC - kCV \qquad (4\text{-}3)$$

where
C = well-mixed concentration in storage unit,
V = volume in storage unit,
I = inflow to storage unit,
C^I =inflow concentration in inflow to storage unit,
Q = outflow from storage unit,
t = time, and
k = first-order decay coefficient, 1/time.

All variables except k are functions of time for a variable-volume storage device. Since storage routing is performed before the quality computations, inflows, outflows, and volumes are known at the beginning and end of each time step, leading to a finite difference formulation used in SWMM S/T,

$$\begin{matrix}\text{Change in} \\ \text{mass in basin} \\ \text{during } \Delta t\end{matrix} = \begin{matrix}\text{Mass entering} \\ \text{during } \Delta t\end{matrix} - \begin{matrix}\text{Mass leaving} \\ \text{during } \Delta t\end{matrix} - \text{Decay during } \Delta t$$

$$C_{n+1}V_{n+1} - C_n V_n = \frac{C_n^I I_n + C_{n+1}^I I_{n+1}}{2}\Delta t - \frac{C_n Q_n + C_{n+1} Q_{n+1}}{2}\Delta t - k\frac{C_n V_n + C_{n+1} V_{n+1}}{2}\Delta t \tag{4-4}$$

where subscripts n and n+1 refer to the beginning and end of a time step, Δt. Equation 4-4 is then solved for the unknown CFSTR (and effluent) concentration C_{n+1},

$$C_{n+1} = \frac{C_n V_n + \frac{\left(C_n^I I_n + C_{n+1}^I I_{n+1}\right)}{2}\Delta t - \frac{C_n Q_n}{2}\Delta t - \frac{k C_n V_n}{2}\Delta t}{V_{n+1}\left(1 + \frac{k\Delta t}{2}\right) + \frac{Q_{n+1}}{2}\Delta t} \tag{4-5}$$

When a CFSTR is simulated, the equation (Subroutine EQUATE, called from Subroutine UNIT in the S/T Block) returns the product of $k \cdot \Delta t$, where k = first order decay coefficient (1/time) and Δt = time step. This is done mechanistically by setting the x_2 variable = 1 on the S/T G2 line such that it will equal Δt when Equation 4-1 is evaluated. Then set a_2 and a_{16} = 1.0, and a_{12} = k.

However, there are some undocumented time step limitations to this process with the current Fortran coding. The first is that the maximum value of the product $k \cdot \Delta t$ is input parameter RMX (maximum removal per time step, useful for plug flow but also used for complete mixing), which in turn has a maximum value of 1.0. Hence, the maximum value of k that the program will end up using is RMX/Δt or 1/Δt. Thus, if $k \cdot \Delta t > 1.0$, an effective k value will be used, 1/Δt, that is smaller than that input by the user. The other consideration is the nature of the finite difference form of Equation 4-3, namely Equation 4-4. For constant inflow, outflow and, therefore, constant volume, the finite difference solution (Equation 4-5) can be reduced to:

$$C_{n+1} = \frac{\frac{\overline{C}^I Q}{V}\Delta t}{1 + \frac{\Delta t}{2}(\frac{Q}{V} + k)} + C_n \frac{1 - \frac{\Delta t}{2}(\frac{Q}{V} + k)}{1 + \frac{\Delta t}{2}(\frac{Q}{V} + k)} \tag{4-6}$$

where
n, n+1 = previous and new time steps, respectively,
C = concentration in outflow and in storage unit,
$\overline{C}^I$ = average inflow concentration over the time step,
Δt = time step,
Q = constant inflow and outflow, and
V = constant volume.

It can be seen that if $1 - \frac{\Delta t}{2}(\frac{Q}{V} + k) < 0$, negative concentrations can result, and if the term is

< -1, unstable computations can result. Because Q and V vary during a real S/T simulation, it is not necessarily easy to predict whether or when this will occur.

The conclusion is that Equations 4-4 and 4-5 are a poor method for solving Equation 4-3. Better would be to use a standard 4th order Runge-Kutta (RK4) numerical solver for the ordinary differential Equation 4-3. This is the method that will be implemented in SWMM5.

In passing, it should be observed that the current SWMM solution for complete mixing in Runoff and Transport channels, pipes, and storage elements does not use the S/T finite difference method. Instead, an integrated form of Equation 4-3 is used, wherein the analytical solution is obtained for a duration of one time step, as explained in Appendix IX of the User's Manual (Huber and Dickinson 1988). The method has proven stable and non-negative with respect to the time step. However, the only removal option is first-order decay except for the largely untested implementation of a settling velocity and constant removal fraction, provided to SWMM version 4.4h in spring 2002 and described in the next subsection.

Considerable additional concepts for simulating storage in SWMM are provided during the discussion of ponds, in Section 6.4. The transition between plug flow and complete mixing is also discussed therein.

4.4 MODIFIED STREETER-PHELPS OPTION

4.4.1 Objectives

The Transport Block is typically applied for urban drainage systems consisting of conduits and open channels, but there are often instances in which it would be useful to have a simplified receiving water quality simulation capability available. This is available with the modified Streeter-Phelps option, for which parameters are input in data groups F3 – F6 (with reference to SWMM version 4.4h). These options permit dissolved oxygen (DO) to be linked to decay of carbonaceous biochemical oxygen demand (CBOD) and nitrogenous oxygen demand (NOD). This option is probably warranted only when Transport is used to simulate a stream extending several miles downstream from the urban loading. Most sewer systems remain saturated with DO because of their high turbulence, in spite of the heavy oxygen demand. Nonetheless, this modified Streeter-Phelps option may be run for any Transport channel, conduit, or storage unit. In fact, quality routing is performed for any element containing a volume of water, including wet wells and manholes containing surcharge. As explained earlier, use of SWMM version 4.4h will diminish with the release of SWMM5, hence, the descriptions that follow may be considered as candidate algorithms for inclusion in SWMM5 flow objects.

The modified Streeter-Phelps method is included as a WASP model option (Wool et al. 2001), and the theoretical formulation (described below) in SWMM is very similar. The WASP model itself is another (much more comprehensive) option for simulation of receiving water quality. A special hydrodynamic link from the version 4.4h Transport Block allows Transport to "drive" the WASP water quality model by providing the needed file of flows, velocities, and storages. A similar option is available in the Extran Block. Although the SWMM Transport formulation is based on the WASP model, classic DO kinetics apply (e.g., Thomann and Mueller 1987, Chapra 1997).

The presentation in this subsection also serves the purpose of introducing a more general form of CFSTR kinetics than is provided by Equation 4-3. Although the kinetics to be demonstrated in Equations 4-9 – 4-12 are specifically for the Modified Streeter-Phelps formulation, it will be seen that variants on Equation 4-9 (for just one constituent) are used in the model algorithms of several models that are discussed in Chapters 6 and 7.

4.4.2 Theory

Classic Streeter-Phelps simulation of water quality applies to steady-state flows and considers only input of ultimate biological oxygen demand (BOD_u or CBOD in notation that follows), linked to DO through its first-order decay. In turn, DO is depleted by BOD and replenished by reaeration. The so-called modified Streeter-Phelps includes an additional nitrogenous oxygen demand, wherein total Kjeldahl nitrogen (TKN) decays to nitrate-nitrogen (NO_3-N). Formally,

$$\text{Organic-N} \rightarrow \text{Ammonia} \rightarrow \text{Nitrite} \rightarrow \text{Nitrate} \tag{4-7}$$

during which *Nitrosomonas* bacteria convert ammonia to nitrite and *Nitrobacter* bacteria convert nitrite to nitrate. Hence, there are three reactions to consider. The reactions are usually considered to be first-order, but Michaelis-Menten (or Monod) kinetics may also be employed (Chapra 1997), an option in WASP (Wool et al. 2001) but not SWMM. The modified Streeter-Phelps method simplifies the process by ignoring the conversion of organic-N to ammonia and using their sum, TKN = organic-N + ammonia-N, as total nitrogenous oxygen demand, or NOD. The NOD is decayed directly to nitrate, without the typically-brief intervening nitrite step. Hence, the true three-reaction process is reduced to one. Chapra (1997, p. 425) points out the shortcomings of this method, including the fact that because of the delay in converting organic-N to ammonia, there is usually a delay in the appearance of a nitrogenous oxygen demand relative to CBOD. Nonetheless, the objective was not to develop the Transport Block as a sophisticated receiving water quality simulation model, but rather to provide for at least a minimal DO simulation capability, similar to this intermediate (modified Streeter-Phelps) option in the WASP model.

When ammonia is oxidized to nitrate, oxygen is used according to

$$NH_3^+ + 2\,O_2 \rightarrow NO_3 + H_2O + H^+ \tag{4-8}$$

From the ratio of molecular weights, 64/14 = 4.57 mg of oxygen are needed to convert one mg of NH_3-N to NO_3, where 64 = 2 x molecular weight of O_2 and 14 is the atomic weight of N. This stoichiometric factor is included in the coupled conservation equations.

The SWMM Transport Block simulates the linked DO-BOD-CBOD process in four steps for every element containing storage (including pump wet wells and surcharged manholes), as follows (symbols are defined in Table 4-1 following the four equations):

Step 1, for CBOD $\equiv C_1$:

$$V\frac{dC_1}{dt} + C_1\frac{dV}{dt} = (Q_iC_1^{in} + L_1)(1-R_1) - QC_1 - k_1C_1V - v_{s1}(1-F_1)C_1A_s \tag{4-9}$$

(*Equation 4-9 will also serve as a more general reference equation for CFSTR kinetics in sections that follow, wherein C_1 would be any constituent of interest.*)

Step 2, for NOD $\equiv C_2$:

$$V\frac{dC_2}{dt} + C_2\frac{dV}{dt} = (Q_iC_2^{in} + L_2)(1-R_2) - QC_2 - k_NC_2V - v_{s2}(1-F_2)C_2A_s \tag{4-10}$$

Step 3, for NO_3-N $\equiv C_3$:

$$V\frac{dC_3}{dt} + C_3\frac{dV}{dt} = (Q_i C_3^{in} + L_3)(1 - R_3) - QC_3 + k_N C_2 V - v_{s3}(1 - F_3)C_3 A_s \quad (4\text{-}11)$$

Step 4, for DO *deficit* $D \equiv C_s$ – DO:

$$V\frac{dD}{dt} + D\frac{dV}{dt} = Q_i D^{in} - QD + k_1 C_1 V + 4.57 k_N C_2 V - k_2 D - \frac{SOD}{\bar{d}} V \quad (4\text{-}12)$$

Table 4-1. Symbols and parameters for linked DO-BOD-NOD simulation, Equations 4-9 – 4-12.

Equation Symbol	**SWMM v. 4.4h Name**	**Units**[a]	**Definition**	**Input Para-meter**	**Value Range**[c]
V	VOL	ft^3	Volume		
Ci	CPOL	mg/L	Concentration of constituent i		
D	DEFICIT	mg/L	DO deficit = C_s – DO		
DO	CPOL(4)	mg/L	Dissolved oxygen conc.		
C_s	CSAT	mg/L	Saturation DO conc.		
Q_i	QI	cfs	Inflow to element		
Q	QO	cfs	Outflow from element		
dV/dt	DVDT	cfs	Change in volume of element during one time step		
C_i^{in}	CPOL	mg/L	Concentration of inflow		
L_i		cfs·mg/L	Other loads. In SWMM, these are due to possible scour and deposition		
R_i	TREMOVE	none	Removal fraction for constituent i applied to incoming load to element,	Y	0 – 0.9
k_1	DECAY(1)	1/day	First-order BOD deoxygenation coefficient, sometimes known as k_d.	Y	0.1 - 5
v_{si}	VSETL	ft/s	Settling velocity for constituent i	Y	
A_s	AS	ft^2	Water surface area of element		
F_i	FDIS	none	Dissolved fraction of constituent i (does not settle)	Y	
k_N	DECAY(2)	1/day	First-order NOD deoxygenation coefficient.	Y	0.1 - 2
k_2	DECAY(4)	1/day	Reaeration coefficient, sometimes known as k_a.	Y[b]	1 - 5
SOD	SOD	g/ft^2-day	Sediment oxygen demand	Y	0.006 - 1
$\bar{d}$	DBAR	ft	Average depth = cross sectional area / top width = volume / surface area		

[a] Units are for those required for input if an input parameter, else, for internal use in the Transport Fortran code.
[b]The reaeration coefficient may also be computed by the program as a function of velocity, depth, and wind speed.
[c] Value ranges are given only for input parameters. When no range is entered, values are too site specific to list. Good sources for parameter estimates include Mills et al. (1985), Schnoor et al. (1987), Thomann and Mueller (1987), Chapra (1997) and Wool et al. (2001).

When the modified Streeter-Phelps option is used, CBOD, NOD, NO_3-N and DO must be the first four constituents simulated. Additional quality constituents can be identified starting with constituent 5, etc.

The total mass of a constituent consists of a dissolved and particulate form, i.e.,

$$C_{tot} = C_{dis} + C_{part} \qquad (4\text{-}13)$$

where
C_{tot} = concentration of total mass of constituent, mg/L,
C_{dis} = concentration of dissolved component, mg_{dis}/L, and
C_{part} = concentration of particulate component, mg_{part}/L.

The dissolved fraction may be determined if the *partition coefficient*, K_d, is known (Chapra 1997):

$$r = K_d C_{dis} \qquad (4\text{-}14)$$

where
r = ratio of mass in particulate (adsorbed) form to mass in dissolved form, mg_{part}/mg_{dis}, and
K_d = partition coefficient, L/mg_{dis}.

The relationship (isotherm) for r between the particulate and dissolved fraction may be in a linear, power function (Freundlich), or Monod-type (Langmuir) form (Chapra 1997) and may be determined experimentally. The particulate concentration is related to the concentration of suspended solids, C_{SS}, by

$$C_{part} = r\, C_{SS} = K_d\, C_{dis}\, C_{SS} \qquad (4\text{-}16)$$

Hence, the total concentration is

$$C_{tot} = C_{dis} + K_d\, C_{dis}\, C_{SS} \qquad (4\text{-}17)$$

and the ratio of dissolved to total is

$$F = \frac{1}{1 + K_d C_{SS}} \qquad (4\text{-}18)$$

and the particulate fraction is

$$1 - F = \frac{K_d C_{SS}}{1 + K_d C_{SS}} \qquad (4\text{-}19)$$

Hence, the dissolved and particulate fractions in Equations 4-9 – 4-12 and more generally for any constituent can be determined if the partition coefficient is known.

Equations 4-9 – 4-12 illustrate the typical way of linking the decay of CBOD and NOD to oxygen demand through first-order rate processes. The equations are solved in order, and the solution to earlier equations (e.g., 4-9 for C_1 and 4-10 for C_2) are inserted into later equations; all of which are solved analytically using average values for flow and volume over a time step. The solution process is described in the version 4 documentation (Huber and Dickinson 1988) Appendix XI and documented with comment statements in the Fortran subroutines QUAL, QUALPARM, QUALSOLN, and REAERATE. Conceptually the solution process is likely to be simpler in SWMM5 (if implemented) through the use of a Runge-Kutta solution for the four simultaneous equations.

The equations have similar parameters, with the exception of equation 4-12 for DO deficit. No removal fraction is assumed to be available for DO, nor is there a scour-deposition load. Both effects are accounted for in the relation of DO to CBOD and NOD. Although nitrate is allowed to be non-conservative, in natural waters this is usually manifested by uptake by algae. While a non-conservative effect could be simulated with Transport, this model is unable to "close the loop" involving conversion to organic nitrogen, etc. A complete ecological model that includes algal and phosphorus dynamics is needed if such effects are to be simulated, such as WASP (Wool et al. 2001) or HSPF (Bicknell et al. 1997).

Additional assumptions include:

- First-order rate constants are not differentiated in Transport between sewage or "bottle" rates and in-stream rates. However, a different k_1 value can apply along stream segments than applies for conversion of BOD5 to CBOD. The value used to convert 5-day BOD (BOD5) to ultimate BOD (CBOD) is the DECAY value entered for CBOD. The conversion is (Chapra 1997):

  $$CBOD = \frac{BOD5}{(1 - e^{-5k_1})} \tag{4-20}$$

  with units of day^{-1} for k_1. Hence if BOD loads are provided for BOD5 rather than for CBOD, inflows will be converted to CBOD by Transport using Equation 4-20. Differing in-stream values for k_1 and k_N may be entered for desired elements, else earlier values entered serve as the default values.
- Differentiation of k_1 values used in the stream into stream decomposition and stream settling rates (day^{-1}) must be made by the user. But since settling is included directly as a removal process, this should not be necessary.
- The simplification of nitrogen dynamics into one first-order process has already been discussed. This is the fundamental assumption of the modified Streeter-Phelps equations.
- "Travel time" commonly encountered in Streeter-Phelps formulations here is volume/flow, V/Q, for an element, each of which is treated as a CFSTR. Concentrations are averages for the entire volume of the element, and the same as the outflow concentration.
- It is important to consider the volume change term, dV/dt, especially for situations in which an element is only draining (no inflow) or only filling (no outflow). The dV/dt term can also act to *concentrate* constituents when there is evaporation. Evaporation is not currently included in the Transport Block, but the same equations apply in the Runoff Block where the effect of evaporation can be seen to concentrate pollutants in very shallow flows.
- Settling is allowed for every constituent except DO (or DO deficit). For a constituent that is normally dissolved, such as NO_3-N, simply set $v_{si} = 0$ and/or the dissolved fraction, $f_i = 1.0$.

Temperature corrections may be applied to the k_1 and k_N values according to the Arrhenius equation (Chapra, 1997):

$$k(20^oC) = k(T^oC)\ \theta^{(T-20)} \tag{4-21}$$

where θ is a dimensionless temperature coefficient. Typical values are 1.047 for BOD (i.e., for K_1) and 1.08 for NOD (i.e., for K_N) (Thomann and Mueller 1987). "Theta values" THETA1 for BOD and THETA2 for NOD must be provided as input values to the program.

4.4.3 *Reaeration*

Oxygen is replenished by flow-driven and wind-driven reaeration. Options for both are similar to those of the WASP model (Wool et al. 2001), which uses Covar's (1976) formulation for the flow-driven reaeration coefficient, K_2, and O'Connor's (1983) formulation for wind-driven reaeration. Flow-driven options allow a reaeration equation of the form,

$$K_2(20^{o}C) = \text{Coef1}\frac{U^{\text{Coef 2}}}{\bar{d}^{\text{Coef 3}}} \qquad (4\text{-}22)$$

where

U = average velocity in flow element, ft/s,

and Coef1, Coef2, and Coef3 are empirical coefficients commonly employed in reaeration equations (Rathbun 1977). The user may input his/her own values for the three coefficients or use the Covar (1976) option for which the coefficients are set to values corresponding to three different equations depending on the flow regime. For wind-driven reaeration, O'Connor's (1983) methods are followed, involving an iterative solution for the drag coefficient.

4.4.4 *Summary*

The Modified Streeter-Phelps solution just described is applicable to the discussion of BMP simulation since treatment (e.g., decay, settling) can occur along any flow path. Apart from the linked constituents represented by Equations 4-9 – 4-12, the current Transport and Runoff quality routing formulations are of the form of Equation 4-9 and permit decay, settling, and removal (with load fraction R) within any flow element. To the extent that a BMP may be represented by a fundamental process such as decay or settling, or by a load removal fraction, Equation 4-9 is applicable, and several variations on this basic equation will be seen in sections that follow, particularly for ponds and wetlands. If warranted, the equation could be adapted to one-dimensional analysis instead of the CFSTR formulation, through inclusion of an advective term. Equation 4-9 may be solved analytically (over one time step, with average flow parameters, e.g., Medina et al. 1981, Huber and Dickinson 1988, Appendix IX) or numerically for the downstream concentration. Hence, BMP simulation in SWMM5 will likely include computations involving fundamental processes as well as simple, empirical removal fractions.

4.5 INFILTRATION

Infiltration into the soil is simulated only for Runoff Block overland flow planes. With the capability to route overland flow from one overland flow plane to another, infiltration of runoff diverted to large surfaces such as lawns and vegetated buffers may be simulated easily (see Section 4.6). Miniature overland flow surfaces as might be found in rain gardens, roof vegetation, and infiltration trenches may also be simulated with this option; the key assumption is one of vertical "walls." Hence, many LID options can be simulated in this manner, especially since overland flow planes are also subject to evapotranspiration (ET) and possible groundwater interaction. Since overland flow is also subject to first-order decay, constant removal fractions (based on load), and constant settling velocities, these plane segments offer several options for simulation of infiltration BMPs. However, the lack of infiltration from more general channel segments limits SWMM's ability to simulate swales and infiltration from porous channels, not to mention storage devices in general. Part of the difficulty is determining the effect of sedimentation and water depth on infiltration rates. In real systems, some measure of maintenance and perhaps seasonality should also be considered.

Porous pavement is an important option for reduction of runoff volumes. The current SWMM can simulate porous pavement to the extent that the infiltration through the pavement (or paving stones) can be modeled using Horton or Green-Ampt infiltration equations (see Section 12.3). Subsurface drainage (e.g., out a highway embankment) may be simulated using the groundwater option (James et al. 2001). But SWMM does not simulate quality processes in groundwater.

To summarize, infiltration may be simulated only for overland flow planes, not for channels. This still permits a wide variety of LID and BMP options to be modeled, but the lack of combined infiltration and channel flow routing is a SWMM limitation.

4.6 OTHER WET-WEATHER CONTROL OPTIONS

Other wet-weather control options include non-structural measures such as street cleaning, catch-basin cleaning, and pollutant load reduction. Street cleaning may be simulated directly in the SWMM Runoff Block. Catch-basin cleaning and other pollutant load reduction measures (e.g., "good housekeeping") may be simulated only by a reduction in buildup rates or assumed EMC values.

Several hydraulic control options may be simulated in the Extran Block and to a lesser degree in the Transport, Runoff, and S/T Blocks. Within Extran for instance, flow regulation based on stages and timed orifice settings may be simulated, as well as complex combinations of storage, pumping, and bypassing. The Extran capability points toward real-time control (RTC) simulation, but such capability awaits future enhancements in SWMM5. SWMM5's ability to be stopped in the middle of a simulation for a change in regulator settings and/or other variables is especially intriguing for RTC simulation. Although flow rates and volumes may be tabulated along various pathways, corresponding water quality is not yet simulated within Extran. If a constant EMC can be assigned to various pathways, after-the-fact estimates of loads may be made.

Continuity checks can be used to reflect various removal pathways including infiltration. Iterative application of SWMM can be used to design a facility with the desired combination of quantity and quality control (Huber 2001b) – within the limits of the model to simulate such controls.

4.7 LID SIMULATION OPTIONS

Hydrologic source control is at the heart of LID, for which every effort is made to retain stormwater at or near its source and dispose of it via infiltration and ET. For LID technologies, modeling options are needed that allow runoff to be directed from one subcatchment to another (for areas with different slopes, soil types or ground cover), and that allow impervious areas to be routed over pervious areas (and vice versa), e.g., rooftop runoff to be routed over lawns. Overland flow infiltration-based controls may be simulated in the SWMM Runoff Block by redirecting runoff from impervious areas onto pervious areas, and from one subcatchment to another (Huber 2001a), in the manner shown in Figure 4-1. It shows routing from the impervious sub-area of a subcatchment to the pervious sub-area of a subcatchment. The scheme is similar for flow from pervious to impervious sub-areas, and subcatchment outflow can also be directed to another subcatchment. Lee (2003) demonstrates the model's efficacy for analysis of distributed stormwater quantity and quality control alternatives.

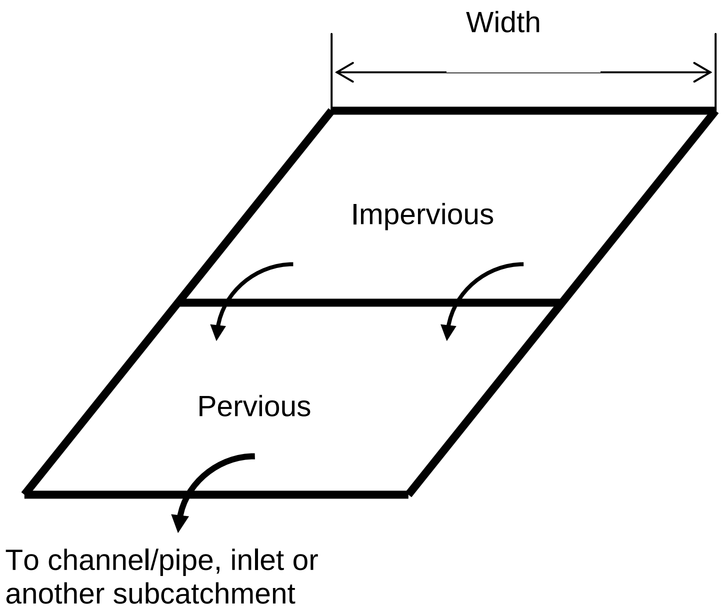

Figure 4-1. Conceptual routing from the impervious sub-area of a subcatchment to the pervious sub-area of a subcatchment (Huber 2001a).

This rerouting also allows for the simulation of buffer strips or riparian zones. Inflow to the downstream subcatchment is distributed uniformly over the downstream subcatchment in the same manner as rainfall. This can be done because of the nonlinear reservoir flow routing method in which there is no longitudinal variation through the subcatchment. Runoff Block overland flow planes simulate surface infiltration and evaporation. Soil moisture accounting is possible if the subsurface flow option is used. In this case, ET from the upper unsaturated zone and lower saturated zone may both be simulated. However, the link with surface infiltration is indirect, affecting infiltration only if the soil becomes completely saturated. Otherwise, infiltration capacity for the Horton or Green-Ampt method is regenerated heuristically, and not as a function of ET. A direct link between regeneration of infiltration capacity and ET is needed. However, the current SWMM can still simulate most LID options for hydrologic source control. This has been shown by Lee (2003) and will be demonstrated in the discussion of SWMM simulation capabilities for infiltration that follow, as well as in a detailed example for Portland, Oregon (Chapter 14).

As mentioned earlier in this report, overland flow planes may be used to simulate any surface with an assumed plane surface and vertical “walls” such as roof top vegetation, vegetated buffers, and infiltration trenches. Redirection of flow from impervious areas to pervious (simulating the effects of downspout disconnection, for instance) has a major impact on the predicted downstream hydrograph. Not only does this affect the peak flow, but also the total flow volume and corresponding pollutant load.

4.8 SWMM LID MODELING NEEDS

A significant feature lacking from the Runoff and Transport Blocks is the ability to simulate infiltration from channels. This means that swales or porous channels in which routing and cross-sectional shape effects may be important cannot yet be easily simulated in SWMM. This capability is lacking primarily because of lack of information on the effect of water depth on infiltration rates and because of the need for a heuristic method to simulate the effects of sedimentation and clogging of pores and possible periodic maintenance.

Apart from processes that truly are first-order, sedimentation in storage or flow devices is the only fundamental treatment process simulated in the model; all other fundamental treatment processes must be mimicked through manipulation of removal equations. Biological processes, chemical and physical processes that occur in wetland, bioswales, and riparian zones can only be simulated to the extent that they may be characterized as defined above.

Infiltration through porous (or permeable) pavement may be performed adequately with current SWMM algorithms for infiltration and groundwater flow (James et al. 2001). This will be explained later in the chapter on porous pavement (Section 12.3).

Finally, sediment transport can barely be simulated in SWMM – only in the Transport Block, and only through the use of simple scour-deposition criteria. Hurdle (2001) reviews options for improving SWMM sediment transport routines, none of which are straightforward. However, for the model to be able to characterize treatment based on solids settling, improvements in the overall ability of SWMM to erode, transport, deposit, and scour sediment need to be provided.

4.9 TIME AND SPACE RESOLUTION ISSUES

The Runoff Block is very stable with regard to size of simulated subcatchments. SWMM can model very small parcels (current research has taken catchment size down to 0.03 ac, but there really is no lower limit); however, the time step needs to be adjusted to be smaller than the retention time (V/Q). For instance, a time step ≤ 1 min might be used, compared to the more typical 5-min value routinely employed. With personal computer power this should not be an issue. Finding rainfall data on a small enough time can be a problem, as most data come at a minimum of 15-min intervals. Fifteen-minute data are much too coarse to simulate micro-scale hydrologic processes, but from the point of view of assessment of the vertical water balance storage options (as opposed to peak flow rates), such data may suffice. This is because the vertical water balance processes that are characteristic of small-scale LID options (ET, infiltration, soil moisture routing) typically occur much more slowly than do overland flow runoff processes. Some of these issues will be clarified through additional experience with application of SWMM continuous simulation to miniature subcatchments (e.g., rain gardens). For a process model like SWMM, the issue of temporal and spatial variability is largely an issue of data availability and the amount of detail desired in the simulation run. In case that rainfall data shorter than 15-minute interval is desired, a data disaggregation procedure is presented in the companion project report (Heaney and Lee 2006).

4.10 EXAMPLES OF SWMM BMP SIMULATION

While some additional SWMM capabilities for BMP simulation will be described while discussing the various BMP options, example SWMM simulations reflecting LID and BMP simulation capabilities will be provided in Chapters 14 and 15 toward the end of this report. This is done in lieu of incorporation of examples into each BMP section since these two examples are lengthy.

5 ALTERNATIVE MODELS AND APPROACHES

For each of the viable BMP/LID alternatives defined in Tables 1-1 and 1-2, modeling concepts and mathematical formulations need to be developed that can be used in applying alternatives to the SWMM model based on the state-of-the-engineering knowledge and information. The modeling parameters, e.g., soil characteristics and antecedent moisture content that reflect local site-specific conditions as well as possible seasonal effects, should be incorporated into the formulations whenever possible, and the modeling limits of each BMP/LID alternative that can be incorporated into SWMM based on present knowledge should be well defined.

Several models were evaluated for their ability to simulate the quantity and quality processes in urban BMP alternatives; a useful review of candidate models is provided by Trepel et al. (2000). For the sake of brevity, only models found to be most applicable in urban areas were evaluated during this study; these are listed in Table 5-1. Hence, a number of models that might be suitable for agricultural BMP analysis are not included (Donigian et al. 1995), nor are models that do not simulate water quality. Also, models that primarily deal with water quality in *receiving* streams are also not considered here, such as WASP (Wool et al. 2001) and HSPF (Bicknell et al. 1997). The point is made later that it would be very useful to provide an easy method to transfer hydrographs and pollutographs from SWMM to models such as WASP and HSPF.

Models reviewed are:

- DMSTA: Dynamic Model for Stormwater Treatment Areas (Walker 1995, Walker and Kadlec 2002)
- MUSIC: Model for Urban Stormwater Improvement Conceptualization (Wong et al. 2002)
- P8: Program for Predicting Polluting Particle Passage through Pits, Puddles, and Ponds (Walker 1990)
- PREWET: Pollutant Removal Estimates for Wetlands (Dortch and Gerald 1995)
- REMM: Riparian Ecosystem Management Model (Inamdar 1998a,b)
- SLAMM: Source Loading and Management Model (Pitt et al. 1999b, Pitt and Voorhees 2000)
- VAFSWM: Virginia Field Scale Wetland Model (Yu et al. 1998)
- WETLAND: Wetland water balance and nutrient dynamics model (Lee et al. 2002)
- WMM: Watershed Management Model (Wayne County, MI 1998)

Components of these models will be discussed in following sections with regards to methods for simulating BMPs. As part of each method-related chapter, current (version 4.4h) SWMM simulation options will also be provided.

Most of the models in Table 5.1 can simulate the performance of a variety of BMPs. In some cases, a model will be discussed primarily within the framework of just one BMP. For example, MUSIC will be discussed primarily in Chapter 7 under the wetlands category, even though some of that discussion will relate to other BMPs as well.

Although WMM has proven very useful for screening analyses, it is a spreadsheet model, not a process model. That is, WMM simulates removal through the use of seasonal or annual removal coefficients, during a static simulation. Hence, WMM procedures will not be discussed in detail in this report. On the other hand, the model and its documentation may well be valuable for data and coefficients.

Table 5.1. Simulation models that can simulate urban BMP performance.

Model	Ponds	Infiltration Trenches	Grass Swales	Extended Detention	Bio-retention and Wetlands	Dry Wells	Filter Strips	Porous Pavement	Other Devices
DMSTA	X				X				
MUSIC	X	X	X		X				
P8	X		X		X		X		
PREWET					X				
REMM							X		
SLAMM	X	X	X				X	X	X
VAFSWM					X				
WETLAND					X				
WMM	X		X	X	X				

6 PONDS

6.1 INTRODUCTION

Modeling ponds as a BMP involves storage and release of excess stormwater capture volumes based on hydraulic controls (Urbonas and Stahre 1993). Ponds may be classified in three ways:

1. Wet retention or "wet pond," in which there is a continuous pool of water that will fluctuate up and down during storm events.
2. Dry detention or "dry pond," which fills during a storm event and drains soon after. Dry detention areas are often designed primarily for flood control and usually serve multiple purposes, such as recreation.
3. Extended dry detention is essentially the same as a dry pond except that it is deliberately designed to drain more slowly, thus providing more of a water quality benefit.

The word "retention" implies that some water retained in a pond and between storms is subject only to the vertical water balance of ET and infiltration. The word "detention" implies that water is only detained, and storage is temporary. Wet ponds will typically include detention storage above the permanent pool for purposes of flood control and additional water quality benefits.

Although this chapter is entitled "ponds," the principles described apply to most types of storage devices, that is, wet-weather control devices that provide storage, including wetlands, overland flow and flow in swales, concrete tanks (as for combined sewer overflow control), in-system storage in pipes or channels, and any control that may enhance sedimentation.

Many processes are responsible for the pollutant removals observed in retention and detention ponds. Physical sedimentation is the most significant removal mechanism (Pitt and Voorhees 2000) and is traditionally modeled based on the hydraulic overflow rate (described below, Metcalf and Eddy 2003). However, biological and chemical processes can also contribute important pollutant reductions. The use of aquatic plants, in a controlled manner, can provide still more pollutant removal. Wet ponds also are suitable for enhancement with chemical and advanced physical processes. Infiltration may or may not be an issue with ponds; often a liner or high groundwater table is required to maintain a permanent pool (hence, infiltration is discussed in the Infiltration Trenches section of this report.)

SWMM currently models storage devices quite well, simulating settling and first–order decay directly, with S/T removal equations for other processes. When searching for improvements to SWMM for pond simulation one issue is biological treatment based on second-order reactions. This would be useful if trying to model a type of activated sludge process or flocculation within a pool, although this may be a minor issue. Mineral suspensions and primary sedimentation are best characterized as discrete-particle sedimentation and are often sufficient for characterizing water quality (Tchobanoglous and Schroeder 1985).

The models investigated have few methods for simulating biological treatment directly. Of the models investigated, only P8 has the limited ability to simulate second-order rates of reaction directly. P8 may also calibrate pond performance with a "particle scale removal factor" that may be used to emulate pond processes indirectly.

Simulation of ponds as a CFSTR is a two-part process. At each time step, flow continuity is maintain, in a variant of the lumped continuity equation,

$$\frac{dV}{dt} = Q_i - Q + A(P - ET - G) \tag{6-1}$$

where

V = pond volume,
Q_i = summation of surface inflows,
Q = pond outflow to surface, e.g., through outflow structures,
A = pond surface area,
P = precipitation on pond,
ET = evapotranspiration from pond, and
G = percolation or infiltration into soil beneath the pond (could be negative).

This same equation applies to all surface BMPs, including wetlands, swales, etc. The change in volume term, dV/dt, appears in the CFSTR kinetic equation 4-9 and is an important component of any numerical solution. Variations of Equation 4-9 will be seen in presentations of the several pond (Chapter 6) and wetland (Chapter 7) models that follow.

6.2 SIMULATION OF PONDS WITH P8

6.2.1 The Model

The Program for Predicting Polluting Particle Passage through Pitts, Puddles and Ponds, or P8, is used to model generation and transport of stormwater runoff pollutants in an urban setting (Walker 1990). Calculations are performed on continuous water-balances and mass-balances. Primary applications are for evaluating site plans for compliance, with treatment objectives expressed in terms of removal efficiency for TSS, and BMP design to achieve treatment objectives. Secondary (and less accurate) predictions from this model are runoff quality, loads, violation frequencies, water quality impacts due to proposed development and generating loads for driving receiving water quality models (Walker 1990).

6.2.2 Second-Order Reactions

A fundamentally different approach to simulating contaminant behavior and partitioning in devices currently under investigation is to assign each contaminant to a separate particle class and use second-order decay kinetics instead of first-order settling. This would reduce removal rates as concentrations decreased. Second-order kinetics are consistent with removal mechanisms involving particle interactions (e.g., flocculation) as opposed to discrete settling. The applicability of second-order kinetics has been demonstrated for hydrocarbons in Nationwide Urban Runoff Program (NURP) settling column tests (EPA. 1983, Volume II), phosphorous removal in reservoirs (Walker 1985) and TSS, phosphorous and zinc removal in settling columns (Walker 1990). The user is required to input decay coefficients, which can make the model more flexible in modeling regional performance differences in devices at the expense of the extra data requirements. But the limits of usefulness in using the second-order decay stem from the lack of parameter estimates for such coefficients currently available in the model and in the literature.

In P8 the rates of reaction are used to calculate device (pond) concentrations. Each device is assumed completely mixed for computing concentrations and outflow loads. This is essentially the same way that SWMM currently calculates concentration in its Runoff and Transport Blocks except for the integration of the second-order rate constant, K_2 (not to be confused with the reaeration coefficient), and inclusion of the particle scale removal factor, f.

Device mass balances are calculated as follows (definition of symbols follows the four equations):

$$\frac{dM}{dt} = \frac{dVC}{dt} = W - D \bullet M \tag{6-2}$$

The right hand side of Equation 6-2 is the following variant of Equation 4-9 that includes first and second-order decay, plus settling:

$$D = \frac{Q}{V} + f\,k + f\,K_2\,C_m + f\,v_s\,\frac{A}{V} \tag{6-3}$$

Assuming a constant volume, and average values over a time interval, Δt, the analytical solution for mass M is:

$$M_2 = \frac{W}{D} + (M_1 - \frac{W}{D})\,e^{-D\Delta t} \quad \text{if } D > 0 \tag{6-4}$$

$$M_2 = M_1 + W\Delta t \qquad \text{if } D = 0 \tag{6-5}$$

where

D	= sum of loss terms (1/hr),
C_m	= average concentration during step (mg/L),
V	= average device volume during time step (ac-ft),
M = C·V	= mass in device, (ac-ft·mg/L), with subscripts 1 and 2 indicating beginning and end of time step, respectively,
Δt	= time step length (hours),
W	= total inflow load to device (ac-ft·mg/L/hr),
Q	= average outflow from device, from flow balance (ac-ft/hr),
v_s	= particle settling velocity (ft/hr),
A	= average device surface area during time step (acres),
k	= first-order decay coefficient (1/hr),
K_2	= second-order decay coefficient (1/hr-mg/L), and
f	= device-specific particle removal scale factor.

Notice that the analytical solution is not “exact” since an average concentration, C_m, is used to evaluate the nonlinear second-order decay term. The addition of a user-defined second-order rate equation can be used to simulate flocculation or biochemical reactions, but this is limited by a lack of defined second-order rate constants in the literature. No default decay coefficients are provided in P8, and if desired must be supplied by the user. Most typical K_2 values available are for activated sludge processes and have specified ranges of usefulness based on temperature and mixed liquor suspended solid concentrations (Tchobanoglous and Schroeder 1985). These parameters are difficult to control in BMPs, and therefore limit this modeling option's usefulness.

6.2.3 Particle Removal Scale Factor

Another P8 approach is to use the particle removal scale factor, f, which allows for the easy calibration of an increase or decrease in pond removal efficiency. The f-values adjust the sum of removal rates for each device, and are usually set to 1.0. The f-values can be used to account for effects of vegetation or other factors that affect particle removal, e.g., macrophytes can increase particle removal by increasing surface area, stabilizing bottom sediments, and through biological mechanisms. Removal efficiency curves developed in Australian ponds with macrophytes (Phillips and Goyen 1987, Lawrence 1986, as presented by Walker 1990) correspond to removal scale factors of 2-3 for suspended solids and 3-4 for total phosphorous attributed to macrophyte presence in wet detention ponds. Alternatively, a removal scale factor f < 1 can account for short circuiting or other poor hydraulic designs. See Section 6.5 for an alternative heuristic approach leading to similar results.

6.2.4 Pollutant Removal

Pollutant removal under dynamic conditions occurs when particle settling velocities exceed the basin overflow rate. Removing solids will also remove much of the pollutants of interest. Notable exceptions of potential concern include dissolved forms such as nitrates, chlorides, soluble zinc, pathogens, 1,3-dichlorobenzene, fluoranthene, and pyrene (Pitt and Voorhees 2000). The P8 model uses the traditional hydraulic loading rate method for dynamic settling (e.g., Fair et al. 1968, Minton 2002, Metcalf and Eddy 2003) for all devices modeled (ponds, swales, bioretention facilities, filter strips), in which particles with settling velocities greater than the hydraulic loading rate (or sometimes, the "overflow rate") are removed. That is, particle removal occurs when

$$v_s > q = \frac{Q_i}{A} \tag{6-6}$$

where
v_s = settling velocity,
q = overflow rate or hydraulic loading rate, flow rate/area or length/time,
Q_i = inflow, and
A = surface area.

The method is used to determine removal in all devices modeled by P8 (ponds, grass swales, bioretention facilities and filter strips). In some applications, the outflow is used instead of the inflow to determine the overflow rate (Sections 6.3 and 6.5).

Within P8, the inflow is computed using a mean storm intensity and watershed area, so that:

$$q = \frac{A_w \, R_v \, i}{12 \, A} \tag{6-7}$$

where
q = average overflow rate (ft/hr),
A = pond surface area (acres),
A_w = watershed area (acres),
R_v = watershed runoff coefficient (runoff volume/rainfall volume),
i = mean storm intensity (in/hr); ~ 0.06 in/hr is used as default in P8, and
12 = conversion factor from inches to feet.

An assumed particle size distribution (Table 6-1) is then used to determine the amount of settling that occurs. The first class represents the dissolved (non-settling) fraction of water quality constituents. The remaining classes are based on NURP settling velocity distributions. Ideally, these would be supplied by the user with site-specific stormwater *treatability* data, but such data are usually sadly lacking.

Particle fractions (mg/kg), sometimes called "potency factors," are used to translate particle concentrations of total suspended solids (TSS) into associated pollutant values. These fractions are similar to SWMM's constituent fractions in the J4 data group of the Runoff Block and are one way to simulate adsorption. That is, a "particle fraction" or "potency factor" or "constituent fraction" is related to the partition coefficient, K_d, used in sorption kinetics through Equation 4-16 (Chapra 1997). These particle fractions have been calibrated in P8 to "typical urban runoff" so that the median SS EMC corresponds to the values reported by NURP, based primarily on runoff concentrations and settling velocity distributions (USEPA 1983, Walker 1990).

Table 6-1. Particle class default values in P8 (Walker 1990).

Class	Description	% of TSS	Settling Velocity (ft/hr)
P0%	Dissolved	0	0
P10%	10th Percentile	20	0.03
P30%	30th Percentile	20	0.3
P50%	50th Percentile	20	1.5
P80%	80th Percentile	40	15

Buildup-washoff parameters have been calibrated for both pervious and impervious areas to produce an EMC of 100 mg/L TSS for a median site, and 300 mg/L for a 90th percentile site as per NURP. This method is independent of stormwater volumes and ignores any variation in concentration (first-flush effects) with large storm events and due to possible construction site runoff, which can yield much higher TSS concentrations.

Particle compositions (mg/kg) are then used to translate particle concentrations into concentrations of total suspended solids (TSS), total Kjeldahl nitrogen (TKN), total phosphorous, copper, lead, zinc and hydrocarbons. These compositions have also been calibrated so that median, event-mean runoff concentrations correspond to values reported by NURP (USEPA 1983) as listed in Table 6-2.

This calibration is based on a simulation of 1983-1987 Providence Airport rainfall. High site-to-site variability is reflected in the 2 to 3-fold differences between the median and 90th percentile sites, and implies considerable uncertainty in predicting actual contaminant concentrations. Calibration with local or regional runoff data will help to reduce this uncertainty (Walker 1990).

Table 6-2. Example of P8 calibrated runoff concentrations (Walker 1990).

Component	**Median, EMC, mg/L**		
	NURP Median Site	**90th % site**	**% Dissolved**
total suspended solids	100	300	0
total phosphorous	0.33	0.7	30
total Kjeldahl nitrogen	1.5	3.3	40
total copper	0.034	0.093	40
total lead	.020 (a)	0.05 (a)	10
total zinc	0.16	0.5	40
hydrocarbons	2.5 (b)	5.0 (b)	10
a - NURP lead values reduced to account for >10-fold reduction in gasoline lead content			
b- Hydrocarbons estimate from load factors reported by Hoffman (1985)			

6.3 SIMULATION OF PONDS WITH SLAMM

SLAMM (Pitt and Voorhees 2000) calculates particle deposition in wet detention ponds using the same hydraulic loading rate methodology just described for P8, although SLAMM's authors refer to it as the "upflow velocity method" (Linsley and Franzini 1964). It is the same as hydraulic loading rate except that Pitt and Voorhees (2000) define it as the ratio of outflow rate to surface area. This is reasonable since pond outflow rates generally govern the time required to drain the dry pond or return a wet pond to its permanent pool level. Hydrograph routing through the pond is first performed using the storage-indication method (see also Section 6.5) as implemented in the RESVOR reservoir routing subroutine of the Natural Resources Conservation Service in Technical Releases 20 and 55 (SCS 1986).

SLAMM expands on the storage-indication procedure by calculating incremental upflow velocities (hydraulic loading rates) for each calculation interval. SLAMM automatically determines the most efficient calculation interval. Any particle that has a settling velocity greater than this upflow velocity will be retained in the pond. The user describes a particle size distribution for the inflowing water that SLAMM uses to calculate the particle settling rates from Stokes' law modified for deviations from laminar flow (e.g., Fair et al. 1968).

Stokes' equation (Fair et al. 1968), used to compute the terminal fall velocity, or settling velocity, v_s (ft/s), for a sediment particle in laminar flow is:

$$v_s = \frac{1}{18}(S_p - 1)g\frac{d^2}{\nu} \tag{6-8}$$

where

d = sediment diameter (ft),

S_p = specific gravity of sediment,

ν = kinematic viscosity (function of water temperature), (ft^2/s),

g = acceleration due to gravity (ft/s^2).

Limitations of Stokes' law are discussed below, in Section 6.5. SLAMM calculates the critical particle sizes retained in each calculation interval and sums the retained particles for the complete event. Hydraulic performance of an outfall pond is also summarized by giving the peak flow rate reduction factor and the pond flushing ratio (ratio of incoming runoff volume to normal pond volume) for each event. Peak flow reduction affects downstream impacts, a BMP evaluation criterion (Section 3.1).

6.4 POND SIMULATION WITH MUSIC

This model simulates sedimentation using methods similar to those described for SWMM in the next section. Since a primary application of MUSIC is to simulate the treatment obtained by storage in wetlands, discussion of the MUSIC algorithms will be deferred to Chapter 7.

6.5 POND SIMULATION WITH SWMM

Hydraulic and water quality procedures for simulation of "storage" in SWMM are explained in considerable detail in the documentation (Huber and Dickinson 1988, James et al. 2002a,b) as well as in texts such as Nix (1994), Stephan Nix being the primary author of the SWMM Storage/Treatment Block. Flow routing through storage is performed using the storage-indication (S-I, also known as the Modified Puls) method (McCuen 1982, Huber and Dickinson 1988, Bedient and Huber 2002). The S-I method is simply a convenient numerical scheme for solution of the combined continuity and reservoir outflow equations for the two unknowns, storage and outflow. As shown in the detailed SWMM example of Chapter 15, required data include surface area vs. depth (from which volume vs. depth may be computed) and an outflow rating curve (flow vs. depth). The latter may be in the form of a table, pump curve, or power equation (e.g., for a weir or orifice). The S-I scheme is also used in the Transport Block for flow routing through storage, whereas storage in the Runoff Block is evaluated by the mathematical solution of the nonlinear reservoir equation, and in the Extran Block, by solution of the Saint-Venant equations applied to the storage device (Roesner et al. 1988). Since the S/T Block (and sometimes the Transport Block) is the primary module used for simulation of quality in storage devices, focus will be upon its application for ponds.

As described in Sections 4.2 and 4.3, quality transformation may be simulated in three ways:
1. first-order decay with either plug flow or complete mixing,
2. a general "all-purpose" removal equation (Equation 4-1), or
3. sedimentation theory, for use with plug flow.

Considering the third option briefly, sedimentation is obviously a function of settling velocity. While Stokes' equation 6-8 could be used, more accurate is to compute settling velocity as a function of turbulence level, as indicated by the particle Reynolds number, **R,**

$$\mathbf{R} = \frac{v_s d}{\nu} \tag{6-9}$$

where
v_s = settling velocity (ft/s),
d = particle diameter (ft), and
υ = kinematic viscosity (ft^2/s).

Settling velocity is then defined by the balance between gravity and drag force,

$$v_s = \sqrt{\frac{4gd}{3C_D}(S_p - 1)} \tag{6-10}$$

where
g = gravitational acceleration, e.g., 32.2 ft/s^2,
C_D = drag coefficient, a function of **R**, and
S_p = particle specific gravity.

The iterative process to determine v_s is explained by Fair et al. (1968) and Minton (2002) and was implemented by Sonnen (1977) in a past version of SWMM. The current S/T Block uses Sonnen's method (Huber and Dickinson 1988). This is in lieu of settling velocities entered directly by the user, from a stormwater treatability analysis. Stokes' law is valid for **R** less than approximately 0.5, for which $C_D = 24/\mathbf{R}$, and for which Equation 6-10 reduces to Equation 6-8.

Two options exist for pond (or any storage) simulation in SWMM: plug flow or complete mixing (CFSTR, discussed in Section 4.3). Most ponds behave in an intermediate fashion between plug flow and complete mixing due to short-circuiting, dead zones, and incomplete mixing (Thackston et al. 1987). There are also two- and three-dimensional effects, including diffusion and dispersion that SWMM cannot simulate. One common heuristic solution is to use Fair and Geyer's (1954) "tanks in series" (TIS) method for analysis of imperfect sedimentation basins for water and wastewater treatment, in which the fraction captured of particles that have settling velocity v_s is:

$$R = 1 - \left(1 + \frac{v_s}{NQ/A}\right)^{-N} \tag{6-11}$$

where

R = 1 - C_{out}/C_{in} = fraction captured or retained in pond,

Q = pond outflow rate,

A = pond surface area, and

N = empirical measure of hydraulic efficiency, or number of CFSTRs in series.

The hydraulic efficiency factor, N, can reflect short-circuiting, for example, and ranges from 1 for "very poor performance," to 3 for "good performance," to 5 or higher for "very good performance" (Fair et al. 1968, Driscoll 1986b, Pitt and Voorhees 2000, Minton 2002). Another interpretation of N is the number of CFSTRs or "tanks" in series, as an approximation leading up to plug flow (Levenspiel 1972, Kadlec and Knight 1996, Chapra 1997). That is, Equation 6-11 is the removal efficiency for N CFSTRs in series, all with the same flow rate, and each with 1/N of the total pond area A_b. (The ratio, v_sA/Q can be expressed in several related forms, as will be discussed below in relation to Equation 6-14.) When N = 1, the equation gives the steady-state performance of one CFSTR (solution of Equation 4-3 for dC/dt = 0). The case $N \rightarrow \infty$ corresponds to perfect horizontal plug flow representation for a tank or pond (Chapra 1997),

$$\lim_{N \to \infty}(\text{Eqn.}6-11) = 1 - e^{-\frac{v_s}{Q/A}} \tag{6-12}$$

The removal efficiency, R (fraction), due to quiescent settling (e.g., in the permanent pool of a wet pond), is given by

$$R = \frac{v_s t_d}{h} \le 1 \tag{6-13}$$

where

t_d = detention time, and

h = pond depth.

All particles with settling velocity v_s will be removed as long as the storm inter-event time is greater than the detention time (neglecting resuspension and several other side effects). Note that Equation 6-13 is a special case of Equation 6-11 with N = -1.

The effect of N is shown in Figure 6-1. Several options for the dimensionless abscissa are shown that represent equivalent design conditions, depending on the nature of the removal process being simulated, including

$$\frac{v_s}{Q/A} = \frac{v_s t_d}{h} = k\,t_d = \frac{kV}{Q} = \frac{k'}{q} \tag{6-14}$$

where

k = first-order decay coefficient (1/time),

k' = k·h = rate constant (depth/time),

q = hydraulic overflow rate (depth/time), and

V = volume.

Various relationships are implied among the parameters, including the fact that settling behaves as a first-order removal process if $k = v_s/h$. Kadlec and Knight (1996) define the *Damköhler number*, D_a, as

$$D_a \equiv k'/q \tag{6-15}$$

Thus, the various dimensionless forms shown in Equation 6-14 are variations of the Damköhler number.

Regarding Figure 6-1, so-called "very poor performance" corresponds to complete mixing in a pond, or N = 1 CFSTR. This means that some entering particles can move "instantaneously" to the outlet, as in short-circuiting. The other extreme is plug flow, in which particles settle while moving on one continuous, horizontal path to the outlet, thereby maximizing their opportunity for removal. The range of perfect mixing to plug flow corresponds to dynamic settling, that is, settling while outflow from the pond is occurring. Quiescent settling corresponds to still water and no outflow (although this may be assumed for discrete plugs, as indicated below). Given enough time, and neglecting turbulence and resuspension due to wind, all particles for which $v_s \cdot h/t_d \geq 1$ will settle out.

SWMM S/T currently simulates the extremes of complete mixing and plug flow. For plug flow, an enhancement is used to account for non-ideal settling conditions that were characterized by Camp (1946) in the form of sediment trap efficiency (removal fraction) as a function of a turbulence factor. (This is the "sedimentation theory" or third S/T option listed earlier.) The procedure was simplified by Chen (1975), and its SWMM implementation is described in Appendix IV of the User's Manual (Huber and Dickinson 1988). In essence, removal in plugs corresponds to ideal quiescent conditions (Equation 6-13) when the turbulence factor is low, and follows a reduced efficiency curve given by Chen (1975) for higher levels of turbulence. However, this still assumes plug flow conditions. A possible enhancement to the simulated removal efficiency would be to use Equation 6-11 to represent the reduction of efficiency as plug flow conditions "deteriorate" to those of complete mixing, with its inherent short circuiting. The user would need to supply the value of N or make an equivalent judgment regarding "very good" to "very poor" performance, as defined for unit operations by Fair et al. (1968) and for ponds by Pitt and Voorhees (2000) and Minton (2002). Additional guidance is provided in Section 7.7.

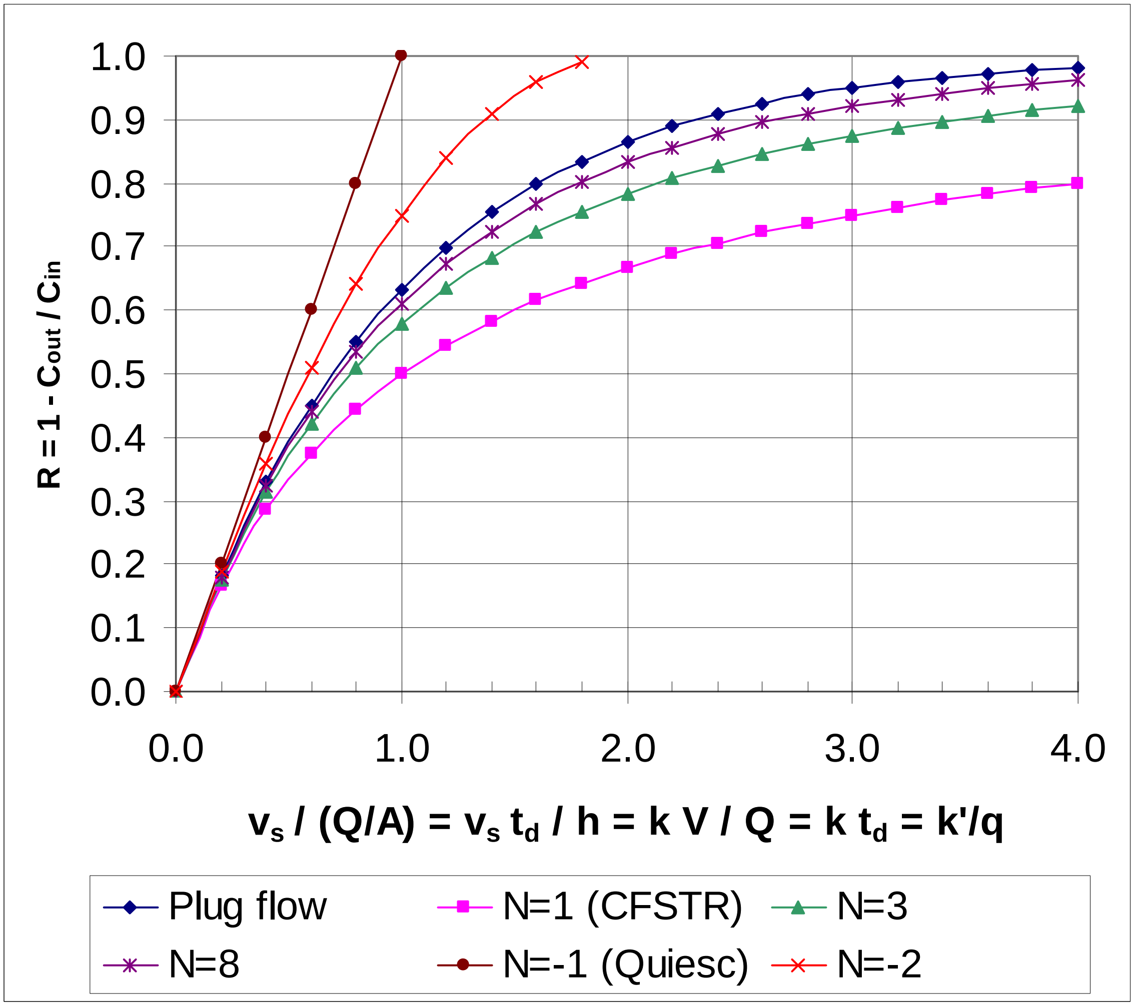

Figure 6-1. Removal efficiency, R, as a function of dimensionless forms of rate of treatment or loading.

The particle scale removal factor, f, in the P8 model serves the same purpose as the hydraulic efficiency parameter, N. Both may be used in a highly empirical fashion to account for increases or decreases in removal efficiency, with plug flow as a starting point (Equation 6-12). For instance, values of $-1 \geq N \geq -\infty$ represent still another set of curves on Figure 6-1 for which removal efficiency is improved to a value between plug flow and quiescent settling (efficiencies for N = -2 are shown on the figure). Hence, use of Equation 6-11 for plug flow in SWMM might be a very versatile way to represent performance, albeit still empirical and of a curve-fitting nature.

6.6 EXTENDED DETENTION

In order to enhance sedimentation, extended detention basins are designed to empty their brim-full volume in 24 to 48 hours, with no more than 50% of this volume being released during the first one-quarter to one-third of the emptying period (Urbonas and Stahre 1993). Of course, local regulations may modify these timing requirements, but these are typical of today's practice. Water quality processes in extended detention (and detention) ponds obey the same principles as just discussed, including dynamic settling. However, it is important to remember that there is no "upflow" velocity (at least not after possible spillway flow has ceased); the "overflow rate" is due to water draining horizontally or downward toward a drain that is usually on or near the pond bottom. In essence, this is the procedure used in SWMM S/T for plug flow with sedimentation. Water is assumed to move horizontally toward the outlet, with pond depths gradually decreasing. Removal is always a function of detention time and may be modified according to Camp's (1946) and Chen's (1975) adaptations for turbulence.

Of special interest is the behavior of dry detention and extended detention basins over the long term of hydrologic events, during which a detention area will fill and empty according to the arrival of storms and duration of inter-event dry periods. For instance, if another storm arrives before the detention area has drained completely, the incoming flow will mix with the remaining water, and the removal analysis must begin anew. The EPA probabilistic analysis (Driscoll 1986b, Urbonas and Stahre 1993), later enhanced and expanded by Adams and Papa (2000), provides a statistical methodology for analyzing detention performance by coupling rainfall and runoff statistics, e.g., storm duration and inter-event times, determined by local meteorology, with detention drainage characteristics, determined by the hydraulic design of the outlets. SWMM and other continuous simulation models provide the same type of analysis by using long-term historic rainfall time series as drivers for the stormwater runoff that enters (and drains) from detention and retention ponds. Hence, the complex interaction between storm events and filling and drying of detention areas is analyzed empirically, through continuous simulation, with a statistical analysis (e.g., using SWMM's Statistics Block) of the results.

7 WETLANDS AND BIORETENTION FACILITIES

7.1 INTRODUCTION

Bioretention facilities include constructed wetlands, wetland basins, bioswales, and wetland channels. Bioretention is a combination of processes served by other BMP types, and the processes that might enhance SWMM's ability to model bioretention facilities are a combination of those previously listed. Sedimentation resulting from storage is a fundamental unit process that occurs in these facilities. Infiltration, filtration, flocculation, biochemical interaction, increased settling and decreased erosion due to the presence of macrophytes also occur and have been discussed in conjunction with Tables 1-2 and 1-3. A highly comprehensive analysis of the use of wetlands for stormwater treatment is provided by Kadlec and Knight (1996).

Pollutants in stormwater can be removed by wetlands and ponds through a combination of 1) incorporation into or attachment to sediments or biota, 2) degradation, and 3) export to the atmosphere or groundwater (Strecker et al. 1992). The wetland *hydroperiod* (the seasonal pattern of water levels) defines the rise and fall of surface and subsurface water that in some cases can lead to export of pollutants to groundwater. In general, removal mechanisms are known to be physical, chemical and biological in nature. Some important removal mechanisms include sedimentation, filtration, oxidation, adsorption, volatilization, precipitation, nitrification and microbial decomposition. Removal mechanisms and which pollutants they affect (Horner 1995) are shown in Table 7-1.

The mechanisms shown in Table 7-1 are not assumed to be independent from one another. Sedimentation due to reduced flow velocities, adsorption, and filtration by vegetation are three of the major removal mechanisms in many wetlands. However, it should be noted that any one of the eight mechanisms of Table 7-1 can be dominant depending on the wetland's characteristics (e.g., as influenced by its hydrology and hydraulics). Strecker et al. (1992) note this as a major reason why wetlands differ so greatly in their pollutant removal efficiencies. Yu et al. (1998) also note differences in removal efficiencies attributable to design parameters such as inlet and outlet configuration, length to width ratio, and consequent residence times. Yu et al. (1998) reported greatest removal for sites that maximized the length to width ratio. These factors obviously play an important role when designing a wetland or pond system. Several options exist to an engineer who is interested in designing a wetland/pond that will maximally remove pollutants from urban stormwater. These include increasing the hydraulic residence time (HRT), providing an environment that encourages flow at a low level of turbulence so sedimentation can be maximized, propagating fine, dense, and herbaceous plants, and establishing the system on a medium-fine textured soil (Horner 1995, Kadlec and Knight 1996).

Although a two or three-dimensional model could be used for simulation of surface flows through wetlands, in all cases evaluated in this report, the lumped storage approach of Equation 6-1 has been used for water volumes. Constituent kinetics are simulated through assumption of a CFSTR, through variations on Equation 4-9, as for ponds.

Table 7-1 Wetlands/ponds pollutant removal mechanisms. (After Horner 1995.)

Mechanism	Pollutants Affected	Promoted By
Physical Sedimentation	Solids, BOD, pathogens, COD, P, N, metals	Low turbulence
Filtration	Solids, BOD, pathogens, COD, P, N, metals	Fine, dense herbaceous plants. Outflow through porous media also has an obvious filtration effect.
Chemical Adsorption	Dissolved P, metals, synthetic organics	High soil Al, Fe; high soil organics, circum-neutral pH
Oxidation	COD, petroleum, hydrocarbons, synthetic organics	Aerobic conditions
Volatilization	Volatile petroleum hydrocarbons and synthetic organics	High temperature and air movement
Precipitation	Dissolved P, metals	High alkalinity
Biological Nitrification	NH_3-N	Dissolved oxygen >2 mg/L, Low toxics, temps. >5-7 ˚C Circum-neutral pH
Microbial Decomposition	BOD, COD, petroleum hydrocarbons, synthetic organics	High plant surface area and soil organics

7.2 DIFFICULTIES WITH MODELING MULTIPLE PROCESSES

The difficulty in modeling multiple processes comes from choosing the order in which each process occurs, or trying to model processes concurrently. Some pollutants increase during some processes and decrease during others (e.g., BOD in activated sludge vs. sedimentation). The order in which these processes are modeled will affect the estimated effluent concentrations as well as the removal efficiency in each process. Coupling standard removal efficiencies for multiple processes may also overestimate pollutant removal. Many BMPs cannot remove pollutants below a certain level. Some BMPs output consistent effluent quality that is not strictly dependant on influent concentrations (Strecker et al. 2001). Two simple methods are described in the next sub-section, followed by descriptions of three more comprehensive models for wetlands treatment.

7.3 BIORETENTION IN WMM AND P8

One simple method for performance simulation is to use overall device efficiency. For example, WMM (not a process model) uses an assumed efficiency for removal that is only suitable for annual load reduction predictions rather than for individual events. P8's use of a particle removal scale factor calibrated for simulation of the bioprocesses that occur during bio-filtration is quite simple. The presence of macrophytes can increase the removal factor (usually f=1) to 2 or 3, as discussed in the Section 6.2.

7.4 SIMULATION OF WETLANDS WITH THE WETLAND MODEL

7.4.1 Introduction

One of the newer models available, WETLAND has been designed to model constructed wetlands and can be adapted to model natural wetlands as well. WETLAND is a dynamic, compartmentalized simulation model (Lee et al. 2002) and was designed as a continuous-flow, stirred-tank reactor (CFSTR), so complete mixing is assumed to occur. Two input forms are accepted for this model. First, daily values for hydrologic parameters and nutrients can be input, in either user-defined data or from output from a nonpoint source model (e.g., SWMM hydrographs and pollutographs). Secondly, data can be input based on the SCS curve number method (McCuen 1982, SCS 1986). The SCS method determines the amount of daily runoff from the watershed, which is then multiplied by an EMC for each respective nutrient parameter to determine nutrient inflow to the wetland system (Lee et al. 2002).

Written in Fortran 77, WETLAND models both free-water surface (FWS) and subsurface flow (SSF) wetlands, and is designed in a modular manner that gives the user the flexibility to decide which cycles and processes to model (Lee et al. 2002). This model has one main program that calls upon and manages the sub-models and options that need to be simulated (Lee et al. 2002). The relationship between WETLAND's main code and its sub-models is shown in Figure 7-1 (Lee et al. 2002).

7.4.2 WETLAND Cycles and Sub-models

Within the model, there are many wetland cycles that can be modeled. These include the hydrologic, nitrogen, carbon, dissolved oxygen (DO), bacteria, vegetative, phosphorous and sediment cycles. In the hydrologic sub-model, WETLAND uses a vertical water balance to account for surface storage. Treating the wetland as a storage unit, the spatially-lumped continuity equation (Equation 6-1) used is (Lee et al. 2002) in the following form:

$$dV/dt = Q_c + Q_p - Q + dV_b/dt + dV_p/dt + (P - PI - ET) \cdot A \qquad (7\text{-}1)$$

where:
dV/dt = change in surface storage (m^3/day),
Q_c = watershed catchment runoff additions (m^3/day),
Q_p = additions from point sources (m^3/day),
Q = daily outflow rate (m^3/day),
dV_b/dt = change in living biomass water volume in the surface water (m^3/day),
dV_p/dt = change in standing dead plant water volume in the surface water (m^3/day),
P = daily precipitation rate (m/day),
PI = percolation/infiltration rate (m/day),
ET = evapotranspiration rate (m/day), and
A = wetland surface area (m^2).

Evaporation may be computed from pan data – the primary model option. ET may also be modeled using Thornthwaite's method (Dingman 2002). An hourly time step is used for the hydrology sub-model.

The vegetative sub-model simulates biomass growth and death rates. At the beginning of the growing season, biomass growth rate is assumed to increase linearly for up to 20 days until a maximum rate for the growing season is obtained (Lee et al. 2002). After 20 days, the biomass growth rate remains constant

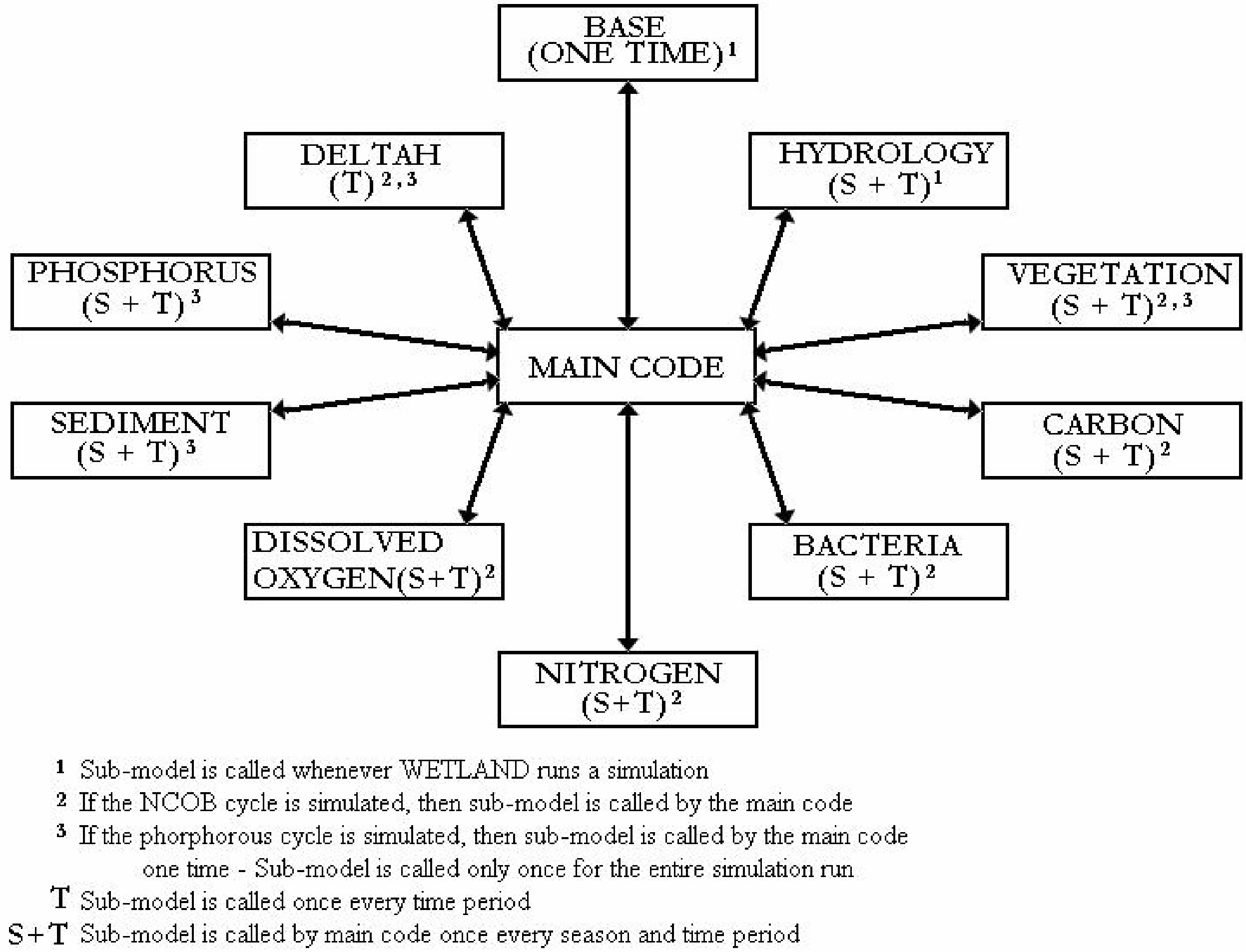

Figure 7-1. Relationship between the MAIN CODE and respective sub-models for an entire simulation run (Lee et al. 2002).

until the end of the growing season, when the growth rate decreases linearly to zero over a period of 10 days.

Unlike most wetland models, WETLAND explicitly accounts for the effects of biomass and microbial dynamics in a wetland system using the dynamically-linked NCOB (nitrogen, carbon, DO, and bacteria) cycle (Lee et al. 2002). Carbon/nitrogen ratios are accounted for in this model. WETLAND is unique in that nitrification and denitrification are modeled using Monod kinetics, not just empirical relationships.

There are five different state variables in the carbon (denoted as "C") sub-model. They are biomass C, standing dead C, particulate organic C (POC), dissolved organic C (DOC), and refractory C (Lee et al. 2002). The vegetative sub-model is connected to the standing dead C and the biomass C, respectively. Biomass C is determined by multiplying a biomass C concentration by the existing biomass in the wetland. Standing dead C is determined similarly; however, physical degradation and DOC leaching also are taken into account. A mass balance for POC is constructed for FWS wetlands. This depends on particulate BOD influx, microbial death, peat accumulation, and POC mineralization. The mass balance for DOC depends on soluble BOD influx, DOC mineralization, DOC leaching, and diffusion (Lee et al. 2002).

The nitrogen sub-model simulates processes such as ammonification, denitrification, immobilization of nitrogen, and peat accumulation. Inclusion of NH_3 volatilization, atmospheric deposition and nitrogen fixation in the modeling of the overall nitrogen cycle is optional (Lee et al. 2002). Variables included in the nitrogen sub-model are dissolved organic nitrogen (DON), particulate organic nitrogen (PON), ammonia (NH_3), ammonium (NH_4^+), nitrate (NO_3^-), immobilized N and refractory N. DON influent may enter a wetland from point sources, direct catchment runoff, atmospheric deposition and percolation. The physical degradation of decaying plant mass can add to the accumulation of DON. PON modeling is similar to DON, except that PON is also assumed to accumulate in peat as refractory N (Lee et al. 2002). Nitrogen fixation is also an available option for modeling as a way to increase DON and is modeled with a zero-order equation. NH_4^+ enters a wetland from catchment runoff, seepage, atmospheric deposition and point sources. Immobilized N is the sum of DON and PON immobilization, NO_3^- uptake and NH_4^+ uptake. Increases in nitrate in wetland waters come from influent and nitrification, whereas decreases come from plant uptake and denitrification.

In the dissolved oxygen (DO) sub-model, oxygen is assumed to be added to the wetland by point sources, incoming streamflow, precipitation, reaeration from the atmosphere, and biomass flux. Dissolved oxygen is the only state variable in the DO sub-model. Oxygen is assumed to be passed from vegetation to the wetland bottom at a constant rate during the growing season.

The bacteria sub-model describes the microbial interactions within the wetland and includes all of the microbial activity of the model. Both autotrophic and heterotrophic bacteria are modeled in WETLAND. pH is not modeled because wetlands are known to drive pH towards neutrality (pH = 7). The growth rate of bacteria is modeled using Monod kinetics (Chapra 1997). Heterotrophic bacteria are modeled using Monod kinetics where growth is dependent on TOC.

Sedimentation is modeled in the sediment sub-model of WETLAND. There are five different sediment classifications in this sub-model. They are inflow, outflow, deposition, resuspension and decomposition (Lee et al. 2002). Sediment inflow depends on the option chosen, while outflow is a function of the resuspension, settling velocity, and total amount suspended in the water. A first-order user-defined rate equation is used to model decomposition of sediment.

The remaining sub-model is the phosphorous sub-model and is based on the assumption that all of the suspended sediment particles provide surface area to which phosphorous can be attached and consequently settled, resuspended, or transformed (Lee et al. 2002). Phosphorous coming into the wetland is modeled as a direct input, or by sorption using the Freundlich or linear isotherms (Chapra 1997). Additions of phosphorous from biomass decomposition and mineralization can also be modeled. Input phosphorous concentrations are used to determine the particulate phosphorous concentrations when modeling with Freundlich or linear isotherms. Mineralization is modeled with first-order equations for each particle class (Lee et al. 2002). Settling and resuspension of particulate phosphorous is related to the quantity of sediment particles. The amount of phosphorous from physical degradation is directly related to plant biomass.

7.4.3 WETLAND Output

Time series of concentrations in the effluent (and within the well-mixed wetland) are provided for: NH_4, NO_3, Org-N, DON, PON, DO, BOD5, TSS, DP (dissolved P), and TP (total P). Hydrologic state variables include water depth (implying a water volume) and the outflow hydrograph. In comparisons with measurements, all predicted chemical constituents except DO had a significant correlation with measured data in the example presented by Lee et al. (2002). Wetland effectiveness as a BMP is computed as the reduction of effluent loads relative to influent loads, i.e., percent removal based on loads. The strength of the model is its linked Monod kinetics for the chemical state variables. In this respect it is similar to EPA WASP model capabilities (Wool et al. 2001). Weaknesses include the need for requisite kinetic parameters (similar to the need of most models) and the well-mixed assumption, which does not allow the study of hydraulic short-circuiting and shape effects.

7.5 SIMULATION OF WETLANDS WITH VAFSWM

7.5.1 Introduction

The Virginia Field Scale Wetland Model (Yu et al. 1998) was developed with the help of extensive monitoring of constructed wetlands and detention ponds in eastern Virginia. The model is available through the Virginia Transportation Research Council in Charlottesville and was developed to fill a gap in analytical tools perceived on the basis of the Virginia studies.

7.5.2 VAFSWM Components

The water balance is essentially that of Equation 6-1. Constituents are simulated in three forms: 1) water column suspended solids, 2) water column constituent, for which the particulate form is a fraction of the SS concentration, and 3) sediment/substrate concentration. The substrate consists of sediment and near-surface root zone for the aquatic vegetation. The substrate water volume must account for the porosity of this zone. The model is simpler than WETLAND inasmuch as nutrient kinetics are not linked; each constituent is simulated in each of the three forms in the manner of a CFSTR using a variant of Equation 4-9. Suspended solids simulation includes only settling as a removal mechanism. Water column constituent simulation includes the following removal mechanisms:

- Settling of the particulate fraction (Equation 4-19)
- First-order decay of the dissolved fraction (Equation 4-18) by adsorption to plants and plant uptake
- Adsorption to substrate

Substrate concentrations are affected by:

- Settling from the water column
- Settling within the substrate area (using a different settling velocity)
- Adsorption and plant uptake from the water column

Dissolved and particulate fractions are based on a partition coefficient, K_d (Equations 4-18 and 4-19). The three CFSTR equations are solved simultaneously using a fourth-order Runge-Kutta (RK4) integration scheme. Insufficient data were available for the authors to fully verify the model, but TSS and TP simulations provided removal efficiencies (efficiency ratio, Section 3-4) in the range values observed at the test site.

7.5.3 Implications for SWMM Improvements

The principal contribution of the VAFSWM formulation is explicit inclusion of settling, adsorption, and first-order kinetics for the water column and substrate in a simpler form (i.e., one that does not involved the linked nutrient dynamics of WETLAND). Although there are approximately 15 required input parameters (for simulation of just one constituent in addition to TSS), some guidance is available for parameter estimates, and the overall formulation is amenable to inclusion in SWMM, since the fundamental processes are already included in Equation 4-9 (currently implemented in SWMM 4.4h). The main additional effort would be to include a representation of substrate concentrations, with possible complications in linkages to subsurface flow pathways.

7.6 SIMULATION OF WETLANDS WITH PREWET

7.6.1 Introduction

Another program used to model wetlands is the Pollutant Removal Estimates for Wetlands (PREWET) model, developed at the Waterways Experiment Station of the Army Corps of Engineers. Online help is available to answer some questions about the model, and a brief description is available from the website http://www.wes.army.mil/el/elmodels/index.html#wqmodels. PREWET uses equations and logic that are programmed in C++, and a commercially available graphical user interface (GUI) library, "Zinc," which is also written in C++, is used (Dortch and Gerald 1995).

PREWET contains algorithms to model a wetland's ability to remove contaminants such as TSS, BOD, total nitrogen, total phosphorous, total coliform bacteria and other contaminants. PREWET does not model microbial growth and decay or seasonal/annual processes like vegetation growth and decay. Therefore, this model cannot be used to assess seasonal/annual effects. In fact, because it is a *steady-state* model, its usefulness is limited primarily to help in parameter estimation, as will be seen. Hydrological parameters are input into the model based on knowledge about the wetland.

7.6.2 PREWET Removal Mechanisms

The main assumption made in PRETWET is that the modeled wetland is at steady-state. This means flows and concentrations of pollutants are constant over time. Obviously wetlands are rarely at steady-state. However, average values or long-term values are the goal of PREWET, for which the steady-state assumption is valid. There are two conditions for which this model works. It assumes the wetland is either a CFSTR, or plug-flow reactor (PFR). The mass balance equation PREWET uses for a CFSTR is essentially the same as Equations 4-3 and 6-2 under steady-state conditions,

$$d(VC)/dt = 0 = W - QC - kVC \qquad (7\text{-}2)$$

where:
V = volume of the wetland (volume)
C = pollutant concentration leaving the wetland (mass/volume)
t = time
W = loading of pollutant entering wetland (mass/time)
Q = flow rate (volume/time)
k = first-order biological degradation rate of pollutant (1/time)

Equation 7-2 can be solved for the steady-state concentration (Chapra 1997),

$$C = \frac{W/Q}{1 + kV/Q} \qquad (7\text{-}3)$$

The ratio V/Q is recognized as the hydraulic residence time. Sedimentation is modeled in PREWET by total suspended solids removal (settling velocity) based on a balance among settling, resuspension, and sediment layer burial (Thomann and Mueller 1987). BOD removal occurs through settling of the particulate fraction of BOD from the water column to the sediments, and adsorption to benthic biota (Dortch and Gerald, 1995). These removal processes are combined into a first-order rate constant, k_r.

Total coliform bacteria are also modeled using a first order decay rate. This is due to death, settling, and predation of the bacteria (Dortch and Gerald, 1995). First-order decay rates, such as k_B for bacteria, are adjusted for temperature using the customary Arrhenius formulation, Equation 4-21, with a recommended value for θ of 1.07 for bacteria.

In PREWET, the only phosphorus component modeled is total phosphorous (TP). The model only considers the natural, long-term removal mechanism of sediment burial (Dortch and Gerald 1995). TP retention decreases with time in wetlands due to the sediments becoming saturated with phosphorous. After a period of time, the sediment reaches saturation equilibrium, reducing the rate of phosphorous uptake. A net first-order removal rate is obtained from the coupled water column processes of settling, resuspension, burial, and diffusion from the bed.

Total nitrogen (TN) is the only nitrogen variable considered with this model, and denitrification is the only TN process modeled. A nitrate balance cannot be used to compute the loss of TN because nitrate may also be gained through nitrification (Dortch and Gerald 1995). Instead, TN is estimated through loss of nitrate via denitrification. The steady-state relationship between TN and NO_3-N is used to help determine the TN loss rate from better knowledge of the denitrification rate, by

$$dTN/dt = 0 = -k_{TN}\, TN = -k_{dn}\, NO_3 \tag{7-4}$$

where:
k_{TN} = first-order removal rate for TN,
k_{dn} = denitrification rate,
NO_3 = concentration of NO_3-N

Equation 7-4 is then used to estimate the TN first-order removal rate, k_{TN}, on the basis of better-known values for k_{dn} and representative TN and NO3-N concentrations. Finally, wetlands removal efficiency, R (%), as a BMP is evaluated on the basis of steady-state load reduction,

$$R\ (\%) = 100 \times (W - QC)/W \tag{7-5}$$

7.6.3 PREWET Usefulness for Urban BMP Evaluation

Steady-state hydrology and water quality are of little usefulness in the urban stormwater setting. However, the value of PREWET is in its array of parameters (beyond those discussed above) related to sorption, settling, phosphorus cycling, and decay and other degradation processes. As a model it could also be used to check order-of-magnitude removal efficiencies on an annual basis to those computed using a continuous simulation model such as SWMM.

7.7 SIMULATION OF WETLANDS WITH DMSTA

7.7.1 Introduction

The Dynamic Model for Stormwater Treatment Areas (DMSTA) simulates daily water and mass balances in a user-defined series of wetland treatment cells, each with specified morphometry, hydraulics, and phosphorous cycling parameters (Walker and Kadlec 2002). An in-depth description of this model can be viewed at http://wwwalker.net/dmsta/index2.htm. However, the web site does not fully explain all the processes included in the model, and there appears to be no more detailed explanation available short of obtaining the model code. DMSTA was designed primarily to model total phosphorous concentrations in Everglades stormwater treatment areas (STAs) near Lake Okeechobee, Florida that receive agricultural runoff and releases. The goal of the STA project is to achieve TP outflow concentrations of 50 μg/L or less by 2007. The DMSTA model will help predict outflow phosphorous concentrations through the means of sedimentation, filtration, and adsorption. Within this model, up to six stormwater treatment areas (STAs) can be linked together at one time. STAs are areas capable of treating stormwater that contain vegetation, and are often wetlands, though not always. These STAs can be linked in either parallel or series systems to show compartmentalization and management to promote specific vegetation types (Walker and Kadlec 2002). Furthermore, each STA is broken down individually as a CFSTR and can be modeled as such. This model has been coded in Visual Basic for Applications, and the user interface is a Microsoft Excel workbook (Walker and Kadlec 2002).

7.7.2 DMSTA Model Features

The primary purpose of this model is to predict phosphorous cycling in STAs. Several of the input parameters required by the model are given below (Walker and Kadlec 2002):

- Linkage of treatment cells (up to 6 cells in series and/or parallel)
- Morphometry (length, width, area and cell configuration)
- Number of stirred tanks in series for each treatment cell
- Daily time series (for calibration runs only)
  - Inflow and outflow volume
  - Inflow and outflow TP concentration
  - Mean depth
  - Rainfall
  - Evapotranspiration
- Descriptive data
  - Seepage rates
  - Community description
  - Phosphorous storage (metadata: macrophytes, periphyton, and soil)

Other factors considered by the model include (Walker and Kadlec 2002):

- Temporal variations in inflow volume, load, rainfall, and evapotranspiration
- Hydraulic compartments (cells, flow distribution levees)
- Residence time distribution
- Water level regulation
- Compartmentalization of biological communities
- Dry-out frequency and supplemental water needs
- Bypass frequency, quantity and quality
- Inflow pulse modulation by upstream storage reservoir
- Seepage collection and management

The model is operated on a daily time step and has been run for periods of up to 31 years. Required input time series include daily values for inflow, inflow phosphorus concentration, rainfall, and ET. Seepage and outflows are computed through a model specific to the Everglades STAs being simulated. Spatial variation within a cell (i.e., within an STA) can be approximated by breaking the cell into a series of CFSTRs.

7.7.3 DMSTA Phosphorous Cycling Model

TP computations within DMSTA are similar to those within WETLAND and are essentially a dynamic version of PREWET. That is, phosphorus parameters are very similar to those required by PREWET, but unlike PREWET, DMSTA is a dynamic model, not a steady-state model. However, DMSTA output has been compared to steady-state simulations of the same areas (Walker 1995).

DMSTA considers storage of phosphorus in biomass, and nonlinear relationships (typically second-order kinetics) are used to simulate the exchange of phosphorus between the water column and the biomass. Biomass itself is essentially wetland vegetation (emergent macrophytes, submerged aquatic vegetation, and periphyton). It appears that DMSTA does not include growth models for the three types of vegetation and instead represents their biomass only by a long-term means. Many coefficients have been obtained for the Everglades region through calibration using several monitored systems.

7.7.4 DMSTA Usefulness for Urban BMP Evaluation

This model is useful in similar ways to PREWET, that is, for evaluation of coefficients that might be included in a similar algorithm in SWMM. WETLAND, PREWET and DMSTA all include complex, nonlinear kinetics for phosphorus cycling in the manner of WASP (Wool et al. 2001). Inclusion of such terms in SWMM might be warranted for large wetland BMPs with standing water throughout the year. But such complexity is probably not warranted for simpler devices such as bioswales.

7.8 SIMULATION OF WETLANDS AND OTHER BMPS WITH MUSIC

7.8.1 Introduction

The Model for Urban Stormwater Improvement Conceptualization (MUSIC) was developed by the Cooperative Research Centre for Catchment Hydrology (CRCCH) in Melbourne, Australia. MUSIC is capable of continuous simulation, with time steps ranging from 6 minutes to 24 hours. The model was designed to operate over a range of temporal and spatial scales, suitable for catchment areas from 0.01 km^2 to over 100 km^2 (Wong et al. 2002). It is important to note that MUSIC was developed as a *decision support system* and is not a detailed design tool (Wong et al. 2002). The model is intended to be a tool used in conjunction with other techniques to evaluate differing strategies for treating urban stormwater. MUSIC is capable of modeling wetlands, ponds, infiltration strips, buffer strips, swales, sedimentation basins, and gross pollutant traps. For the purpose of this report, only the wetland and pond portion of the model will be reviewed. Output from the model includes time-series graphs of flows, pollutant loads or concentration, statistical summaries, and cumulative probability plots (Wong et al. 2002). MUSIC does not contain the necessary complex algorithms for runoff routing, catchment contaminant build-up and wash-off processes, and does not enable the detailed sizing of structural stormwater quantity and/or quality facilities. Additional general and specific information about MUSIC is available at the web site: http://www.toolkit.net.au/products/music/index.htm.

7.8.2 MUSIC Algorithms

The algorithms used in MUSIC are based on the recognized routine characteristics of known stormwater quality improvement measures. The rainfall-runoff algorithm used in MUSIC was developed by Chiew et al. (1997). This is a water-balance equation similar to those used in models such as WETLAND and PREWET. MUSIC routes runoff using the Muskingum-Cunge equation (Bedient and Huber 2002, Dingman 2002).

TSS, TP, and TN concentrations are generated using a stochastic process involving cross correlation between TSS and TP and serial correlation of water quality time series (Wong et al. 2002). Pollutants that enter a stormwater treatment area (e.g., wetland or pond) in MUSIC are treated in CFSTRs. The option exists to treat the wetland/pond as either a series of CFSTRs or as a plug flow reactor (PFR), as discussed in Section 6.5. Plug flow implies a very small amount or no short-circuiting of flow.

Wong et al. (2001) present the background for the estimate of the parameter, N, the number of CFSTRs in series (Equation 6-9). Hydraulic inefficiency, i.e., the tendency to deviate from the ideal of plug flow and not to realize the full potential residence time of a wetland or pond can be due to at least two primary factors (Thackston et al. 1987):

- Dispersion effect caused by unsteady flow rates, wind, entrance and outlet effects, shear stresses at the sides and bottom, etc.

- Volume effect caused by “dead zones” in which velocities toward the outlet are considerably less than average and in which recirculation currents exist. Dead zones are not part of the volume through which water flows; hence, the effective volume is less than the total volume, V, and the effective residence time is less than the theoretical residence time, V/Q (Thackston et al. 1987).

These effects can be quantified through tracer tests described by several authors, including Fair et al. (1968), Levenspiel (1972), Thackston et al. (1987), and Kadlec and Knight (1996). Some of these studies attempt to derive an overall indicator of “hydraulic efficiency” such that a number near 1.0 indicates good hydraulic efficiency (close to plug flow) and a number near zero indicates poor hydraulic efficiency (close to complete mixing). Recall that the number of CFSTRs (or “tanks in series,” TIS), N = 1 (Equation 6-11) for complete mixing and ∞ for plug flow. Thus a natural efficiency indicator for mixing is

$$\varepsilon_{mix} = 1 - 1/N, \quad \text{typically: } 0 \le \varepsilon_{mix} \le 1 \tag{7-6}$$

The value for N can be estimated from tracer data from moments of the residence time distribution (RTD). There is a continuous analog to N discrete tanks in series, for which the analytical solution for the unit impulse response is a gamma distribution (Kadlec and Knight 1996, Eqn. 9-108),

$$f(t) = \frac{N}{\Gamma(N)}\left(N\frac{t}{\tau}\right)^{N-1} e^{-N\frac{t}{\tau}} \tag{7-7}$$

where
τ = mean of tracer distribution, not necessarily = V/Q for real data,
$\Gamma(N)$ = gamma function = (N-1)! if N is an integer (N is not required to be an integer).

The function is plotted in Figure 7-2. The concept is that each “tank” contributes a fractional residence time τ/N. The mean of the distribution is at $t = \tau$, or equivalently, $t/\tau = 1$. It is clear that as N increases, the distribution becomes more peaked, approaching a PFR. The special case N=1 corresponds to the exponential distribution for complete mixing in one tank,

$$f(t) = e^{-t/\tau} \tag{7-8}$$

If the mode (temporal location of peak) is t_p, then from the moment relationships (Kadlec and Knight 1996),

$$\frac{\tau - t_p}{\tau} = \frac{1}{N} \quad \text{or} \quad 1 - \frac{1}{N} = \frac{t_p}{\tau} = \varepsilon_{mix} \tag{7-9}$$

where Equation 7-7 has been included to indicate one measure of mixing efficiency. Equation 7-10 provides one way to evaluate N from moments of the tracer distribution (Kadlec and Knight 1996, Eqn. 9-111), although Kadlec and Knight offer Levenspiel’s (1972) assessment that another moment relationship is preferable for flat observed distributions where the mode may be difficult to identify,

$$\frac{1}{N} = \frac{\sigma^2}{\tau^2} = CV^2 \tag{7-10}$$

where

σ^2 = variance of tracer distribution, and
CV = coefficient of variation of tracer distribution.

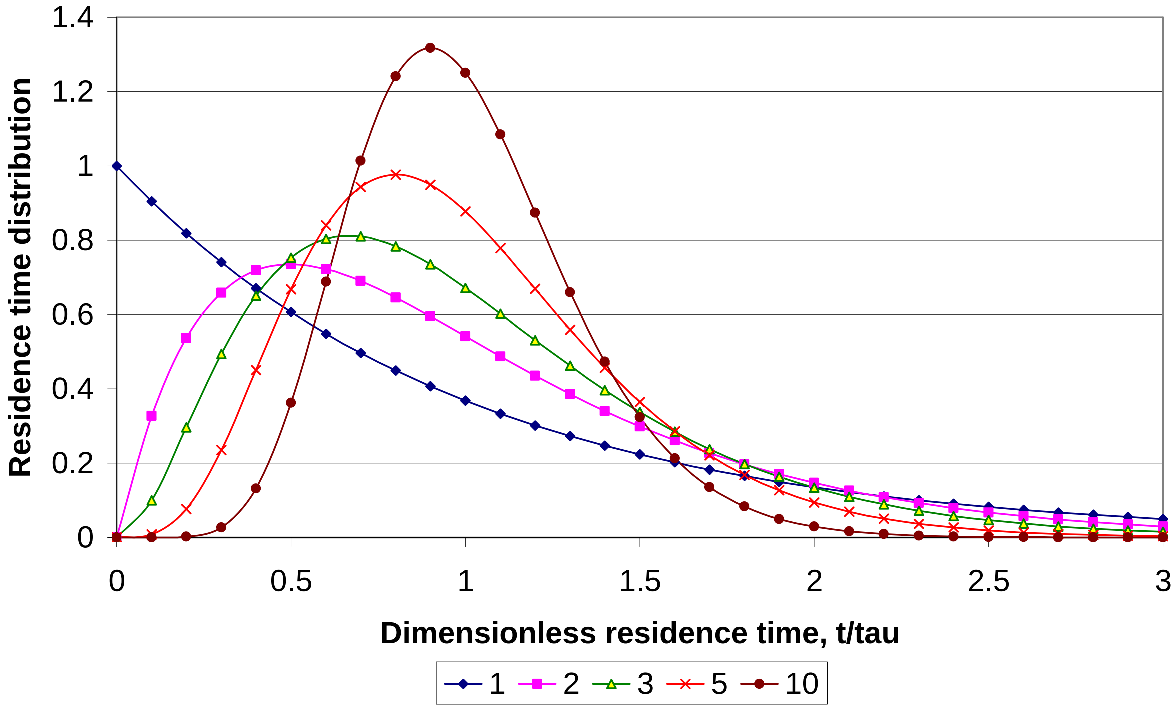

Figure 7-2. Theoretical residence time distribution (Equation 7-7) for unit impulse to N tanks in series. Values of N are given in the legend.

Persson et al. (1999) discuss difficulties of obtaining moments from real tracer data and provide alternatives to Equations 7-9 and 7-10 based on percentiles of the distribution. However, these authors end up using Equation 7-9 to evaluate mixing efficiency on the basis of simulated flows (using the MIKE-21 model, http://www.dhisoftware.com/mike21/) for various geometric layouts of wetlands (or ponds). The authors simulated 13 hypothetical ponds (Figure 7-3) and generated an output tracer distribution from a “spike” (unit impulse) input. Moments of the simulated tracer distribution were analyzed to compute ε_{mix} from Equation 7-9. The effective volume ratio, $\varepsilon_{vol,}$ was computed in the manner of Thackston et al. (1987) for a through-flow Q,

$$\varepsilon_{vol} = V_{effective}/V_{total} = \tau/t_d \qquad (7\text{-}11)$$

where
$V_{effective}$ = effective volume through which passes the flow,
V_{total} = total volume of wetland or pond, not all of which is encountered by the flow,
τ = mean detention time or first moment of the tracer distribution, = $V_{effective}/Q$, and
t_d = nominal or theoretical detention time = V_{total}/Q.

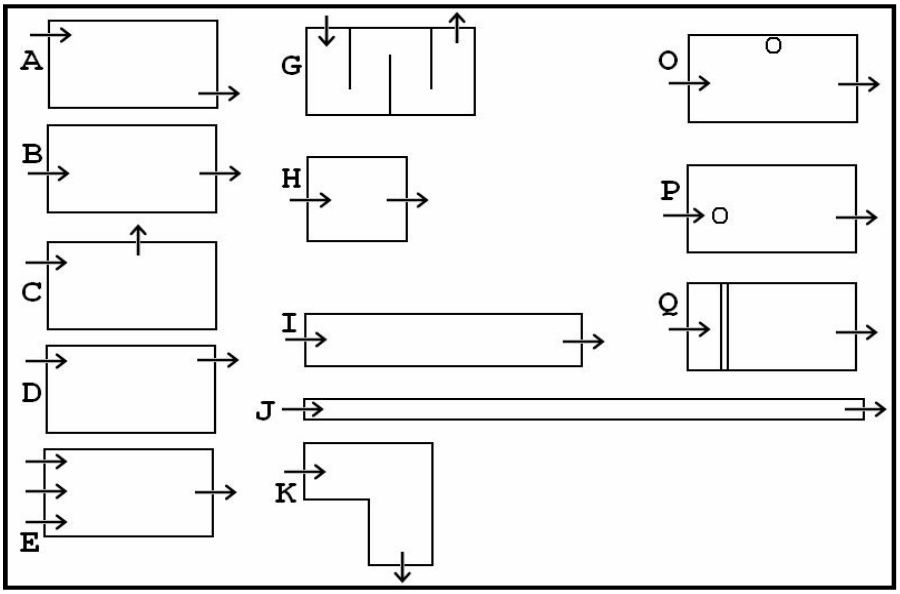

Figure 7-3. Pond shapes simulated by Persson et al. (1999). Pond G has baffles, ponds O and P have islands, and pond Q has a sill.

Again, the measured mean detention time, τ, is usually less than the nominal detention time, t_d, due to dead zones. Hence, the volume efficiency is usually a number between 0 and 1 (Thackston et al. 1987). Finally, Persson et al. (1999) arrive at an overall "hydraulic efficiency," λ, as the product of the mixing and volume efficiencies,

$$\lambda = \varepsilon_{mix} \cdot \varepsilon_{vol} \tag{7-12}$$

Values of λ for the 13 shapes in Figure 7-3 are given in Table 7-2, ranked in order of decreasing hydraulic efficiency. Highest efficiencies are for ponds with a distributed inflow (pond E), baffles (pond G), and very elongated flow or high length to width ratio (pond J).

Table 7-2. Numerical results of Persson et al. (1999) for pond shapes of Figure 7-3. The qualitative rating of hydraulic efficiency is by Persson et al.

Pond	ε_{vol}	ε_{mix}	λ	$N \approx 1/(1-\lambda)$	Qualitative Rating
J	1.00	0.90	0.90	10.0	Good
G	1.00	0.76	0.76	4.2	
E	0.89	0.85	0.76	4.1	
P	0.96	0.64	0.61	2.6	Satisfactory
Q	0.93	0.64	0.60	2.5	
I	1.00	0.41	0.41	1.7	Poor
K	0.78	0.46	0.36	1.6	
A	0.74	0.41	0.30	1.4	
B	0.79	0.33	0.26	1.4	
O	0.73	0.35	0.26	1.3	
D	0.34	0.52	0.18	1.2	
H	0.44	0.25	0.11	1.1	
C	0.46	0.23	0.11	1.1	

From an analysis of the data of Persson et al. (1999), Wong et al. (2001, 2002) recommend for MUSIC an *approximation* for N, the number of CFSTRs or tanks in series,

$$N \approx 1 - 1/\lambda \tag{7-13}$$

Values of N from Equation 7-13 are included in Table 7-2. In effect, this attributes all imperfect mixing just to the dispersion effect and none to the volume effect. Thus, Table 7-2 and Figure 7-3 provide a qualitative estimate for N for use in Equation 6-11. The application of the numerical results of Persson et al. (1999) to real wetlands is also discussed by Wong and Breen (2002).

The pollutants described by MUSIC are modeled using a first-order kinetic model. This is Kadlec and Knight's (1996, Eqn. 9-103) k'-C* model and is expressed similarly as:

$$(C_{out} - C^*) / (C_{in} - C^*) = e^{-k`/q} \tag{7-14}$$

where
C^* = background concentration (mg/L),
C_{in} = input concentration (mg/L),
C_{out} = output concentration (mg/L),
k` = rate constant (m/y), and
q = hydraulic loading or overflow rate (m/y).

The argument of the exponential in Equation 7-15 is the Damköhler number, discussed in relation to Equation 6-14.

Equation 7-14 was adapted from another, earlier Australian model, the Universal Stormwater Treatment Model (USTM), and is used to simulate pollutants as they pass from one CFSTR to another (Wong et al. 2001, 2002). This equation is computed separately for each time step at each CFSTR. The main difference between this equation and ordinary first-order decay modeling is the inclusion of C*, the equilibrium or background concentration. This means that effluent concentrations will not be reduced below C*. It may be noted from Equation 6-14 that

$$e^{-k'/q} = e^{-k t_d} \tag{7-15}$$

That is, using depth, h, as a linking variable,

$$\frac{k'}{q} = \frac{k\,h}{h/t_d} = k\,t_d \tag{7-16}$$

where
k = first-order decay coefficient = k'/h (1/time),
h = average depth, and
t_d = nominal detention time = V/Q.

Some recommended k` and C* values are given in Table 7-3, based on limited model calibration for TSS, TP, and TN in urban areas near Melbourne (Wong et al. 2002). If depths were used to compute k' values, they are not reported. In fact, one advantage of the formulation using rate constant k' (depth/time) is that it avoids having to specify an average depth for odd natural configurations.

Table 7-3 Calibrated k' and C* values from MUSIC based on limited model simulations.

Treatment Measures	k` (m/yr)			C* (mg/L)		
	TSS	TP	TN	TSS	TP	TN
Sedimentation Basins	15,000	12,000	1,000	30	0.18	1.7
Ponds	1,000	500	50	12	0.13	1.3
Vegetated Swales	15,000	12,000	1,000	30	0.18	1.7
Wetlands	5,000	2,800	500	6	0.09	1.3

Refinement of the parameters for the k'-C* model to suit local conditions (particle size distributions in particular) and treatment measure design specifications, is currently being undertaken (Wong et al. 2002). It is expected that the parameter C* will vary with discharge and the influence of chemical and biological processes during the inter-event period. Derived k values for ordinary CFSTR modeling that result from using the TSS and k` values in Table 7-3 are shown in Table 7-4 for three representative depths. But it will be seen in Chapter 15 that these values are probably too high for simulation of "decay" of TSS in most ponds.

Table 7-4. First-order decay values converted from k` values for TSS in Table 7-3 for assumed depths.

Treatment Measures	Depth = 1 ft	Depth = 3 ft	Depth = 5 ft
	TSS k, 1/day	TSS k, 1/day	TSS k, 1/day
Sedimentation Basis	134.8	44.9	27.0
Ponds	8.99	3.0	1.80
Vegetated Swales	134.8	44.9	27.0
Wetlands	44.9	15.0	8.90

The same k'-C* model is used for ponds, wetlands (Wong and Breen 2002), grass swales (Fletcher et al. 2002), and gravel filters (Wong et al. 2002). It has proven adaptable to fitting of many observed BMP performance data in Australia (Wong and Breen 2002).

7.8.3 MUSIC Evaluation

MUSIC has been calibrated and tested for its ability to predict pollutant removals by various different companies and consultants. The equations used in MUSIC are fairly simplistic and are first order. It appears to predict the removal of TSS and TN well, and TP moderately well. The uniqueness of this model is its ability to link together different treatment options such as a wetland coupled with a swale, or a wetland coupled with a pond, through a convenient GUI. This type of simulation (known as a "treatment train") yields higher pollutant removal efficiencies than from just one BMP alone. Presently, SWMM is capable of simulating treatment devices in series in the Storage/Treatment Block, or in Runoff and Transport channels and storages. However, sufficient information is not stated within the MUSIC training manual or other references reviewed as to whether or not correct simulation of the treatment train occurs. Correct simulation depends upon recognition that the upstream device removes the easiest materials (e.g., heavy solids). Downstream devices are left to remove fine particles and dissolved constituents. Therefore, downstream removal efficiencies will progressively lessen, and it is not clear from the documentation whether or not this is observed in MUSIC.

Once again, another model recognizes the efficacy of multiple CFSTRs or the tanks in series approach (Equation 6-9). The use of tracer data to identify the number, N, of the series of CFSTR would be helpful for parameter estimation, should such an approach be implemented in SWMM.

The way MUSIC models first-order decay (Equation 7-14) may be worthy of implementation into SWMM. Specifically, the inclusion of the C* term (the background concentration) might be included in

SWMM first-order decay modeling. However, C* is a constant for the simulation. A better alternative might be a distribution of effluent concentrations (Strecker et al. 1991).

7.9 SWMM SIMULATION OF WETLANDS AND BIORETENTION DEVICES

As described in Chapter 1, SWMM simulates wetlands in the same manner as it simulates storage or ponds (Chapter 4 and Section 6.5). Hydraulic efficiency options range from plug flow in the S/T Block, to complete mixing (CFSTR) in S/T or in any Runoff or Transport Block channel or storage device. Similarly, bioretention devices may be also be simulated in the manner of storage (Minton 2002) as discussed in Section 6.5. First-order decay and settling are readily simulated, and the "universal removal equation" (Equation 4-3) provides the option for curve-fitting of observed removal performance. What is missing is interaction among state variables, as exemplified by the WETLAND model, and performed more simply with the VAFSWM model. However, the complex nutrient dynamics involved in a model like WETLAND might be much too sophisticated for the typical analysis and design employed by stormwater engineers. The EPA WASP model (Wool et al. 2001) provides an alternative should such complex dynamics need to be considered. Hence, the most useful recommendation regarding SWMM is probably to enhance the linkage between land-surface runoff models, such as SWMM, and receiving water quality models, such as WASP or HSPF.

Interfacing of different models is not necessarily an easy task; Lin and Medina (2003) present one example of linking (through appropriate interface files) three USGS models: a stream transient model (DAFLOW), a groundwater flow model (MODFLOW), and a solute transport model (MOC3D plus a one-dimensional stream solute transport model. Even for models from the same agency, time-step and other considerations made the interfacing difficult. Interpolation in time series in order to match time steps between models can be a particularly difficult problem. There is an opportunity for leadership with the development of SWMM5: an interface file "standard" could and should be developed to facilitate exchange of time series data between models that are primarily watershed runoff models, such as SWMM, with models that are primarily receiving water quality models, such as WASP.

8 INFILTRATION TRENCHES

8.1 INTRODUCTION

Modeling infiltration trenches involves storage and release of stormwater capture volumes to groundwater. Infiltration is the major process and is simulated in several models.

8.2 SIMULATION OF INFILTRATION TRENCHES WITH SLAMM

8.2.1 Introduction

Infiltration devices are one of the control practices evaluated by SLAMM (Pitt and Voorhees 2000). Volume reduction from infiltration is dependant on the study area, runoff rates, infiltration rates and physical trench parameters. One approach is to take infiltration rates from SCS data, altered to reflect infiltration during micro-storms (storms with depths less than about 0.10 in.), and to adjust volumetric runoff coefficients. Similarly, the SWMM Horton or Green-Ampt methods could be used. But infiltration in trenches, swales, and channels may also depend upon the depth of water in the device, depth to shallow groundwater, clogging, etc. and thus require additional simulation efforts. The SLAMM procedure essentially ignores these complications, and is discussed below.

8.2.2 SLAMM Calculation Procedures for Infiltration Devices

Infiltration devices are assumed to affect water volume, but not pollutant concentrations. As the water volume is reduced, the pollutant yield (load) is obviously decreased. SLAMM calculates the runoff volume reductions for each source area (served by an infiltration device) for each individual rain event in the study period. Figure 8-1 shows the spatial breakdown for the SLAMM model.

Runoff volume reduction fraction is assumed to be:

$$\text{fractional volume reduction} = \left(\frac{Q_p}{Q_r}\right)\left(\frac{A_s}{A_t}\right) \qquad (8\text{-}1)$$

where

Q_p = the percolation volume rate of the device (cfs),
Q_r = the runoff rate to the device (cfs),
A_s = the area draining to the device (acres), and
A_t = the total study area (acres).

It can be seen that the fractional volume reduction is the product of the fraction of runoff infiltrated times the fraction of area served.

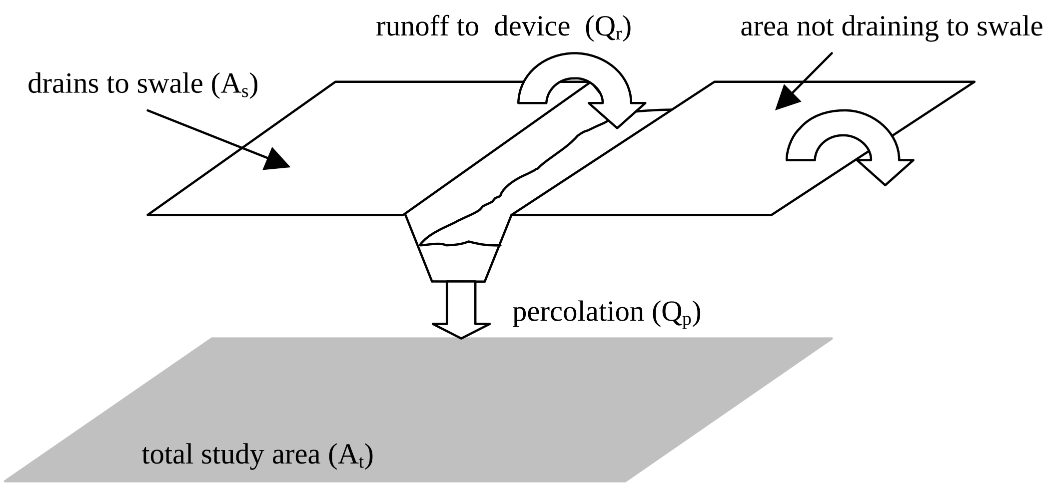

Figure 8-1. Schematic of SLAMM model area breakdown.

The ratio Q_p/Q_r used in this equation can never be greater than 1.0, because the device cannot infiltrate more water than is delivered into the device. The percolation volume rate, Q_p, is the capacity of the infiltration device to infiltrate runoff, expressed as cfs. Pitt and Voorhees (2000) assume (no other basis given) that each side wall of a vertical trench infiltrates 1/3 of the rate along the trench bottom. For a vertical-walled (rectangular) trench of depth h, width w, and length L, and infiltration rate (percolation rate) f (depth/time), the volume rate of percolation is thus:

$$Q_p = Lwf + \frac{2}{3}Lhf = Lwf(1 + \frac{2h}{3w}) \tag{8-2}$$

This yields the version cited by Pitt and Voorhees (2000) in which percolation area = Lw,

$$Q_p = \left(1 + \frac{0.67}{\text{width to depth ratio}}\right)(\text{percolation rate})(\text{percolation area}) \tag{8-3}$$

No specifications are given for trench design (trapezoidal or vertical side walls are not specified) in SLAMM.

Much of the effort within SLAMM is involved in generation of runoff, including the use of runoff coefficients obtained from extensive analysis by Pitt (1987) based on the evaluation of data obtained from NURP (EPA 1983), the EPA's Urban Rainfall-Runoff-Quality Data Base (Huber et al. 1982*)*, and from the Humber River portion of the Toronto Area Watershed Management Study (Pitt and McLean 1986). Since runoff generation is already provided by SWMM and not the main thrust of this presentation about BMPs, runoff generation by SLAMM is not included herein.

However, for purposes of presenting a brief SLAMM example, volumetric runoff coefficients, R_v, are defined conventionally by

$$\text{runoff volume} = R_v \text{ (area draining to device) (rain depth)}, \quad (8\text{-}4)$$

and Pitt (1987) presents empirical data relating the duration of runoff (hours) to the duration of rainfall as

$$\text{Runoff duration} = 0.90 + 0.98\,(\text{rain duration, in hours}) \quad (8\text{-}5)$$

An example of use of this procedure follows:
Percolation rate = 3 in./hr
Total rain = 1.7 in.
Rain duration = 6 hours
Volumetric runoff coefficient = 0.35
Area served by infiltration trench = 1.3 acres
Total area in study = 5.6 acres
Trench bottom area (percolation area) = 5500 ft^2
Trench width/depth ratio = 2

Therefore:

runoff volume = 0.35 (1.7 in.)(1.3 acres) = 0.774 ac-in.
runoff duration = 0.90 + 0.98(6 hours) = 6.78 hours
Q_r = 0.774/6.78 = 0.114 ac-in./hr = 0.115 ft^3/sec.
Q_p = [1 + 0.67/2] (3 in./hr) (5500 ft^2) (ft/12 in) (hr/3600 sec) = 0.510 ft^3/sec.

Therefore Q_p/Q_r = 0.51/0.114 = 4.43, which is greater than 1.0, so 1.0 must be used in Equation 8-1.

In this example the infiltration trench is oversized for this event since all of the runoff from the service area is infiltrated. This means that Pitt's effectiveness criterion is simply the ratio of area served to the total area. The study area volume reduction performance is therefore: 1.3 acres/5.6 acres = 0.23 (23 % of the runoff and pollutant load are infiltrated).

8.2.3 Infiltration in Disturbed Urban Soils

Disturbed urban soils do not behave as indicated by typically used models. More rain infiltrates through pavement surfaces and less rain infiltrates through soils than typically assumed (Pitt et al. 1999a, Pitt and Voorhees 2000). Double-ring infiltrometer test results from urban soils in Oconomowoc, WI (Table 8-1) indicated highly variable infiltration rates for soils that were generally sandy (NRCS A/B hydrologic group soils).

Many infiltration rates actually increased with time during these tests. In about one third of the cases, the observed infiltration rates remained very close to zero, even for these sandy soils. Areas that experienced substantial disturbances or traffic (such as school playing fields) had the lowest infiltration rates, typically even lower than concrete or asphalt (Pitt and Voorhees 2000). These values indicate the large variability in infiltration rates that may occur in areas having supposedly similar soils.

In an attempt to explain much of the variation shown in the Wisconsin tests, Pitt and his students conducted tests of infiltration through disturbed urban soils in the Birmingham, AL area (Pitt and Voorhees 2000). Eight categories of soils were tested, with about 15 to 20 individual tests conducted in each of eight categories (comprising a full factorial experiment). Numerous replicates were needed in each category because of the expected high variation in infiltration rates. The eight categories in Table 8-2 were tested. These tests resulted in the default infiltration parameters distributed with SLAMM (Table 8-3).

Table 8-1. Ranked double ring infiltration test results and observed urban soil infiltration rates from Oconomowoc, WI (Pitt and Voorhees 2000).

Initial Rate (in/hr)	Final Rate (after 2 hours) (in/hr)	Total Observed Rate Range (in/hr)
25	15	11 to 25
22	17	17 to 24
14	7 9	4 9.4 to 17
5.8	9.4	0.2 to 9.4
5.7	9.4	5.1 to 9.6
4.7	3.6	3.1 to 6.3
4.1	6.8	2.9 to 6.8
3.1	3.3	2.4 to 3.8
2.6	2.5	1.6 to 2.6
0.3	0.1	<0.1 to 0.3
0.3	1.7	0.3 to 3.2
0.2	<0.1	<0.1 to 0.2
<0.1	0.6	<0.1 to 0.6
<0.1	<0.1	all <0.1
<0.1	<0.1	all <0.1
<0.1	<0.1	all <0.1

Table 8-2 Categories tested for infiltration rates (Pitt and Voorhees 2000).

Category	Soil Texture	Compaction	Moisture
1	Sand	Compact	Saturated
2	Sand	Compact	Dry
3	Sand	Non-compact	Saturated
4	Sand	Non-compact	Dry
5	Clay	Compact	Saturated
6	Clay	Compact	Dry
7	Clay	Non-compact	Saturated
8	Clay	Non-compact	Dry

Table 8-3 Percolation rates for different soil texture and moisture used in SLAMM (Pitt and Voorhees 2000).

Soil Description	Number of tests	Average Infiltration rate (in/hr)	CV
Non-compact sandy soils	29	17	0.43
Compact sandy soils	39	2.7	1.8
Non-compact and dry clayey soils	18	8.8	1.1
All other clayey soils	60	0.69	2.1

The CV, or coefficient of variation, is the ratio of the standard deviation for a variable to the mean value of the variable. This is used by Pitt and Voorhees (2000) to measure the imprecision in survey estimates introduced by sampling. A coefficient of variation of 1% would indicate that an estimate could vary slightly due to sampling error, while a coefficient of variation of 50% means that the estimate is very imprecise.

Although the CV values shown in Table 8-3 for the infiltration tests are generally high, Pitt and Voorhees (2000) claim that they are much less than if compaction was ignored. The high variation within each of the four main categories makes it difficult to identify legitimate patterns, implying that average infiltration rates within each event may be most suitable for predictive purposes. Other infiltration rates for clayey and sandy soils can be taken from Figures 8-2 and 8-3.

8.2.4 SLAMM Procedures for SWMM

Infiltration trench procedures used in SLAMM appear to have little value for SWMM except for the very positive aspect of providing infiltration rate data (previous section). Runoff generation is already performed in SWMM using a dynamic procedure not involving runoff coefficients or regression. The latter work well in SLAMM and may be additionally useful in spreadsheets, but do not appear to usefully enhance the current SWMM Runoff Block procedures. Infiltration itself is also dynamic (Horton or Green-Ampt), and a work-around procedure is available to simulate trenches, explained in the following section.

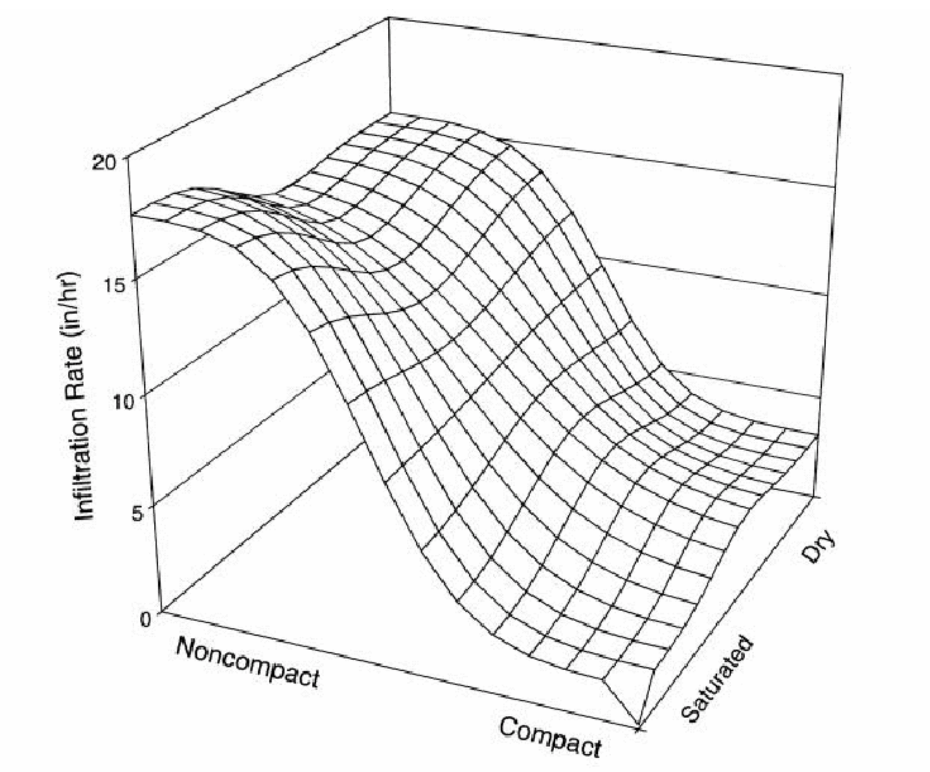

Figure 8-2. 3-D plots showing interactions affecting infiltration rates in sandy soils (Pitt and Voorhees 2000).

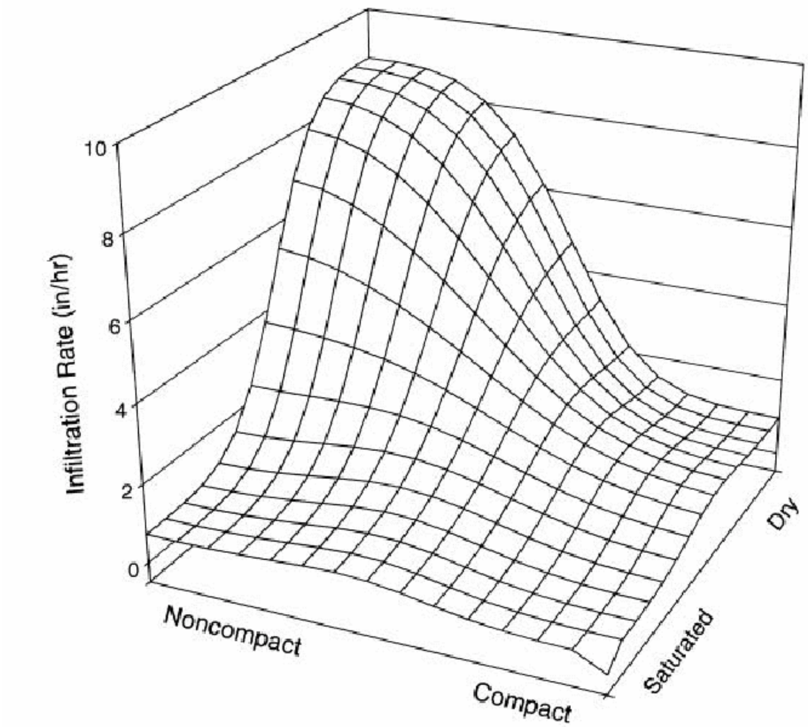

Figure 8-3. 3-D plots showing interactions affecting infiltration rates in clayey soils (Pitt and Voorhees 2000).

8.3 SIMULATION OF INFILTRATION TRENCHES IN SWMM

In spite of the fact that the current SWMM flow routing procedures do not allow infiltration from *channels*, the ability to route flow from one overland flow plane onto another (Section 4.7) allows runoff block subcatchments to serve as vertical-walled infiltration trenches. The procedure in SWMM (current or SWMM5) is as follows:

1. Simulate subcatchment runoff by usual procedures and route it downstream to the infiltration trench subcatchment.
2. Simulate an infiltration trench as a 100% pervious subcatchment of width w and length L (trench dimensions). The depth is implicitly infinite since there is no maximum subcatchment water depth for overland flow planes. However, depression storage could be set equal to the trench depth, thus ensuring no horizontal outflow for water depths less than or equal to the depression storage depth. But the modeler would have to ensure that the trench could accept all inflow (that is, not flood), unless there was provision to accept such "overflow" as legitimate flow to an auxiliary drain.
3. Infiltration may be simulated by Horton or Green-Ampt; if a constant rate is desired, it is easier to manipulate the Horton equation (maximum infiltration rate = minimum infiltration rate). Note that water depth will have no effect on infiltration within the SWMM model formulation. The infiltration rate might be adjusted higher to reflect the fact that there will be some infiltration out through the side walls that the model cannot simulate. Alternatively, a larger planar area than the actual length and width could be used, but both methods are judgmental.
4. A combination of low slope, high Manning's n, and/or very small conceptual width should be provided to eliminate horizontal outflow out of the trench – unless such outflow actually occurs, into a drain, say, when the water level is above the depression storage. If this work-around procedure does produce water depths higher than the trench depths, results should be carefully checked.

Drainage from the infiltration trench subcatchment can be directed to a groundwater component if further tracking is desired, an advantage. Note that water in the trench will also be subject to evaporation and rainfall on the trench itself. There is no ready way to simulate reduction of infiltration capacity due to sedimentation apart from running the model with different parameter sets.

A better formulation would consist of a channel (with the option of a weir, to prevent outflow for water levels below the weir level = trench depth). This would permit other cross sections besides vertical walled, such as trapezoidal. But the model needs to be modified to allow infiltration from channels, including the possibility that infiltration might increase with water depth, and some consideration of clogging over time. Infiltration methods, including SWMM's Green-Ampt procedure, which might be suitable for this purpose, are reviewed by Williams et al. (1998).

8.4 TRANSITION TO SIMULATION OF RAIN GARDENS AND GREEN ROOFS

In the same manner that SWMM may currently be used to simulate infiltration trenches, the model can be used to simulate rain gardens and green roofs. Here, the conceptualization is more accurate than for infiltration trenches because the drainage is confined between vertical walls (i.e., the walls of the vegetated plots, as in a planter). The vegetated area may simply be a source subcatchment or a subcatchment to which flow is directed from an upstream source, such as an impervious portion of the roof. It is important to include the groundwater modeling option in order to obtain a vertical water balance and to provide for vertical drainage through subsurface geo-fabrics, screens, or other soil structures.

Another, more indirect simulation of green roofs can be performed by modeling each soil layer as an S/T unit. The upper unit drains to a lower unit on the basis of a prescribed rating curve. The advantage of this conceptualization is that quality parameters may be tracked through the units. The disadvantage is that the vertical water balance must be simulated indirectly, as in a prescribed time series of ET and/or outflow hydrograph.

9 GRASS SWALES AND FILTER STRIPS

9.1 INTRODUCTION

Grass swale drainages can be used in place of concrete curb and gutter drainages in most land uses, except strip commercial, manufacturing industrial, and high-density residential areas (Pitt and Voorhees 2000). Grass swales reduce urban runoff problems by a combination of mechanisms. Infiltration of the runoff and associated pollutants is probably the most important process of removal in grass swales. Filtering of particulate pollutants in grassed waterways may also occur, but the flows are usually too large (and deep) to permit effective filtering by grass (Pitt and Voorhees 2000). However, Minton (2002) points out that settling is a primary unit process as water flows through swales, and in fact, as long as there is storage (water depth) along the swale, solids removal may be simulated in the same way as for ponds; see Section 6.5. This will be explored further at the end of this section.

Filter strips differ from swales in the sense of not necessarily consisting of a channel, rather just an overland flow path. A large swale may be conceptualized as having its upper banks consist of filter strips, whereas the active channel is the swale. Performance data are sometimes differentiated on this basis.

Groundwater contamination concerns are frequently raised whenever stormwater infiltration is proposed. Pitt et al. (1996) reported that groundwater contamination is not a major concern for most stormwater if using surface spreading (such as occurs in grass swales). Pitt et al. (1999b) also reported on the accumulation of stormwater pollutants in the surface soils of swales, minimizing groundwater contamination problems.

9.2 SIMULATION OF GRASS SWALES WITH P8

P8 uses the same methods to describe settling and decay in swales as in ponds. Particle and pollutant removal are also calculated similarly, although runoff velocities in swales are calculated with Manning's equation. An added process modeled in swales is infiltration, and the associated filtration. For pervious areas, infiltration is calculated with the SCS runoff curve number technique, and thus does not include the additional complexities of depth and sedimentation mentioned earlier. Infiltration rates used in P8 are listed in Table 9-1.

Filtration efficiencies for all infiltration particle fractions are assumed to be 100%, and 90% is assumed for the dissolved fraction to account for the adsorption, precipitation and other reactions between dissolved contaminants and the soil matrix. There is no method to calculate resuspension of particulates if critical velocities for incipient motion are reached. Therefore, surface water outflows from grass swales have reduced pollutant mass due to the reduction in water volume from infiltration.

This method for tracking groundwater may be useful if modeling BMP interactions with a shallow groundwater system, although a 90% removal rate of all dissolved pollutants seems optimistic. This should strongly be a function of the soil type and the pollutant in question. For instance, nitrate does not sorb strongly, but heavy metals might. This would only be important to modelers wishing to track pollutants in groundwater, which is not available in SWMM (but is in HSPF).

Table 9-1 Infiltration rates used in P8.

References:	**(a)**	**(b)**		**(a)**	**(c)**
	Infiltration rate (in/hr)			**Infiltration rate (in/hr)**	
Soil Textures			SCS soil group		
sand	4.64	8.27	A	0.43	.30-.45
loamy sand	1.18	2.41	B	0.26	.15-.30
sandy loam	0.43	1.82	C	0.19	.05-.15
silt loam	0.26	0.27	D	0.03	.00-.05
loam	0.13	0.52			
silt loam		0.27			
sandy clay loam	0.06	0.17			
clay loam	0.01	0.09			
silty clay loam	0.04	0.06			
sandy clay	0.03	0.05			
silty clay	0.02	0.01			
clay loam	0.01	0.02			

Sources: a- Rawls et al. (1983) values for saturated hydraulic conductivity. b-Shaver (1986) c-Musgrave(1955)

9.3 GRASS SWALE PERFORMANCE CALCULATIONS IN SLAMM

SLAMM calculates the performance of grass swales in a similar manner as other infiltration devices, by assuming (Q_p/Q_r) (A_s/A_t) as indicative of swale infiltration (refer to Equation 8-1).

The water percolation rate in the swale is calculated by:

$$Q_p = \text{(dynamic percolation rate) (percolation area)} \qquad (9\text{-}1)$$

where
percolation area = swale length times the swale width, and
percolation rate in the swale is for dynamic flow conditions and has been found to be about one-half of the typically measured static infiltration rate in some Florida locations (Wanielista et al. 1983).

This procedure is generally independent of swale routing; it assumes that the water is in the swale long enough to be infiltrated. “Long” swales serving “small” service areas encourage infiltration. Grass swales include infiltration as a function of flow distance for different slopes and infiltration rates and can therefore be used to estimate needed flow length in swales (Pitt 1985, 1987). Obviously, swale design (like all other controls) must be carefully done to encourage performance. As an example, these procedures would not be appropriate for steep swale gradients. The ratio of area served by swales to total area therefore needs to be reduced if steep swales are present, or if the swales are "short."

An example of the calculations for swale performance follows:
Total contributing flow volume = 1140 ft^3
Rain duration = 5.5 hours
Dynamic percolation rate in swale = 3.5 in./hr (1/2 of measured static infiltration rate)
Swale density = 350 ft/acre
Wetted swale width = 5 ft
Area draining to swales = 1.5 acres
Study area = 3.3 acres

Therefore the runoff duration (Equation 8-5) = 0.90 + 0.98 (5.5 hours) = 6.29 hours, and
Q_r = 1140 ft^3/6.29 hrs = 181 ft^3/hr = 0.050 cfs
Q_p = (3.5 in./hr)(350 ft/acre)(1.5 acre)(5 ft)(hr/3600 sec)(ft/12 in.) = 0.21 cfs

Therefore Q_p/Q_r = 0.213/0.05 = 4.26, which is greater than 1.0 and the swale is larger than necessary for this rain (total infiltration). The study area runoff reduction is therefore 1.5 acres/3.3 acres = 0.46 (46 percent reduction in flows and pollutant yields due to the swales).

Once again, SLAMM procedures for swales appear to offer little to be added to SWMM, except for good data. A review of output from SLAMM also suggests that SWMM might be improved through additional tables of effectiveness measures, such as volume reduction, etc.

9.4 SIMULATION OF VEGETATED FILTER STRIPS WITH REMM

The processes that occur in filter strips are sedimentation, filtration, infiltration and biochemical interactions. Discussions of these processes in the previous sections are applicable to this BMP. A model that thoroughly investigates the biochemical processes in filter strips is the Riparian Ecosystem Management Model (REMM), developed by the USDA in partnership with the Southeast Watershed Research Laboratory at Tifton, GA. It was developed for natural resource agencies and researchers as a tool that can help quantify the water quality benefits of riparian buffers in response to changes in upland agricultural use (Inamdar et al. 1998a,b; Inamdar et al. 1999a,b; USDA 1999; Lowrance et al. 2000). The following discussion is based on portions of all of these six references. Additional discussion of this model is in the Appendix. REMM simulates: (a) the movement of surface and subsurface water; (b) sediment transport and deposition; (c) transport, sequestration (capture) and cycling of nutrients; and (d) vegetative growth.

The strengths of the REMM model are its ability to deal with subsurface fate and transport of nutrients. REMM can be applied to:

- Quantify nitrogen and phosphorus trapping in riparian buffer zones and to determine buffer width for a given set of riparian conditions and upland loadings.
- Determine buffer effectiveness under increased loads.
- Evaluate influence of vegetation type on buffer effectiveness.
- Determine impacts of harvesting on buffer effectiveness.
- Investigate long-term fate of nutrients in riparian zones, sequestration in vegetation, or loss to atmosphere (denitrification in case of N).
- Investigate N / P saturation in riparian buffers.

The REMM model is the only model reviewed that integrates subsurface flow (three layers) and groundwater interaction when simulating buffers. REMM also closely models the N, P, and C cycles in the buffer that is broken into three zones: 1) closest to stream, 2) middle, and 3) farthest from stream. Although determining site-specific nutrient cycling parameters requires extensive field data to simulate accurately, the most useful information from the REMM model may be the default rate constants used in these calculations.

The model operates on a daily time step and requires daily loadings from upland areas and daily meteorological data. Output includes daily time series of surface and subsurface flows and water quality. Comparison of outflow and influent loads yields BMP effectiveness. The model can be used to study the effectiveness of buffer strips of various lengths (in the direction of water flow), for instance. If subsurface outflow to an adjacent stream is important, REMM can also provide those fluxes.

Because the writers were unable to obtain feedback from the USDA at Tifton, current support for this model is minimal and algorithms used for modeling calculations are unavailable. As documentation for REMM becomes available the model should be revisited. But overall, the nutrient dynamics algorithms appear to be beyond what would be reasonably expected of SWMM, in the manner of the WETLAND model. A more general description of this model is presented in the Appendix.

9.5 SIMULATION OF GRASS SWALES WITH SWMM

The "obvious" way to simulate grass swales in SWMM is to model them as channels that infiltrate. Unfortunately, SWMM version 4.4h cannot infiltrate directly from open (or closed) channels. Work-arounds for infiltration include the option for entering a negative hydrograph upstream of the channel, to simulate outflows, but this is not very satisfactory. Alternatively, a swale could be modeled as a rectangular channel by simulating it as a subcatchment, as discussed in Section 8.3 for infiltration trenches. Again, since real swales are usually trapezoidal and since infiltration might depend upon water depth, this is less than satisfactory, but perhaps the best current option since pollutant routing across downstream subcatchments does reflect first-order decay, a constant settling velocity, and/or constant removal fraction.

Still another option within the current SWMM is to simulate swales with the S/T Block, as a storage device. Infiltration may be simulated either as 1) monthly evaporation, or 2) residuals outflow. The latter could be adjusted to provide infiltration as a function of depth through a rating curve. Minton (2002) indicates that swales are essentially shallow clarifiers, and sedimentation occurs by dynamic settling, which is therefore a function of hydraulic loading rate, as discussed in Section 6.5. This is supported for TSS removal by data from Brisbane by Fletcher et al. (2002), for which the k'-C* model discussed in Section 7.7 provides a good fit. S/T simulation is probably the most accurate, if least intuitive method available in the current SWMM for simulation of pollutant removal in swales. It also has the advantage of providing options for simulating removal of soluble pollutants, should any such removal be noted in swale BMPs.

It is discouraging that the literature review for this and related projects has not uncovered simpler functional relationships to represent removal in buffer strips or along overland flow. Overland flow and riparian removal effectiveness intuitively would depend upon the length of the overland flow pathway. An example is shown in Figure 9-1 (Huber et al. 2000) and is similar in principle to pollutant reduction in swales observed by Fletcher et al. (2002). As one would intuitively expect, removal efficiency does increase with length of overland flow, but there is no unifying result from this set of experiments, at any rate. Overland flow removal effectiveness data tend to be found in the agricultural literature, although the data of Barrett et al. (1997) are from a highway median. A problem in analysis of such data is the lack of consistent reporting of the physical conditions of the filter strips, such as slope, soil type, vegetation, antecedent conditions, etc. In addition, agricultural experiments often utilize animal wastes as loading materials and thus may not be reasonable approximations of stormwater characteristics for urban runoff. Additional review of the literature may yield better data in an effort in support of SWMM algorithm improvement.

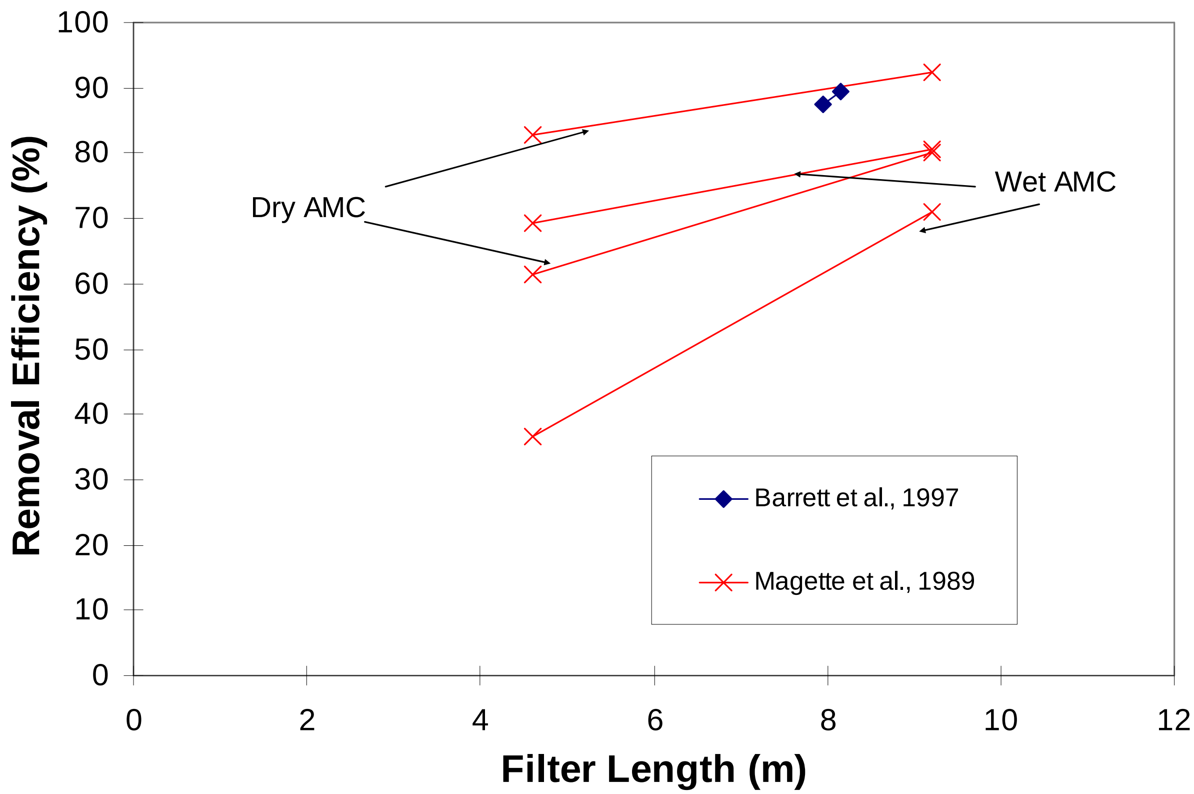

Figure 9-1. TSS removal effectiveness of vegetated filter strips, based on total mass of suspended solids entering and leaving the strip (Huber et al. 2000). (“AMC” refers to antecedent moisture condition.)

10 DRY WELLS

10.1 INTRODUCTION

Dry wells are usually holes several feet deep, that are filled with porous material and that fill with water, which then infiltrates. The P8 method for modeling storage and infiltration would adequately describe storage and removal in dry wells that capture a given stormwater runoff volume and infiltrate it to the ground. In P8, infiltration is simply subtracted from the device flow according to the device dimensions and the SCS infiltration rates. However, in reality, infiltration of moderately deep water in a dry well is a three-dimensional problem of unconfined flow from a partially penetrating well and can involve very complex analytical techniques (Freeze and Cherry 1979). Under suitable soil conditions, however, it may be possible to design dry wells that infiltrate the entire design inflow over the duration of a storm or somewhat longer, and thus may be modeled simply by assuming all the water will infiltrate, without becoming bogged down in groundwater modeling efforts. In any event, SCS infiltration rates are very unlikely to be appropriate for infiltration from the bottom and sides of a dry well.

10.2 SIMULATION OF DRY WELLS WITH SWMM

In SWMM, dry wells can be modeled as a pipe or inlet with a specified inflow capacity (in the manner of a combined sewer overflow regulator). This assumes that the well has the capacity to accept the diverted flow from the “regulator.” Another option is to follow exactly the same procedure as outlined in Section 8.3 for infiltration trenches. This would provide one way in which flooding of the well could be simulated if infiltration capacity were exceeded. But for the case of deep dry wells with a small surface area, the one-dimensional model of infiltration is even more inappropriate.

Still another means of simulation is as a storage device with constant or head-driven outflow. This could be done using 1) a Transport Block storage element, 2) a S/T Block storage unit, or 3) an Extran Block orifice or rating curve. (Rating curves in the current Extran must be mimicked using a pump Q vs. h curve, which works well but is not intuitive.) It is unlikely that the quality of the water in the dry well would need to be simulated, but if it were, Extran could not be used. SWMM offers the option for continuous simulation in all cases, with which to characterize the performance and operation of the well.

11 CISTERNS

11.1 INTRODUCTION

Cisterns are usually a barrel or other tank placed beneath a downspout, with outflow controlled by a valve. The difference between a cistern and, say, a dry well, is that the cistern has a fixed capacity, and excess or bypassed runoff must be accounted for. In addition, the emptying of cisterns is more complex from a modeling viewpoint since they are usually drained very deliberately, for use for irrigation, for example. Hence, information on the timing of releases is required.

In a modified version of SLAMM (Pitt and Voorhees 2002), it is possible to designate only a fraction of flow to treatment areas. As an example, a fraction of the roof runoff and driveway runoff can be directed to a cistern for storage for later use during dry weather for on-site irrigation, toilet flushing (gray water), boiler feed water, etc. On-site water treatment might be required to improve quality for some uses, such as gray water. In the rain barrel/cistern “outlet/discharge” option in SLAMM, monthly water uses are entered so the model can track water use and re-filling of the tanks during storms. Hence, a storage accounting method is needed for the cistern storage units including supply and use schedule.

11.2 SIMULATION OF CISTERNS WITHIN SWMM

A cistern may be simulated in the manner of any storage device, but since cisterns usually collect relatively clean runoff from roofs, quality simulation may not be necessary. Outflows and bypasses may be directed downstream in the catchment as well. The principal need in SWMM is the ability to input a water use schedule, as described above for SLAMM.

12 POROUS PAVEMENT

12.1 INTRODUCTION

Porous or permeable pavement is a "hard" surface that can support a certain amount of activity, while still allowing water to pass through. Porous pavement is generally used in areas of low traffic, such as service roads, storage areas, and parking lots. Several different types of porous pavement exist (Pitt and Voorhees 2000). Open mixes of asphalt have a much higher porosity than regular asphalt, and concrete grids can have open holes up to several inches wide, possibly containing sand, gravel or planted with grass. This kind of surface is often marketed as "pavers," i.e., concrete paving stones designed to allow easy passage of water between joints. Porous pavements can be effectively used in areas having soils with adequate percolation characteristics. The percolation rates of the soils underlying the porous pavement installation only need to exceed the rain intensity directly. In most cases, several inches of storage is available in the pavement base to absorb short periods of very high rain intensities. Diniz (1980) states that the entire area contributing to the porous pavement can be removed from the surface hydrologic regime if all runoff infiltrates. Porous pavement can be designed to eliminate all of the runoff from paved areas, and recent tests have found few problems with porous pavement in areas having severe winters (Pitt and Voorhees 2000). Work at the University of Guelph in Ontario (Thompson and James 1995) has shown that porous pavement systems can also be effective filters to remove particulate pollutants from runoff.

Experiments in Bordeaux and Paris, France have shown that porous pavements were very efficient in reducing the pollutant loads discharged into the receiving water (Baladès et al. 1995a,b). These French studies have shown that the pollutant removal efficiencies for suspended solids can be between 50 and 70%, between 54 and 89% for COD, and between 78 and 93% for lead. These reductions were associated with the high amount of infiltration of water, and associated pollutants, through the pavements, away from the surface drainage. These experiments confirm results from previous studies in other countries (Hogland et al. 1987, Pratt et al. 1989, Pratt et al. 1995).

The primary objective of using porous pavements is to mimic natural flow and infiltration conditions as closely as possible. It is therefore very important to pay attention to the following aspects to reduce groundwater contamination potential (Pitt et al. 1996):

- Depth to groundwater
- Groundwater uses
- Risks due to industrial activities in the catchment
- Use and traffic levels on the porous pavement

- Use of de-icing salts on the street

12.2 SLAMM CALCULATION PROCEDURES FOR POROUS PAVEMENTS

SLAMM (Pitt and Voorhees 2000) uses a calculation procedure for porous pavement performance that is almost identical to the general infiltration device procedure of Chapters 8 and 9. However, porous pavements are only assumed to treat the paved area, with no additional flows from upland areas discharging to the pavement.
Therefore:

$$\text{fractional volume reduction} = \left(\frac{f}{i}\right)\left(\frac{A_p}{A_t}\right) \qquad (12\text{-}1)$$

where

f = the percolation (infiltration) rate of the porous pavement: the pavement base, or the soil, whichever is less (depth/time),

i = the rain intensity = total rain/rain duration,

A_p = the paved area, and

A_t = the total study area.

Again, the ratio f/i of Equation 12-1 must be less than or equal to 1.0.

An example follows:
Percolation rate = 3 in./hr
Total rain = 1.7 in.
Rain duration = 6 hrs
Porous pavement area = 0.7 acres
Total study area = 5.3 acres

Therefore i = 1.7 in./6 hrs = 0.283 in./hr
The ratio of f/i therefore = 3/0.283 = 10.6 which indicates an over-design for this rain, requiring the use of 1.0 in the performance equation.

The volume reduction is therefore 0.7 acres/5.3 acres = 0.13 (13% reduction in flow and pollutant yield).

SLAMM documentation does not cite a source for porous pavement percolation rates. The example used in the SLAMM inputs a percolation rate of 3 in./hr. Use of this method to determine porous pavement performance requires the user to have a percolation value, which may be equal to the underlying substrate percolation rate.

12.3 SIMULATION OF POROUS PAVEMENT WITH SWMM

When using SWMM, porous pavement may be treated as a pervious surface and either the Horton or Green-Ampt infiltration equations employed. Because porous pavement installations often have provision for subsurface drainage, laterally away from the paved area, the SWMM subsurface flow routines may be used for this purpose. James et al. (2001) demonstrate how SWMM can be used effectively in this manner. These authors discuss the key parameter choices necessary to simulate surface and subsurface runoff from the current Runoff Block hydrologic routines and include an extensive discussion of parameter selection. For ease of use, an interface for parameter selection has been included in the PCSWMM graphical user interface (www.chi.on.ca). The reader is referred to James et al. (2001) for details. One limitation is that the SWMM groundwater routine does not perform routing of infiltrated quality constituents. Instead, the quality of effluent groundwater is input as a constant concentration in the current SWMM. If linked surface-groundwater quality routing must be performed, HSPF is one model that does this (Bicknell et al. 1997).

13 HYDRODYNAMIC DEVICES

13.1 INTRODUCTION

Hydrodynamic devices range from oil-water separators, which are essentially flotation devices but may be simulated on the basis of distinct pathways for one portion of the flow vs. another portion of the flow, to much more complex and often proprietary devices, such as a swirl concentrator, StormCeptor™, Vortechs™, etc. (Minton 2002 provides a description of several such devices). These latter devices often rely on a vortex or similar secondary flow pattern to separate heavier grit from a cleaner overflow. Modeling these devices by process would be difficult due to the individual variation between devices. When trying to describe performance of this widely varying group, a black box method may be the most realistic choice that identifies volume treated vs. volume bypassed and manufacturer specifications. This approach neglects maintenance, malfunctioning, or poorly sized devices, and would probably reflect maximum treatment rates. The EPA/ASCE BMP Database is currently collecting performance information on such devices. In the future this may be a good source for treatment efficiencies, although currently there are few listed.

13.2 SIMULATION OF HYDRODYNAMIC DEVICES WITH P8

Particle (and associated pollutants) removal from hydrodynamic devices can be modeled in P8 with a particle scale removal factor. Hydrodynamic devices only work for particulates (no dissolved nutrient removal, no decay). Loading rate, particle size, and maximum treatment rates are all of concern when modeling these devices.

Just as this parameter was adjusted in the Pond Section, the Particle Removal Scale Factor allows for the calibration of an increase or decrease in device removal efficiency. This adjusts the removal rates for each device, and is usually set to 1.0. Other values can be used to account for effects of filters or other factors that affect particle removal and can be calibrated to the manufacturer's specifications (essentially outflow concentration = removal factor times inflow concentration). However, the possible usefulness of the particle scale removal factor in SWMM has already been discussed (Chapter 7). It may be preferable to use the tanks in series model (N CFSTRs, Equation 6-11) as a more conventional empirical tool for unit process design.

13.3 SIMULATION OF HYDRODYNMIC DEVICES WITH SWMM

The SWMM S/T Block is currently able to use performance data (i.e., removal equations) to simulate these kinds of devices. The Extran Block is often able to simulate the complex hydraulics of such devices, but without corresponding water quality computations. Nonetheless, flow treated vs. flow bypassed could likely be computed with Extran.

Although the data are seldom presented in terms of overflow rate, because residence time in a secondary flow device increases as volume and surface area increase, so must sedimentation. A brief review of vortex separation by Minton (2002) indicates that performance increases as the diameter of the vortex chamber increases. Hence, removal based on overflow rate (Equation 6-6) will likely work, if performance data are available.

14 CASE STUDY: LID SIMULATION IN PORTLAND

14.1 OBJECTIVES

To aid in identifying strengths and weaknesses of SWMM's ability to simulate LID and BMP options, application to real basins is most useful. Various locations for LID simulation were discussed in Chapter 2, including locations in Portland, Oregon, for which monitoring data are available for both catchments and BMPs, although the latter are more limited, as will be seen in Chapter 15. Advantages of the Portland location were discussed in Chapter 2, including proximity to Oregon State University (OSU), good cooperation and information by the operative agency, the Bureau of Environmental Services (BES), and most of all, an extensive data collection and archival effort (http://www.cleanrivers-pdx.org/).

This chapter discusses a Portland study area used for LID simulation in detail, and the application of SWMM for this purpose. A different Portland area is described in Chapter 15 for a BMP simulation example. In this chapter, the Runoff and Extran Blocks are used to represent a portion of the Sullivan area combined sewer system down to the parcel (individual lot) level using actual rainfall-runoff data monitored by the BES during fall 1998 – spring 1999. The ability of SWMM to simulate LID scenarios is demonstrated. In the next chapter, the Runoff and Transport Blocks are used to perform quantity and quality simulation of a detention pond serving a small catchment in the Lexington Hills area of southeast Portland, including water quality simulation. LID and BMP simulation capabilities and limitations are noted on the basis of these simulations.

14.2 PORTLAND COMBINED SEWER STUDY AREA

The Portland, Oregon Sullivan combined sewer basin is a 1,700-ac area located on both sides of the Banfield Freeway (Interstate 84), from about NE 25th to NE 55th Avenue in northeast Portland. The land use is primarily single-family residential with localized zones of commercial properties. The overall basin imperviousness is about 46%. Detailed information on the Sullivan area and neighboring Stark and Holladay areas is provided by Carollo Engineers (1999). Additional information is provided by Adderley and Mandilag (2000a, 2000b) and Hoffman and Crawford (2000). Additional information on the study area and modeling is given by Huber and Cannon (2002). A portion of the Sullivan Basin is shown in Figure 14-1.

The City of Portland's BES has been modeling this area since the late 1990s. The area has been targeted for the City's Downspout Disconnection Program, and the program has very detailed information on percentages of roof area that are currently disconnected, including a complete description of each parcel. Hence, optimization for LID can be studied using this monitored, real system.

Initial efforts to use BES data resulted in huge amounts of data in formats not readily accessible to researchers at OSU. BES has modeled much of the City's combined sewers and uses MapInfo for their GIS information. With the help of BES personnel, OSU staff were able to define a usable study area for modeling and convert GIS information from MapInfo into the ArcView format usable at OSU. The BES performed much of the basic data preparation described subsequently, and the OSU study described below is a subset of the larger Sullivan area evaluated by the BES.

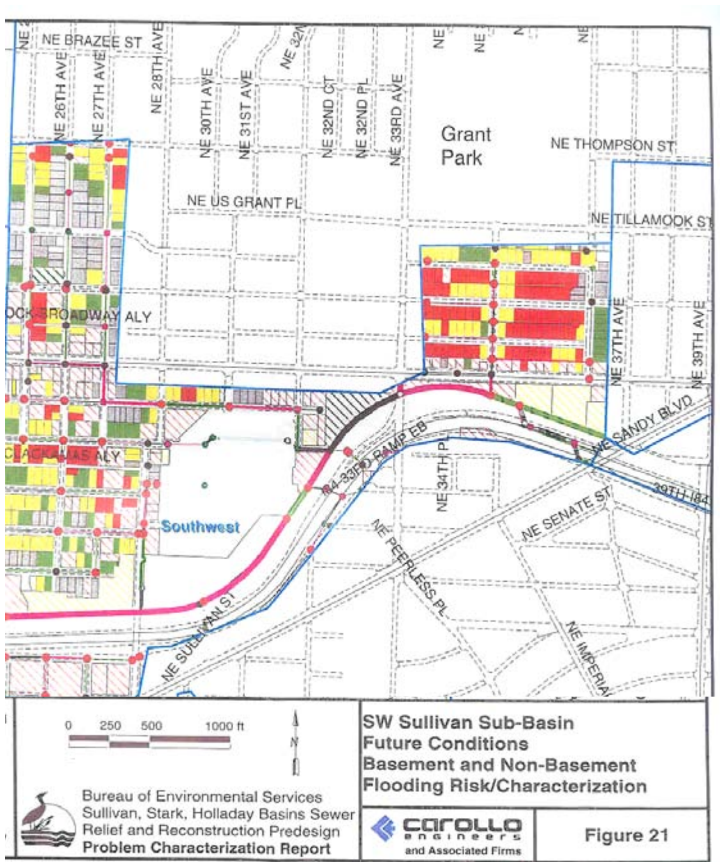

Figure 14-1. Sullivan area (Carollo Engineers, 1999). The sub-area used for this study is just south of Grant Park. Color code relates to basement flooding risk. Red: very high risk; yellow: high risk; green: medium risk; gray: some risk; white: low risk.

The selected study area is between 33rd Ave. and 37th Ave. (west and east boundaries) and between the Banfield Freeway (I-84) on the south and Grant Park on the north (Figures 14-2 and 14-2). A six-block area of 115 single-family residential lots (also referred to as parcels or taxlots) totaling 16.9 acres with 35% impervious area was modeled. Average lot size is just over 0.1 acres. An aerial photo of the area is shown in Figures 14-3, which emphasizes the density of housing and small lot size.

This area has been monitored extensively because of basement flooding in the area. There is a flow monitor in a 16-in. pipe draining the study area and a tipping bucket rain gage a few blocks away (Figure 14-4). Monitoring data are available from November 98 – May 99 at 5-minute intervals. Evaluation of rating curves indicated flows measured at depths greater than 2.3 inches, corresponding to 2 cfs, were more reliable than lower flows; hence most reliance was placed on these higher flows. The ArcView layer in Figure 14-4 also shows the roof density of the neighborhood and the main sewer laterals.

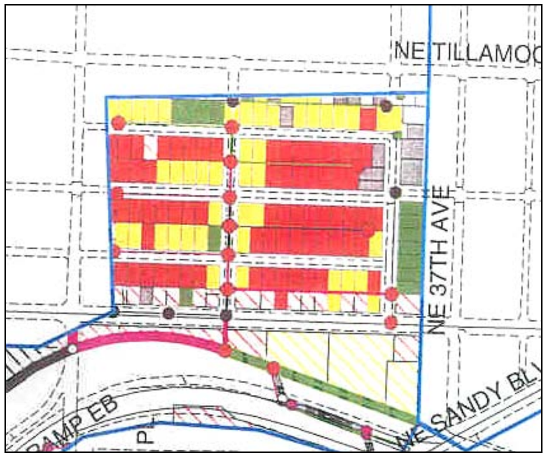

Figure 14-2. Combined sewer study area, south of Grant Park (Carollo Engineers 1999). See color codes in caption to Figure 14-1.

Figure 14-3. Aerial photo of combined sewer study area.

14.3 DATA PREPARATION METHODS

14.3.1 Required Parameters

The SWMM Runoff Block converts rainfall into runoff using a nonlinear reservoir technique (Huber and Dickinson 1988). Each subcatchment is characterized by the following parameters:

- Area
- Imperviousness
- Width
- Slope
- Depression storage (pervious and impervious subareas)
- Manning's roughness (pervious and impervious subareas)
- Three Green-Ampt (G-A) infiltration parameters

These parameters were developed for both an aggregated and disaggregated schematization of the 16.9-acre basin, as described in the following sections. However, the same values for depression storage, roughness, and infiltration parameters were used for all subcatchments, as indicated in Table 14-1. These values are based on BES simulations and data. *All model runs shown are uncalibrated!* Although this report's authors relied on BES estimates for baseflow, there was no attempt to improve upon the parameter estimates described below to obtain better fits, mostly because the initial comparisons were very good. In fact, all comparisons of simulated and monitored flows are generally good, due in part to the relatively high imperviousness of the overall basin.

Table 14-1. Constant model parameters.

Parameter	Impervious Area	Pervious Area
Depression storage, in.	0.03	0.25
Manning roughness	0.013	0.25
G-A suction, in.	n/a	2.56
G-A hydraulic conductivity, in./hr	n/a	1.1
G-A initial moisture deficit	n/a	0.08

The Runoff Block was used only for surface runoff simulation; the sewer network was simulated in the Extran Block. Hence, while most of the discussion that follows deals with the Runoff Block, simulated vs. monitored flows rely upon Extran Block output at the monitor location. Extran block input for all pipes in the system is shown in Table 14-2.

14.3.2 Directly Connected, or DCIA Subcatchments

14.3.2.1 Introduction

Directly connected impervious area (DCIA) consists of the impervious area of each parcel that is directly connected to the sewer through laterals. DCIA subcatchments are delineated for each pipe with service laterals, i.e., for every parcel with a sanitary sewer connection. They contain only the impervious area of a parcel and are therefore 100% impervious (except for the disconnection program described below). It is assumed that the pervious areas of the parcels (and impervious areas not directly connected to the sewer system) drain to the street and are therefore included in the Surface Water (SW) Subcatchments. Although this is generally the case for single-family parcels it may not always be the case for commercial areas – but there are no commercial areas in the small study area.

Table 14-2. Extran Block conduit input data. Conduits are identified in Figure 14-6.

Conduit No.	Upstream Junction	Downstream Junction	Diameter, ft	Length, ft	ZP1*, ft	ZP2*, ft	Manning n
62	62	64	1.33	130	0	0	0.013
61	61	62	0.67	433	0	0.31	0.013
57	57	62	1.17	109	0	0	0.013
58	58	57	0.67	526	0	0.31	0.013
56	56	57	1.17	108	0	0	0.013
60	60	56	0.67	427	0	0.64	0.013
55	55	56	1.00	133	0	0	0.013
54	54	55	0.67	477	0	0	0.013
51	51	55	1.00	129	0	0	0.013
49	49	51	0.67	429	0	0.3	0.013
50	50	51	0.67	97	0	0	0.013
52	52	50	0.67	580	0	0.3	0.013

*ZP = vertical displacement of pipe invert above junction invert. ZP1 = upstream end, ZP2 = downstream end.

14.3.2.2 Delineation

Two primary models will be described below. In one model ("disaggregated"), every individual house parcel (lot) is considered individually, and separated into directly connected impervious area (DCIA) and the remainder (pervious plus non-DCIA imperviousness). Although the computation of DCIA is reasonably precise, through a combination of aerial photos and GIS analysis, it is complicated by Portland's downspout disconnection program (described below), which applies to the study area. The second basic model is one in which DCIA and the remaining surface area is aggregated into 14 bigger subcatchments, to test the effect of aggregation.

Parcels draining to multiple sewers (i.e., along a low ridge, such that front and back yards drain in different directions) were divided into smaller areas, each with its own (sewer) lateral pointer in the GIS. Runoff from the DCIA Subcatchments is typically inserted into the model at the upstream manhole of each major sewer lateral.

14.3.2.3 Impervious Area

The impervious roof and parking areas for each parcel (see Figure 14-4) were obtained from photogrammetric maps of the City. The model was based on actual impervious areas only; no impervious area assumptions were made based on land use. However, the City of Portland has an active downspout disconnection program, with incentives for homeowners of $53 per disconnected downspout (http://www.cleanrivers-pdx.org/get_involved/downspout_disconnection.htm). Hence, some of the roofs and parking lots (none of the latter in this study area) are disconnected from the service laterals and flow to vegetated areas or drywells. Areas with drywells are termed "sumped" areas in text below. DCIA imperviousness was then modified, as follows, based on information provided by the BES that applies to the whole Sullivan area, not just this small study area:

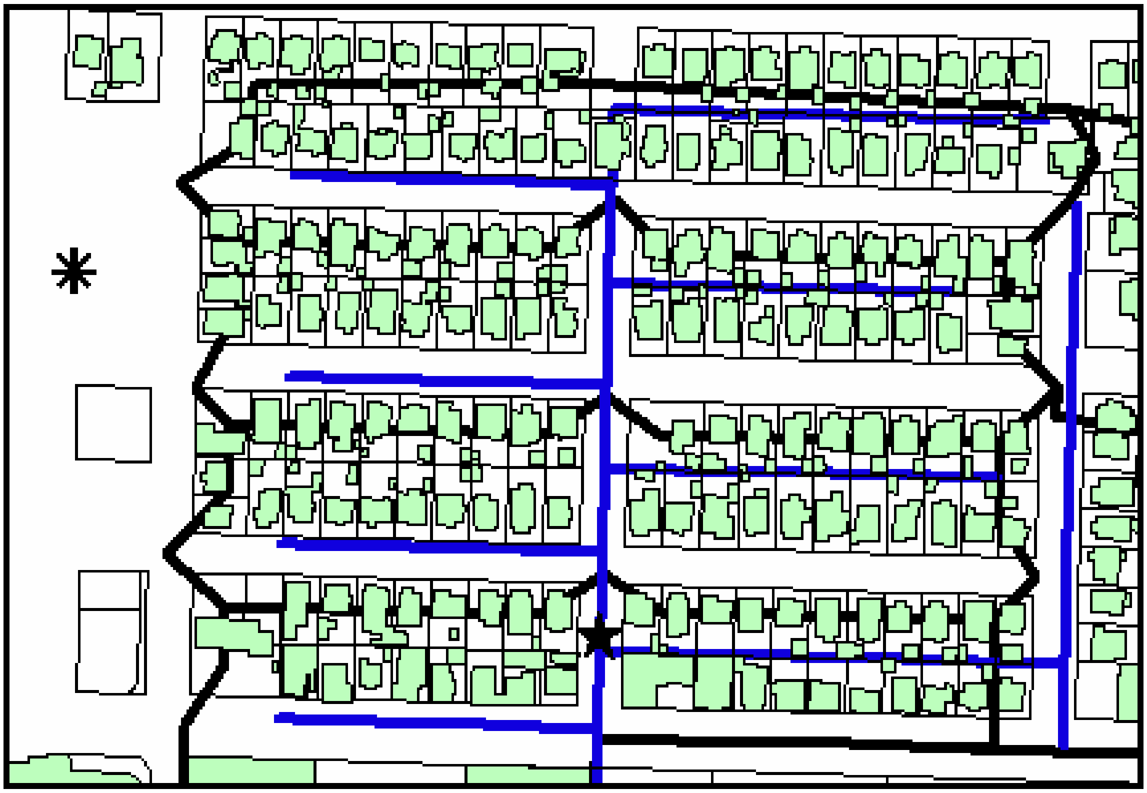

Figure 14-4. ArcView map of study area showing individual house parcels and rooftop imperviousness. Aggregated subcatchments used by BES are shown in heavy black. The sewer network is shown in heavy blue. The monitoring station on NE 35th Ave. is shown with a star and the raingage with an asterisk (*). The top (north) east-west street is Tillamook, the next is Hancock, and the east-west street above the monitoring station is Schuyler. The north-south street on the east is 37th Ave. The north-south street on the west, not drawn, would be 33rd Ave. Dimensions of the figure border are approximately 1050 ft (320 m) wide by 590 ft (180 m) high.

1) Twenty percent (20%) of single-family roofs are disconnected from the combined sewer in unsumped areas. This disconnection is assumed to be 70% effective (i.e., some of the disconnected water flows over the curb and into the sewer through a street inlet). The effective disconnection rate accounting for both of these factors is 14%, with the surface water subcatchment receiving the remaining 6%, as discussed in the next section.
2) Thirty percent (30%) of single-family roofs are disconnected in sumped areas, and this disconnection is 100% effective since the sumps (drywells) are assumed to have capacity to accept all the roof runoff.
3) Twenty percent (20%) of commercial roofs and parking lots are disconnected to a drywell in sumped areas, but this is irrelevant to this small study area.

Based on disconnection survey data, the impervious area for the DCIA Subcatchments was computed by the BES as outlined below. The following equations use the existing impervious areas (measured from aerial photos using the GIS) and assumed disconnection rates to calculate the impervious area, subcatchment area, and impervious percentage of each parcel that is directly connected to the sewer (assuming no commercial or parking areas in the catchment). That is, each DCIA area is scaled back uniformly for each lot over the study area.

In an unsumped area:
Impervious Area = 0.86 x Area Single Family Roofs (14-1)

In a sumped area:
Impervious Area = 0.70 x Area Single Family Roofs (14-2)

The assumptions are incorporated into the Runoff Block subcatchment data supplied by BES and incorporated into the SWMM runs described subsequently. The remaining area of each parcel may include some imperviousness and is described below under “Surface Water Subcatchments” (SW Subcatchments). DCIA runoff enters the sewer system in the model at the upstream end of each main lateral.

14.3.3 Surface Water Subcatchments

14.3.3.1 Introduction

Surface Water Subcatchments (SW Subcatchments) are delineated for each of six sub-basins. Street and sidewalk pavement and pervious areas from the parcels are included in the SW Subcatchments. All surface water runoff is collected by sub-basin and delivered to the six corresponding nodes. When this is broken down to the parcel level, each parcel has a SW Subcatchment associated with it. However, they do not include the impervious areas (DCIA) of each parcel that are connected to the sewer through service laterals (i.e., sewer pipes connected to individual homes). These areas are already included in the DCIA Subcatchments, as discussed in the previous section. Delineation of subcatchments and determination of impervious area, width, and slope with the GIS are discussed below.

14.3.3.2 Delineation

The SW Subcatchments were delineated by the BES with the aid of a digital terrain model (DTM). The DTM was created from contour maps of the project area using Vertical Mapper, a third party application for MapInfo. Slope and aspect grids were created from the DTM. The aspect grids show the direction that each grid drains expressed as degrees from north. A vector representation of the aspect was created for each grid to assist with delineation of the SW Subcatchments; an example is shown in Figure 14-5.

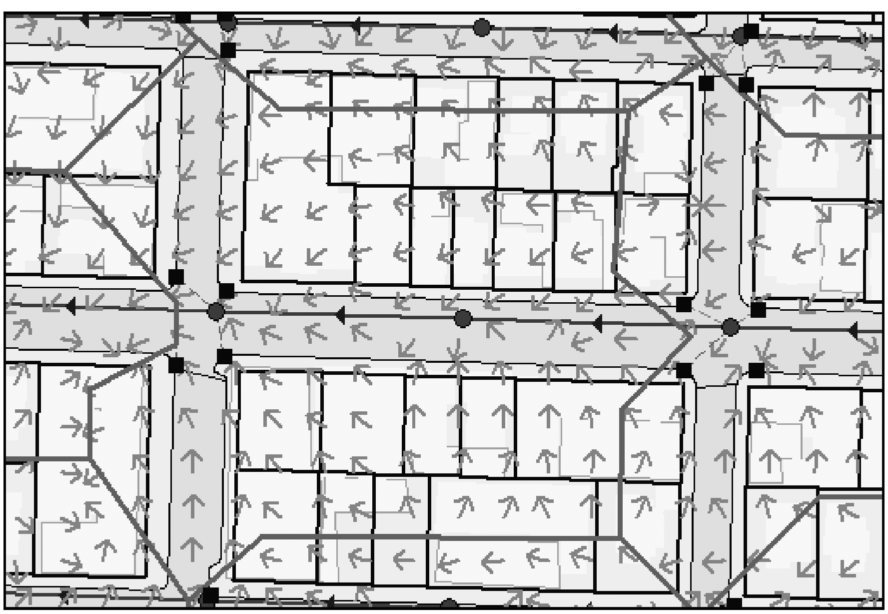

Figure 14-5. Example slope and aspect grids from the digital terrain model.

14.3.3.3 Impervious Area

SW Subcatchments with infiltration sumps that were installed prior to the date being modeled were given a data flag so that they would not be included in the SWMM Runoff model. This assumes that 100% of the runoff is taken by the sump and removed from the combined sewer system. Otherwise, parcel imperviousness was computed for SW Subcatchments as follows:

Unsumped area:

Parcel Impervious Area = Inefficient Portion of Disconnected Single Family House
= 0.14 x Area Single Family Roof (14-3)

Sumped area:

Parcel Impervious Area = 0 (14-4)

For both areas:

Parcel Pervious Area = Area of Parcel – Impervious Area (14-5)

Imperviousness in sumped areas is just the street pavement since the sumps are assumed to be 100% effective. The pervious area includes sidewalks and driveways that drain onto adjacent lawns. “Parcel Impervious Area” above is treated like DCIA in the model.

Aggregated and disaggregated subcatchment models were run, as described below. For the aggregated simulation,

Impervious Area = Parcel Impervious Area + Street Pavement Area (14-6)

14.4 THE MODELS

14.4.1 Two Model Types

Two models have been created for this area:
1. Aggregated subcatchment model (A-Model): 11 DCIA subcatchments plus three combination (pervious plus impervious) subcatchments = 14 total subcatchments. The main purpose of these runs was to compare OSU’s efforts with prior BES efforts, and to compare aggregated vs. disaggregated simulation results.
2. Disaggregated subcatchment model (I-Model): 115 house parcels (containing DCIA and pervious area) plus three street subcatchments =118 total subcatchments). The purpose of these runs was to determine the added value of a highly discretized and detailed subcatchment schematization and to be able to separate impervious from pervious area more accurately.

Both models contain three sub-basins (moving north to south) that correspond to the three east-west cross streets (north to south: Tillamook, Hancock and Schuyler, Figure 14-4). Each sub-basin has a north-south sewer main line and two east-west service laterals on either side of the main line (Figure 14-6). Pipe sizes range from 6 to 16 inches throughout the study (Table 14-2). The street and sidewalk components for the aggregated and disaggregated models are shown in Figure 14-7.

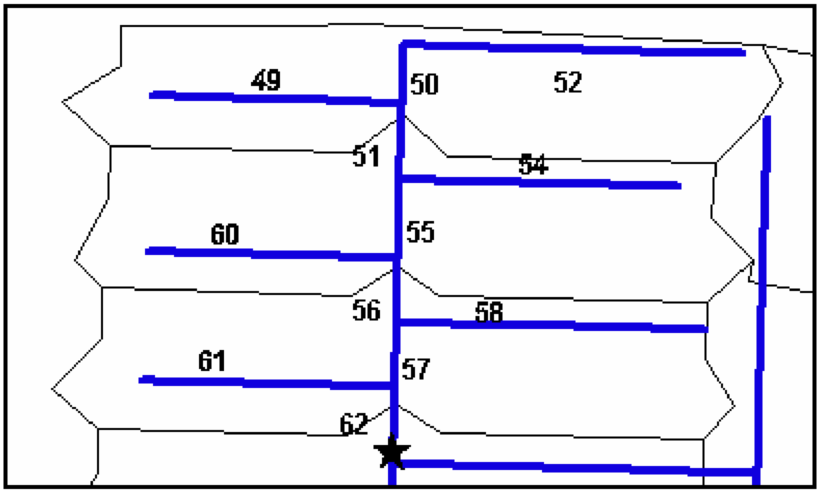

Figure 14-6. Pipe segment ID for the study area, as simulated in Extran. The aggregated subcatchment model (A-model) includes the DCIA in parcels in the six areas draining to laterals 49, 52, 54, 58, 60 and 61. Five smaller DCIA areas (not shown) drain to north-south trunk sewer segments 51, 55, 56, 57 and 62. For the aggregated subcatchment model, streets, sidewalks and all pervious areas are lumped into three surface subcatchments. The street and sidewalk components of these surface subcatchments are shown in Figure 14-7.

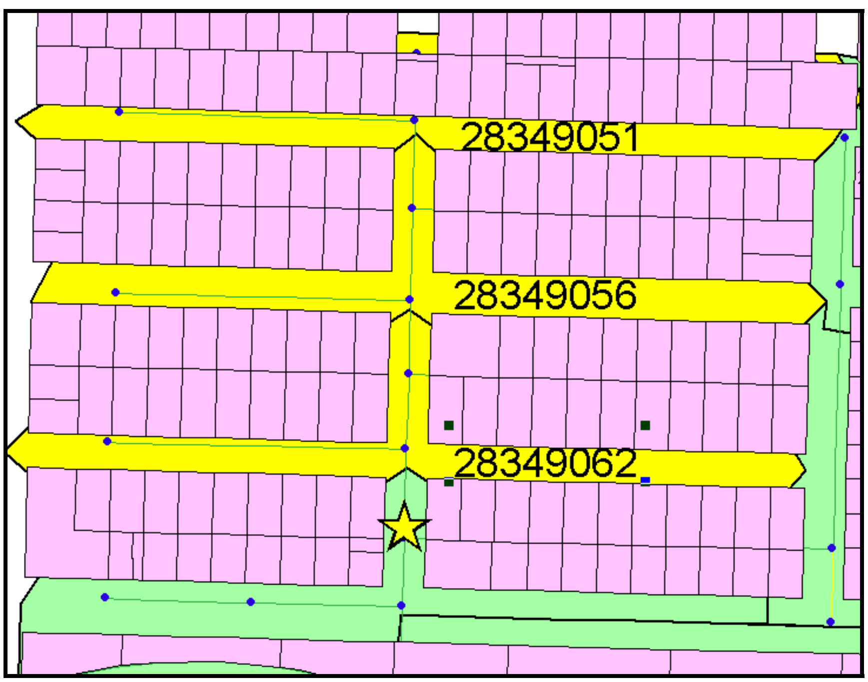

Figure 14-7. Aggregated areas for three street subcatchments include roads, sidewalks and grass strips. The three areas shown are the actual areas used for the disaggregated modeling and conceptual areas for the aggregated modeling, since the aggregated modeling adds all pervious area from house parcels into the three surface subcatchments. Subcatchment 28349051 drains to pipe 51 (Figure 14-6), etc. (The first four digits of the subcatchment IDs are not included in the model data.)

Model A aggregates all DCIA for each of the six main laterals (49, 52, 54, 58, 60 and 61 in Figure 14-6) into one DCIA Subcatchment, plus just one SW Subcatchment for the entire sub-basin. SWMM input data for Model A are heavily based on earlier BES SWMM runs. The SW Subcatchment aggregates all surface water in the sub-basin (over both pervious and impervious areas) and concentrates it into the sewer mainline for the sub-basin (see example in Figure 14-8). The three SW Subcatchment for the I-Model are shown in Figure 14-7. Model I takes the catchment down to the parcel level, as shown in Figure 14-9. SWMM input data for Model I were prepared entirely by OSU personnel.

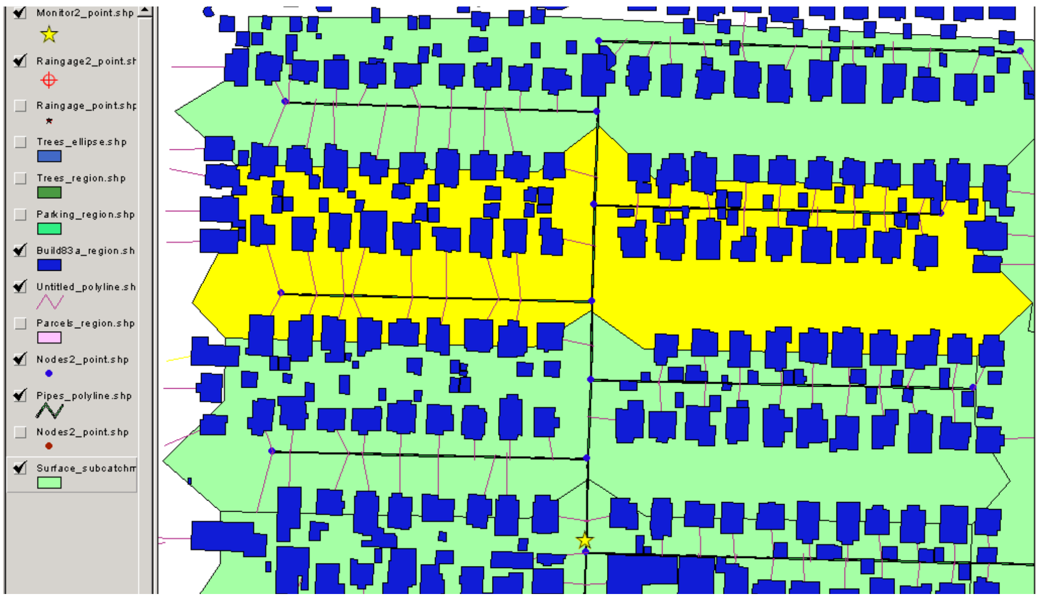

Figure 14-8. The highlighted sub-basin 28349056 (referred to as 56) has four DCIA Subcatchments: one for each pipe segment, main line (1) and lateral (2), with DCIA, and one SW Subcatchment for all the remaining area.

Figure 14-9. Individual parcels for the disaggregated model.

Although it may appear that subcatchment 2268 should drain south (to the bottom of Figure 14-9), in fact, the surface drainage is north across subcatchments 2306 and 2270, as indicated in Figure 14-10. Similarly, it appears in Figure 14-9 that the DCIA of subcatchments 2268 and 2305 drains to the sewer line below the monitor, but according to the BES this is not true and is only an artifact of the MapInfo schematic.

The SWMM Runoff Block simulates a subcatchment as having pervious and impervious (DCIA) subareas. Alternatively, each subcatchment could be split into two, for each surface type. The two methods were compared for the I-Model runs, that is, 115 parcel subcatchments with both pervious and impervious subareas vs. 115 pervious subcatchments plus 115 impervious subcatchments totaling the same area. (Because some individual parcel subcatchments were already 100% impervious, the total number of subcatchments for the latter option was actually 216 + 3 = 219 instead of the expected 115 + 115 + 3 = 233.) Since results were identical, the 115 combined land surface subcatchments were used for most of the simulations. The parcel sub-area distinction is shown in Figure 14-11. Runoff from both the impervious and pervious subareas is routed to the upstream end of the main sewer lateral for the street. Remaining area in the sub-basin not associated with an individual plot (streets, sidewalks, grass strips, etc.) is aggregated as one of three additional SW Subcatchment (Figures 14-7 and 14-8) and routed to the top of the sewer lateral serving the sub-basin.

All DCIA that is within the sub-basin but directly connected to a lateral outside of the study area was omitted in the I Models (Figure 14-12), which had the effect of reducing the total area from 16.9 acres in the A Models to 16.4 acres in the I-Models. Surface runoff from the portion of these plots draining into the modeled sub-basins is included in the aggregated SW Subcatchment (Figure 14-13).

Figure 14-10. Close-up of parcels draining to pipe 62. Monitor (star) is actually downstream of entry of DCIA drainage from subcatchments 2268 and 2305. Likewise, the surface area of these two subcatchments drains north (up in figure) to enter pipe 62 above its drainage divide.

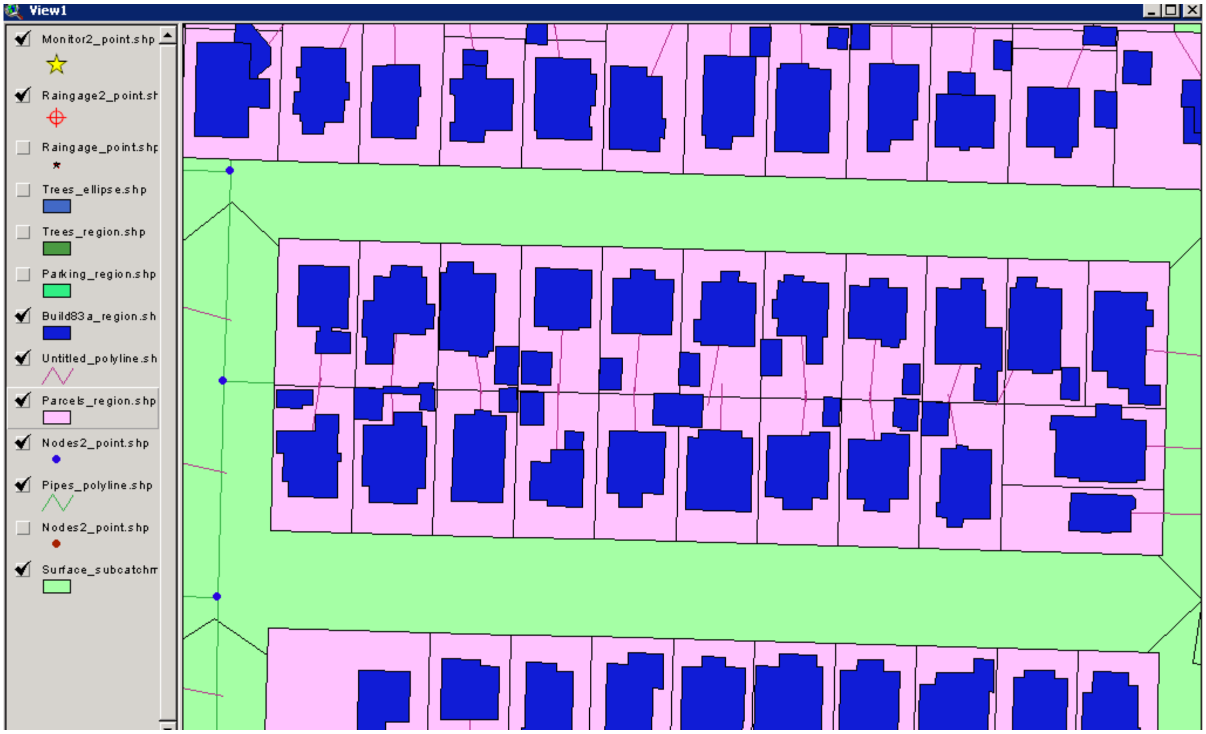

Figure 14-11. Dark areas (blue) are DCIA and are surrounded by lighter (lavender) pervious areas. Both subareas are represented for each parcel in the disaggregated modeling. For the aggregated modeling, the pervious (lighter) areas are combined and added to the appropriate one of five street subcatchments and the DCIA (darker) areas are combined to form six DCIA subcatchments.

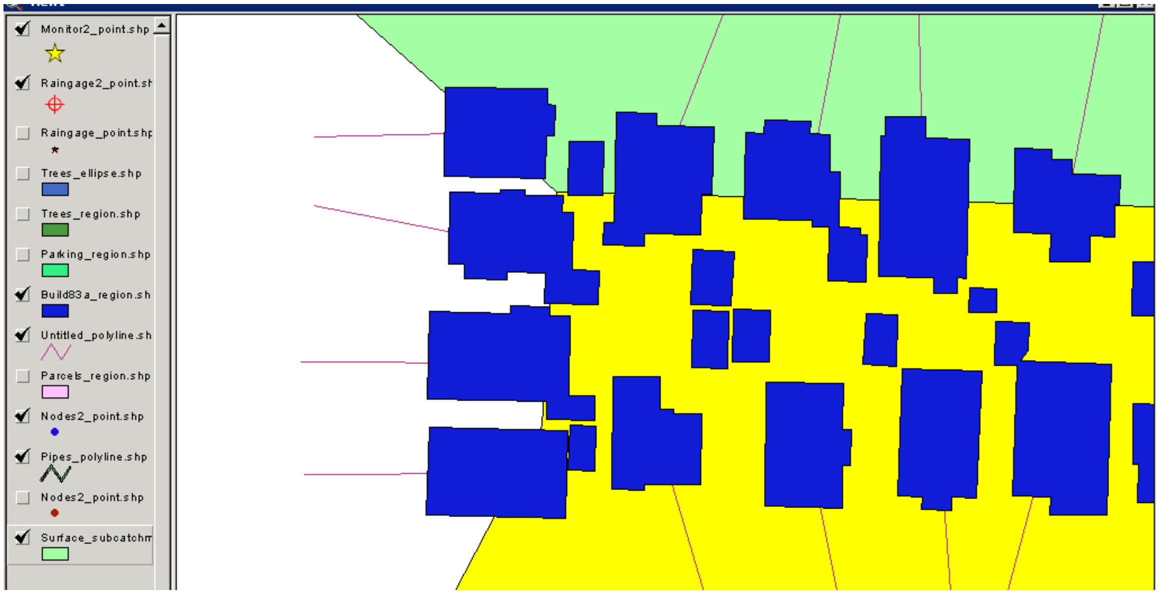

Figure 14-12. The dark (blue) DCIA connected outside the sub-basin is not included in the model as the runoff effects are not noticed at the monitor. The DCIA connected to service laterals in the highlighted sub-basin is all modeled within individual DCIA Subcatchments (I-Models) or summed for the appropriate aggregated subcatchment (A-Models).

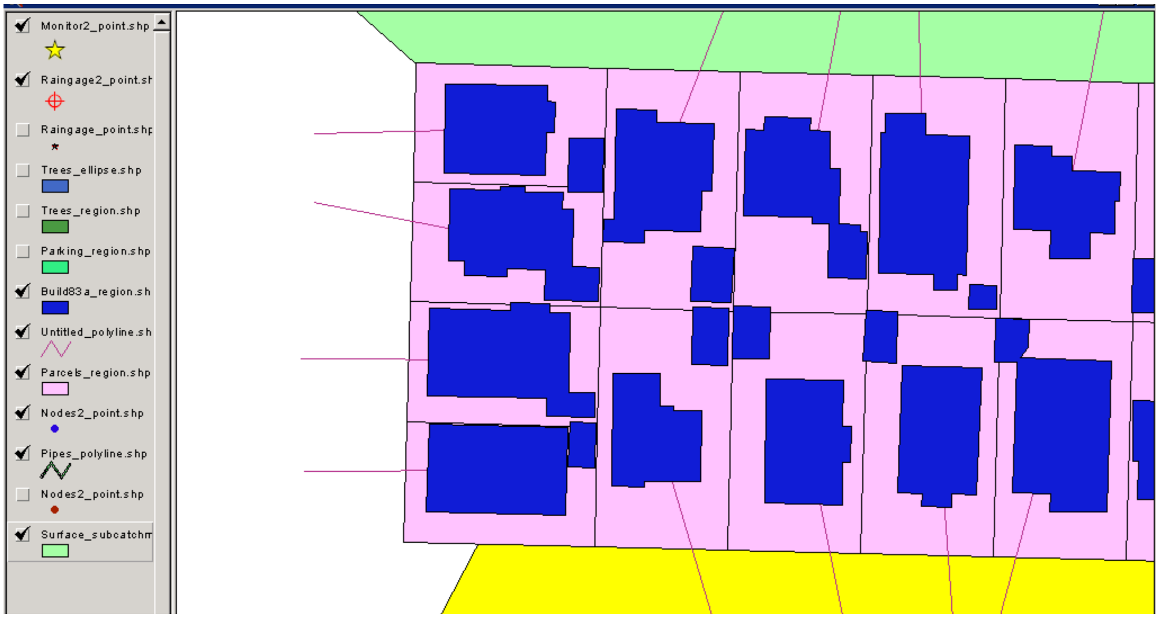

Figure 14-13. Parcel land surface definitions. The pervious (light, pink) parcel area adjacent to the highlighted sub-basin (yellow, at bottom) is included in the aggregated SW Subcatchment for the sub-basin in the A-Model. Pervious area (within the watershed, Figure 14-11) is included within each of the 115 individual parcels for the I-Model.

14.4.2 Width and Slope

The Runoff Block requires a modeling parameter *width* to indicate the shape or flow path of the subcatchment.

Width = Parcel Area/Flow Length (14-7)

where, for individual parcels in the I-Models, Flow Length was assumed to be:

Flow Length (ft) = 25 ft of overland flow +
subcatchment house connection length/6 (14-8)

This equation for width generally attempts to discount the channelized flow in the pipe and make the DCIA Subcatchments concentrate faster then the SW Subcatchments. In the I-Model runs, most widths were rounded to 50 ft in order to facilitate sensitivity analysis. The width parameter does not have a great influence on model results for these relatively small subcatchments, since the time of concentration of the subcatchment is much less then the duration of most storms. Hence, equilibrium (peak) flow is reached for each hyetograph interval regardless of the width.

For aggregated A-Model subcatchments, the length of overland flow was assumed to be 150 ft of flow from the back of a lot to the street plus the distance from the farthest pavement point in the subcatchment to the surface water inlet. The distance from the farthest pavement point was determined with the GIS by using a routine to create points at 5-ft intervals along a street centerline map and querying for the farthest point within the subcatchment. Again, Equation 14-7 was used to compute the widths for the aggregated subcatchments.

The slope of all DCIA Subcatchments is assumed to be 6%, a combination of single-family roof slopes and lawns. The slope of each SW Subcatchment was calculated from the slope grid (Figure 14-5). The average slope of all of the 40-ft grids within the subcatchment was used as the subcatchment slope.

14.4.3 A-Model Input Data

Because there are only 14 subcatchments for the aggregated model, Runoff Block subcatchment data are shown for this simulation in Table 14-3. This includes the three SW Subcatchments (labeled with 9000-numbers) and the 11 DCIA subcatchments (labeled with 8000-numbers). The numbering scheme corresponds to the last two numbers of the sewer segments shown in Figure 14-6. For instance, SW Subcatchment 9056 flows to pipe 56 and drains the entire middle, yellow-highlighted sub-basin shown in Figures 14-7 and 14-8. DCIA Subcatchment 8060 drains to pipe 60, etc. All DCIAs drain to the top end of the lateral. It is assumed that the pervious areas of the parcels drain to the street and are therefore included in the surface water catchments. The remainder of each of the three main sub-basins is aggregated into one SW Subcatchment representing all pervious areas and the impervious area not directly connected to the sewer system (but including the streets, since this was the scheme used by BES). These drain to the sewer mainline for the sub-basin. All slope, hydraulic connectivity, infiltration, and other parameter variables (Table 14-1) are taken from BES data.

Table 14-3. Subcatchment input data for aggregated models (A-Models). Other parameters are the same for all subcatchments and listed in Table 14-1.

Subcatchment ID	Flow to pipe:	Width, ft	Area, ac	Imperviousness, %	Slope
9051	51	248	4.38	16.8	0.077
9056	56	276	4.7	18.1	0.072
9062	62	274	4.6	18.2	0.08
8049	49	190	0.42	100	0.06
8051	51	18	0.02	100	0.06
8052	52	181	0.51	100	0.06
8054	54	226	0.54	100	0.06
8055	55	21	0.02	100	0.06
8056	56	25	0.02	100	0.06
8057	57	36	0.04	100	0.06
8058	58	221	0.57	100	0.06
8060	60	215	0.47	100	0.06
8061	61	245	0.55	100	0.06
8062	62	116	0.12	100	0.06

Since the basin is a combined sewer area, baseflow (dry-weather flow) needs to be considered. BES data indicate an average of about 0.032 cfs at the monitor. The BES provided distributed baseflow estimates for each sub-basin, and these were added to the upstream end of each lateral in the Extran simulation. However, the 0.032 cfs baseflow is almost impossible to discern during storm events.

14.5 MODELING RESULTS

Uncalibrated SWMM output, including baseflows, for January 13-18, 1999 for the aggregated and disaggregated model representations is shown in Figure 14-14. This time period corresponds to a high 5-

day rainfall total of 2.42 in. (61 mm) and was used because BES focused strongly on this time period during their modeling efforts. There is good correlation with monitor flow data, but the comparison is difficult to make visually when the five days are compressed into just one plot. Hence, the event of January 17 is considered in more detail in Figure 14-15, wherein it may be seen that there is practically no difference between the aggregated and disaggregated model simulations. This is good news for continuous modelers, for whom an aggregated model representation will require much less computer time.

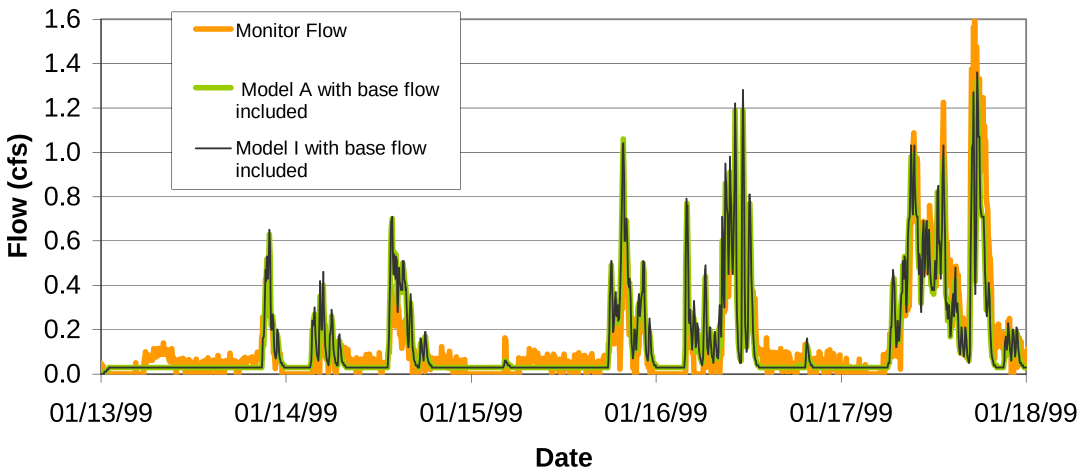

Figure 14-14. Five-day comparison of simulated and measured flows at the monitoring site. A visual comparison is difficult because of the crowded scale.

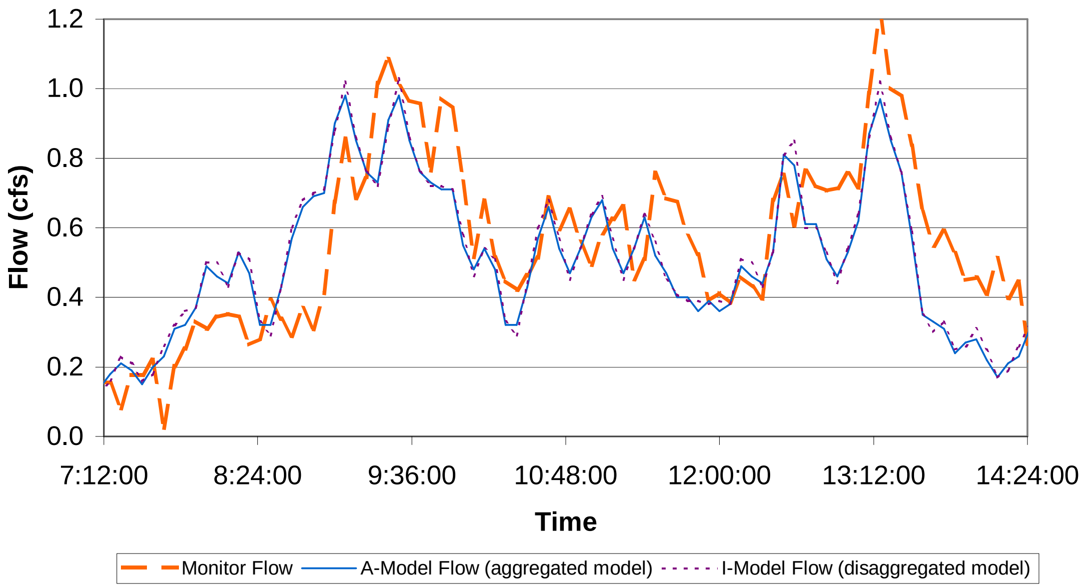

Figure 14-15. Simulated and measured flows for seven-hour period on January 17, 1999. The aggregated and disaggregated models show very little difference.

Generally, the model vs. monitor comparison is good with regard to shape, but the rising limb of the modeled hydrographs is somewhat high, and the modeled recession limb is somewhat low. This might be better simulated by detaining a little more water on the land surface prior to letting it run off. Recall that no calibration or adjustment has been attempted. Extran continuity is very good for all runs, on the order of 0.4%.

Another method for comparison is total runoff volume; these values are shown in Table 14-4. Baseflow is necessary in order to obtain close to the monitored runoff volume. Runoff volumes are comparable for the aggregated and disaggregated models, as would be expected from the inspection of the hydrographs. This confirms the utility of aggregated simulations for long-term simulation.

Table 14-4. Model total flow comparison for five-day event, Jan 13-Jan 18, 1999. In bottom row, K_s is the saturated hydraulic conductivity. (After Huber and Cannon 2002.)

	Total flow comparison for 5-day event, Jan 13-Jan 18, 1999		**7-hour interval, 7:00 am to 2:00 pm, January 17, 1999**	
	(cubic ft)	(in.)	(cubic ft)	(in.)
Rainfall for event	148,987	2.42	37,555	0.61
Monitored flow	58,130	0.94	14082	0.23
Baseflow volume, @ 0.032 cfs	13,824	0.22	806	0.01
Model A (aggregated catchments)				
No baseflow	47,244	0.77	12,492	0.20
With baseflow included	60,459	0.98	13,365	0.22
Model I (disaggregated, separate subcatchment for each parcel)				
With baseflow included	60,492	1.02	13,416	0.23
Model I-LID (re-routing impervious runoff over pervious area)				
With baseflow included	26,829	0.45	4,494	0.076
Same as above but with K_s = 1/10 previous value	27,762	0.47	4,872	0.082

14.6 LID SIMULATION

The essence of LID is to retain as much water on site as possible at the parcel level (Prince George's County 2000, Wright et al. 2000). One essential technique is to minimize direct connections to the drainage system by routing runoff from roofs, driveways, etc. over pervious areas to promote infiltration. The SWMM Runoff Block has been adapted for this purpose by Huber (2001a) as discussed in Section 4.7. As indicated in Figure 4-1, overland flow can be rerouted internally within subcatchment subareas (i.e., from pervious to impervious and vice versa) and also may be routed from one subcatchment onto another. The simulation shown below simply routes the impervious area runoff from each of 115 parcels over the pervious area within the same parcels, sort of a massive, hypothetical LID effort for the neighborhood.

The I-LID simulation uses the I-Model as the base and reroutes 4.0 ac of impervious area runoff from rooftops and driveways over 7.9 ac of pervious areas of each parcel. Street surface runoff (and sidewalk and grass strip runoff from the three street subcatchments is unaffected. A drastic reduction in the runoff hydrograph (Figure 14-16) and runoff volume (Table 14-4) is produced by the LID option. The 5-day runoff volume is reduced by 56% and the 7-hr runoff volume by 67% (Table 14-4). This is to be expected for this hypothetical simulation for which the saturated pervious area hydraulic conductivity of 1.1 in./hr will accept any intensity of runoff associated with typical western Oregon rainfall, including the additional non-DCIA runoff from the roofs, etc.

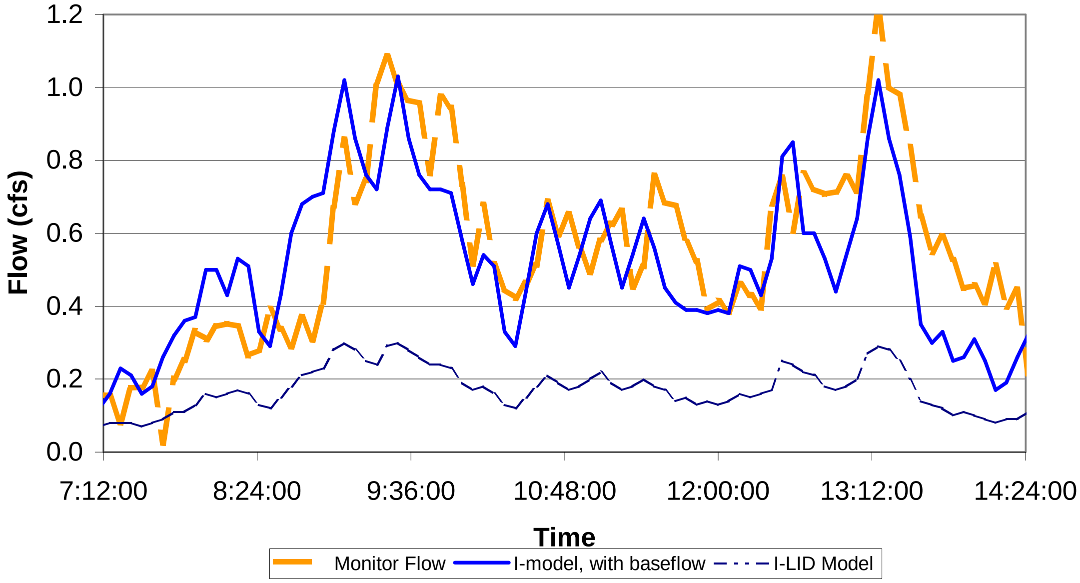

Figure 14-16. Comparison between Model I and Model I-LID simulations for a seven-hour interval, January 17, 1999.

Soil is often compacted in urban developments, with a lower hydraulic conductivity than for pre-development conditions (Pitt et al. 1999a). An additional run was made with a value of saturated hydraulic conductivity, K_s, equal to one-tenth of the value used for the rest of the modeling. That is, the new value of K_s = 0.11 in./hr compared to the previous value of K_s = 1.1 in./hr. The much lower hydraulic conductivity resulted in only slightly higher runoff volumes (Table 14-4) for the 5-day and 7-hr duration events. (The hydrograph is also a little higher but very close to the I-LID hydrograph shown in Figure 14-16 and has not been plotted.) This reflects generally low rainfall intensities in western Oregon (typically less than 0.11 in./hr, but of long duration). Thus, an even lower infiltration rate would be necessary to reduce the effectiveness of the LID option simulated. This is to say that this LID option is likely to be even more effective in climatological regions with rainfall (and runoff) intensities that are characteristically low in magnitude.

This LID simulation is not completely hypothetical. Disconnected flows due to the downspout disconnection program are directed to dry wells. Although this program mainly affects rooftop runoff, its impact could be significant if implemented over a large area.

Infiltration is assumed to remove quality constituents as well, either through advection of dissolved constituents into the soil, or by sedimentation of particulates as water enters the soil. Hence, infiltration is effective in cleaning up surface water. There may be concern, however, about the impact of infiltration on groundwater (Pitt et al. 1996).

14.7 SUMMARY AND CONCLUSIONS

Version 4.4h of SWMM has been applied to a 16.9-ac combined sewer catchment in the Sullivan Basin in northeast Portland, Oregon. With the aid of prior modeling work performed by the Portland Bureau of Environmental Services, the SWMM Runoff and Extran Block simulation of the area compares well with monitoring data for a 118-subcatchment disaggregated simulation (I-Model) of every house parcel (tax lot). The simulations are uncalibrated and could easily be improved through additional effort, including a better definition of baseflow in the combined sewers. In addition, the disaggregated simulation of every individual house parcel compares well with an aggregated simulation that uses only 14 subcatchments to simulate all the parcels, pervious areas and street surfaces. This suggests that long-term modeling (i.e., continuous simulation, if performed) can conveniently be done with a less detailed model representation.

When a typical LID option of routing non-directly connected impervious area runoff over pervious areas is simulated, the hypothetical SWMM Runoff Block output indicates an expected reduction in discharge. Although no hypothetical quality simulation was performed for this study area, quantity reductions by infiltration induce corresponding quality reductions (Huber 2001a,b).

To summarize the key points presented in this chapter (Huber and Cannon 2002):

- SWMM may be used to simulate LID options that involve routing of runoff from non-directly connected impervious area (non-DCIA) over pervious areas, e.g., roof and driveway runoff over lawn surfaces.
- SWMM may also be used to direct surface runoff from one subcatchment over another.
- Simulation of an aggregated model representation (14 subcatchments) performed about as well as a disaggregated model representation (118 subcatchments) for the Portland, Oregon test area.
- A hypothetical LID option that infiltrates all roof and driveway runoff in the dense Portland study area is predicted to result in over a 50% reduction in runoff volumes and peak flows in the combined sewer system. Although a reduction would be expected, the modeling allows a better quantification of the potential results.

15 CASE STUDY: BMP SIMULATION IN PORTLAND

15.1 OBJECTIVES

A detailed example of a hypothetical LID simulation in a real, well-monitored Portland, Oregon catchment was presented in Chapter 14. A similar, if somewhat less detailed example is presented in this chapter for simulation of a detention pond, representing a very typical BMP (Chapter 6). Once again, excellent cooperation was received from Portland BES personnel, including extra help with interpretation of data and graphics. Additional details of the simulation are provided by Stouder (2003).

15.2 LEXINGTON HILLS AREA BACKGROUND

The modeled area was the Lexington Hills BMP area, which is an area with a constructed pond located in the Johnson Creek drainage in southeastern Portland (Liptan 2001). The actual site is just to the west of SE 162nd Avenue, approximately ¼-mile south of SE Foster Road (Figure 15-1). While there are two ponds on the site and one planned, data was collected only for the northernmost pond (Pond 3), constructed in 1996, which is located at SE 162nd Avenue and SE Flavel Drive. Aerial photographs of the pond area are shown in Figures 15-2 and 15-3.

Although during the wet season there can be a small pool below the orifice outlet, Pond 3 is intended to act as extended detention and is designed to fill for a storm size of 0.83 in. of rainfall in a 24-hr period. It receives runoff from a 26.57-ac residential neighborhood with 8.75 ac of (Figure 15-4) impervious area. The pond has approximately 3,000 ft^3 of dead storage plus another 8,000 ft^3 of active storage (Figure 15-5) above the outlet orifice. Influent stormwater enters via a 24-in. concrete pipe, and outflow is discharged through a 6-in. PVC pipe fitted with a 2.5-inch reducer (orifice), designed to empty the live storage (extended detention storage) in about 24 hrs. There is also an overflow weir. Both the orifice and the weir then discharge into a stilling well. After the stilling well, water is routed approximately 75 ft down an open channel to Kelly Creek, a tributary to Johnson Creek.

In 2000 and 2001, pollutant samples were taken by the Portland BES at the inlet and outlet of the pond (Figure 15-6) during seven different storm events (Liptan 2001) in order to test the pond's effectiveness at pollutant removal. Storm events 1,4,5,6 and 7 were the focus of the SWMM modeling. Events 2 and 3 were discarded due to minimal rainfall. All storms except event 7 were selected by the BES so that 24-hour antecedent rainfall was minimized in order to get results for a "first flush" of pollutants. The sampling dates are shown in Table 15-1.

Table 15-1. Storm events used in SWMM simulations.

Event	Date	Event Rain*, in.	Pond inflow, ft^3	Pond outflow, ft^3
1	4/13/00	0.73	8,275	8,075
4	10/9-10/00	0.81	8.276	6,815
5	3/1/01	0.49	7,538	6,217
6	5/14/01	0.56	10,688	13,364
7	5/15-16/01	0.30	7,898	8,917

*Holgate rain gage.

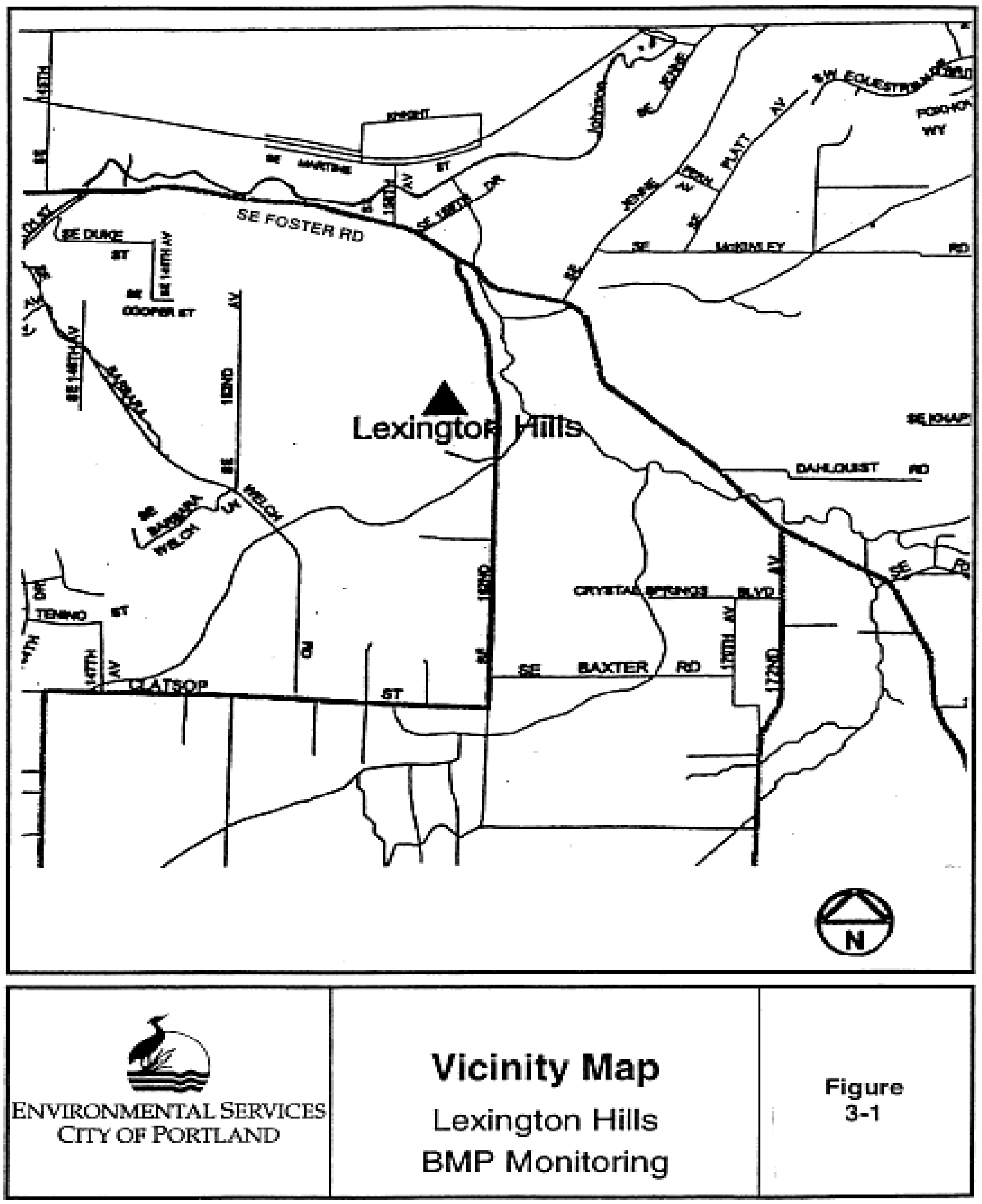

Figure 15-1. Location map for Lexington Hills BMP site (Liptan 2001).

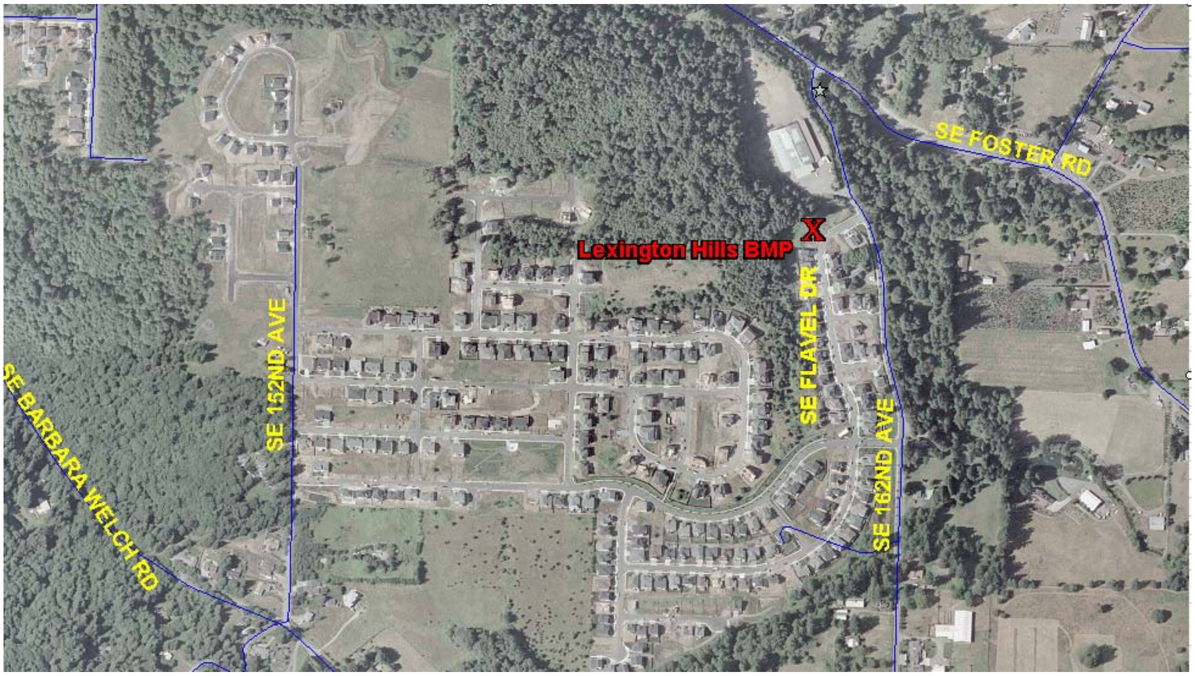

Figure 15-2. Lexington Hills pond vicinity photo (Liptan 2001).

Figure 15-3. Lexington Hills pond site photo (Liptan 2001). Pond 3 is to left of intersection.

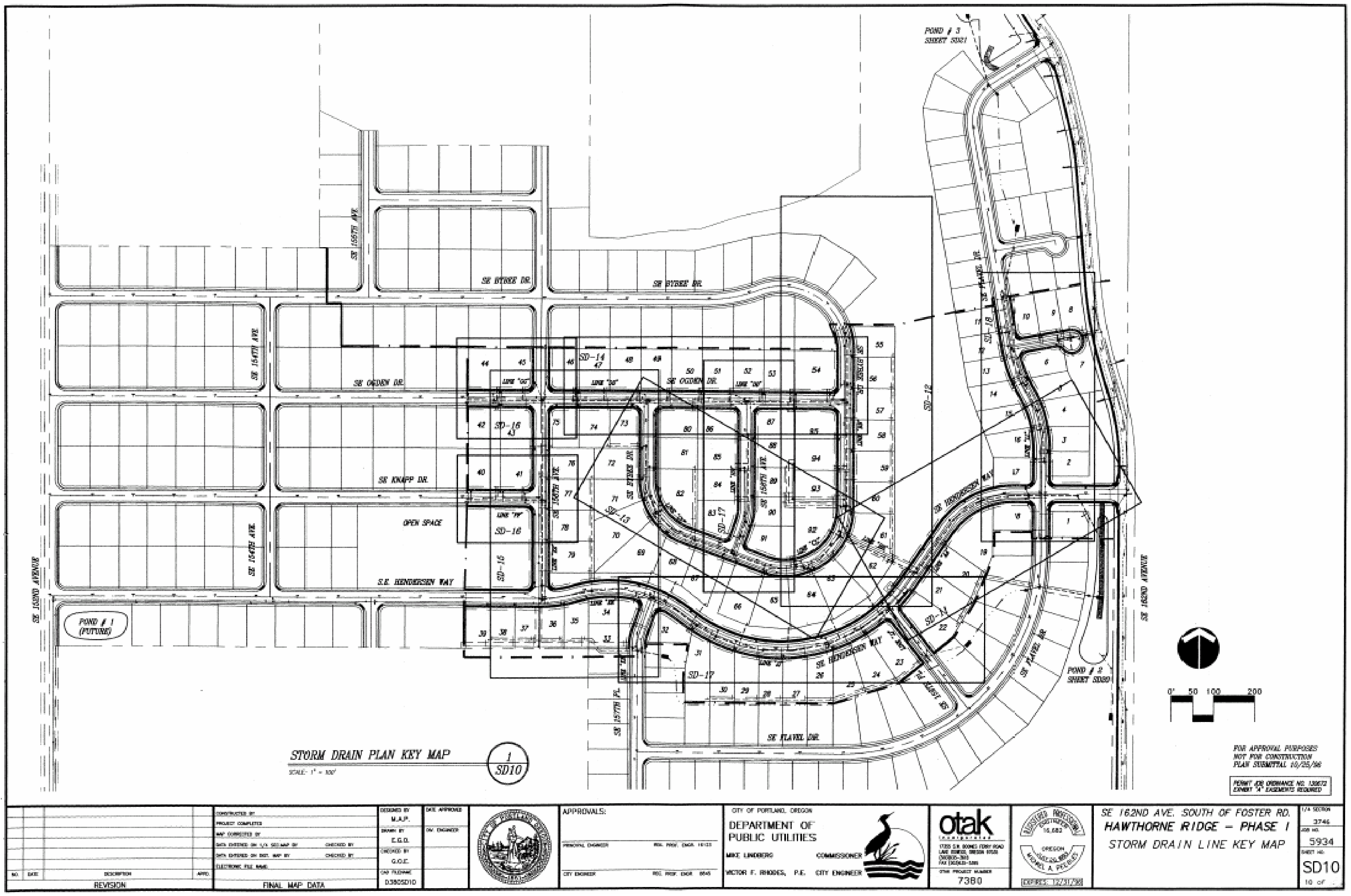

Figure 15-4. Lexington Hills BMP site in relation to catchment (Liptan 2001). Pond 3 is at upper right corner of figure, to the left of the upper-right intersection.

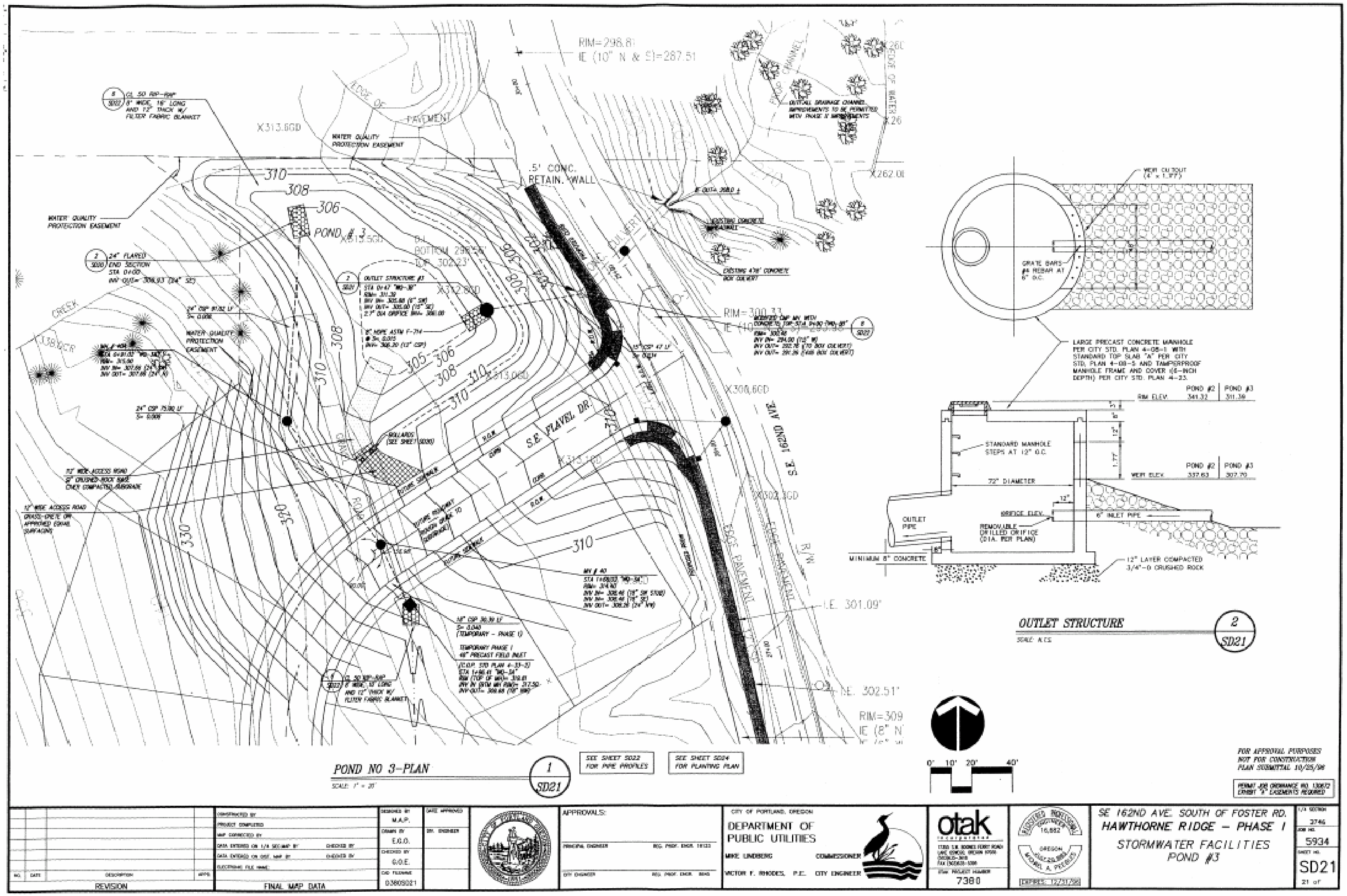

Figure 15-5. Details of Lexington Hills extended detention Pond 3 (Liptan 2001).

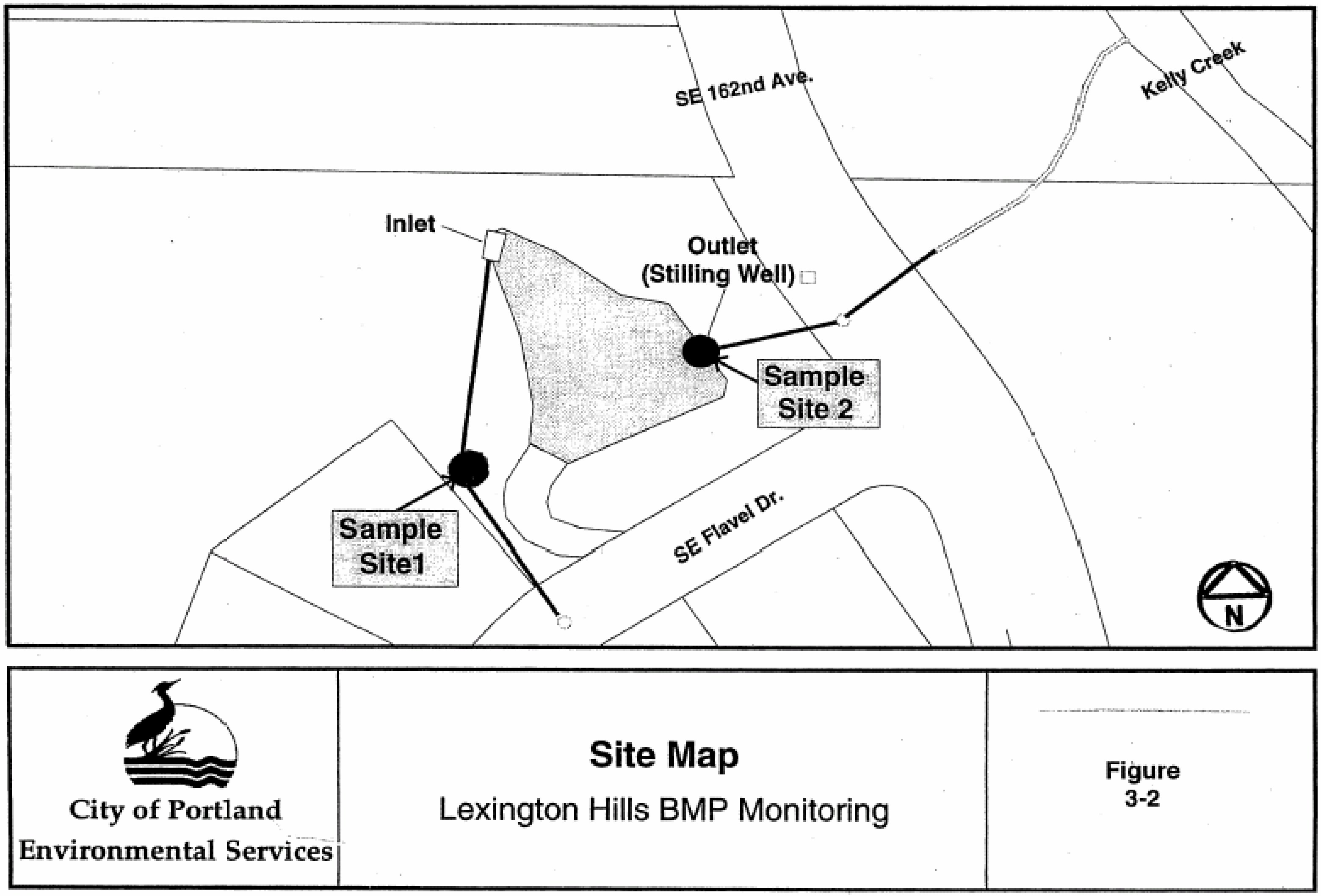

Figure 15-6. Location of influent (Site 1) and effluent (Site 2) monitoring (Liptan 2001).

Flow-weighted composites were prepared from sample bottles in order to determine event mean concentrations (EMCs) at the influent and effluent of the pond. Flow measurements were conducted at the pond entrance and in the stilling well of the pond exit (thus measuring the sum of the orifice and weir flows). Details of the sampling methods are provided by Liptan (2001).

15.3 SWMM MODELING

15.3.1 Background Information

Fortunately for this project, the BES had performed prior SWMM modeling of the overall Johnson Creek watershed. A map of Johnson Creek subcatchments is shown in Figure 15-7, with a more localized view shown in Figure 15-8. BES SWMM data were used as the starting point for simulations described herein. PCSWMM (http://www.computationalhydraulics.com/) was used as the graphical user interface (GUI) to run SWMM version 4.4h.

The City of Portland has an extensive, cooperative network of tipping bucket rain gages (with the U.S. Geological Survey, USGS) that provide rainfall at 5-min. intervals (http://oregon.usgs.gov/non-usgs/bes/). The SWMM Rain Block provides the ability to process such long-term data for input to the model. Rain values were supplied from the Holgate rain gage (“gage 21”), located approximately 2 miles to the northwest of the site (a general map, from the web site, is shown in Figure 15-9). Continuous 5-min. rainfall data for the period from January 1, 2000 to December 31, 2000 were imported into the Rain Block. Rain Block output was then linked to the Runoff Block. At first, actual rainfall values used for the measured data were averaged from the Holgate and Pleasant Valley rain gages (Figure 15-9). However, after reviewing the rainfall data, it was determined that no significant changes in simulated rainfall occurred that warranted the averaging of the two rain gages for SWMM modeling; hence, just the Holgate data were used for all runs.

Next, a Runoff Block input file was generated using the subcatchment data provided by the BES. Interestingly, the BES used an imperviousness percentage of 14, compared to the ratio 8.75/26.57 = 33% given in the BES report (Liptan 2001). The 14% value was used in the simulations because it gave much more reasonable results than the 33% value (see discussion of volumes below). Another slight discrepancy is in catchment area: 26.98 ac in the BES SWMM data vs. 26.57 ac in the data report. The former was used in these simulations.

The wet time step simulated was 15 minutes, while the dry time step was set to 24 hrs. Dates of simulation were set to correspond to the storm events in which sampling took place by the BES. The SWMM 4.4h Runoff Block input file is provided in Table 15-2.

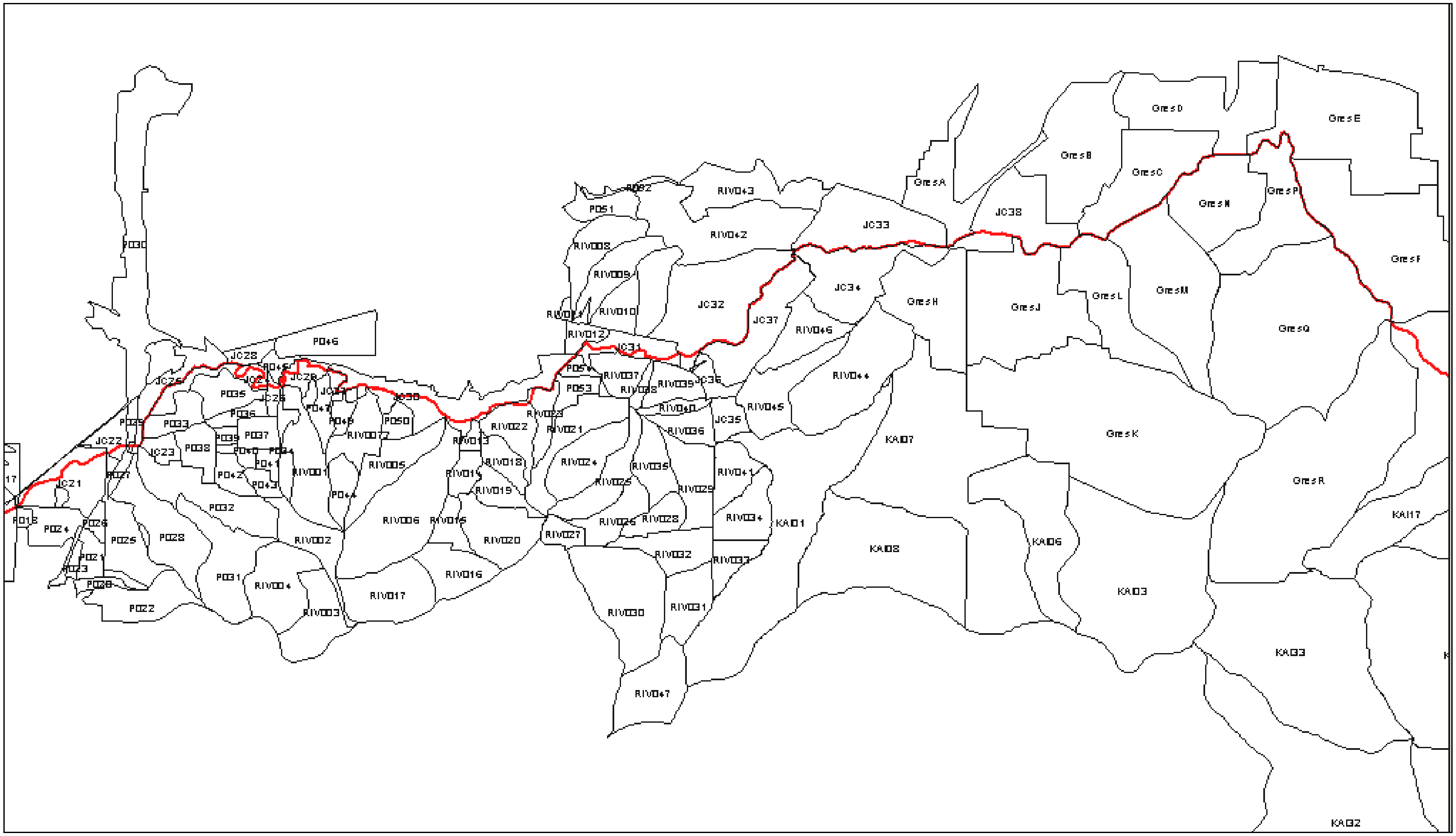

Figure 15-7. Regional subcatchments used in BES modeling of Johnson Creek Watershed. Lexington Hills is at subcatchment JC35 in center of the map. Johnson Creek runs in red, from right (east) to left (west).

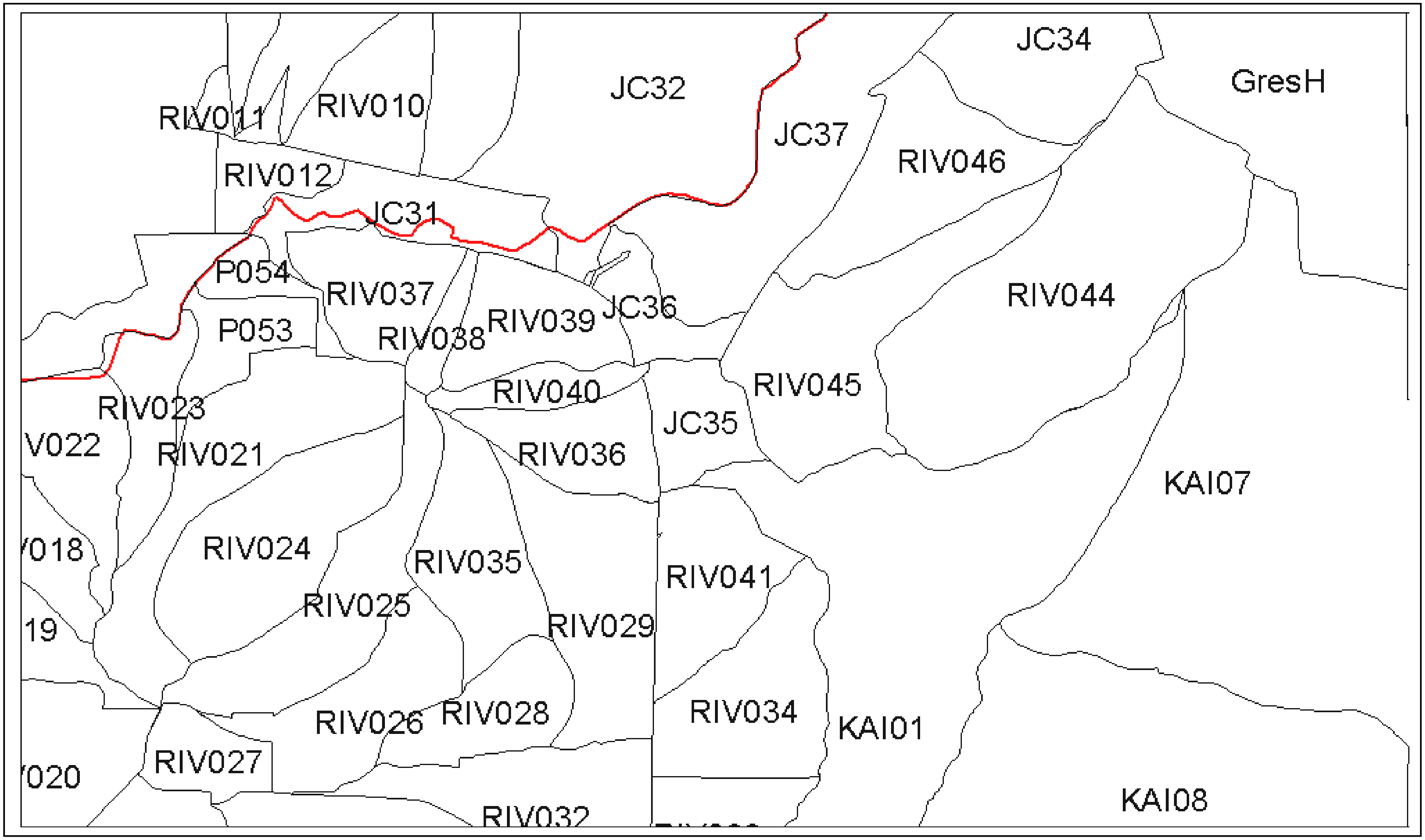

Figure 15-8. Localized view of Johnson Creek Watershed subcatchments. Johnson Creek is shown in red and flows west (right to left) in upper portion of map. The Lexington Hills subcatchment is number JC35 in center of map.

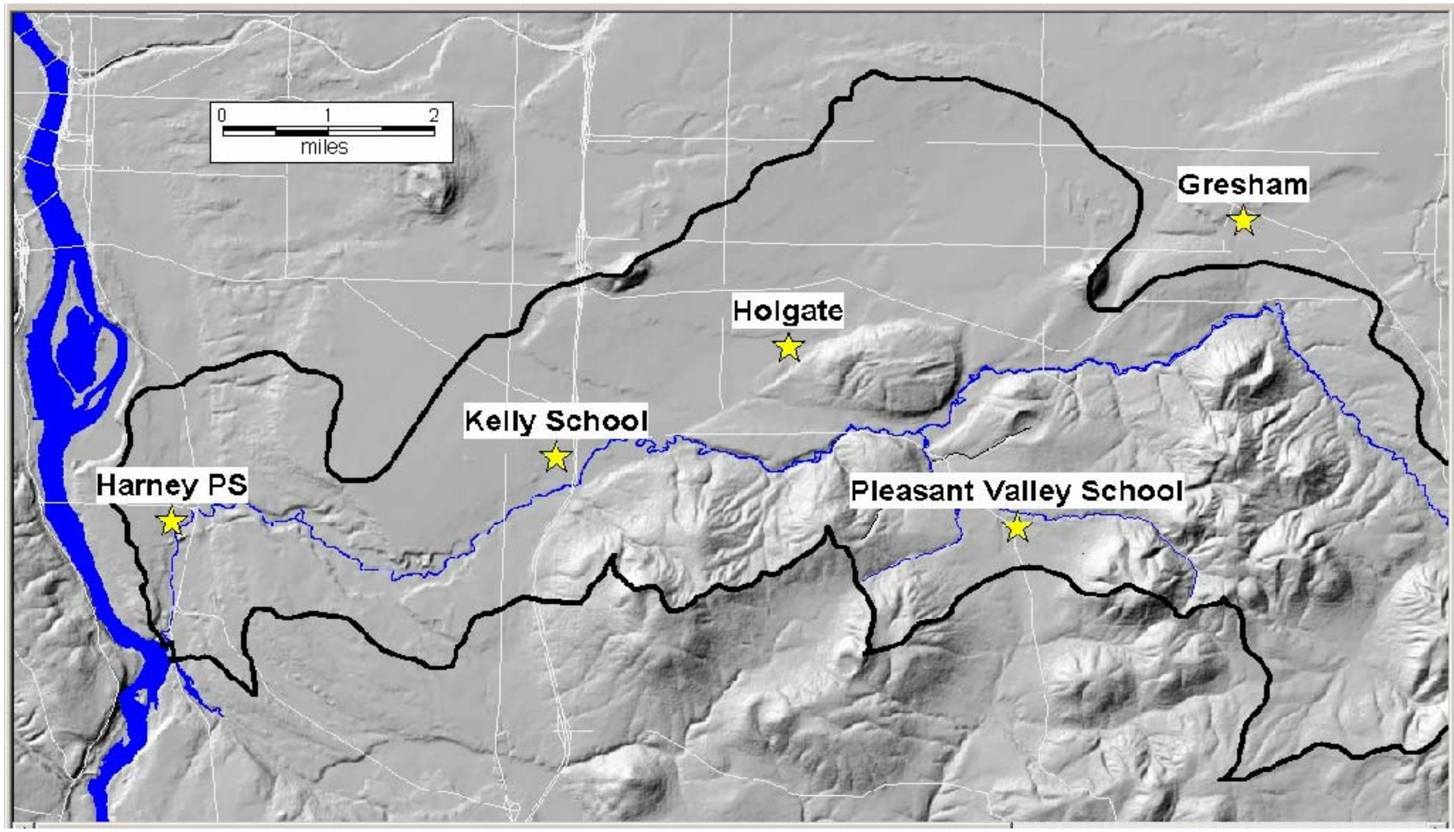

Figure 15-9. General map of BES raingage network for Johnson Creek near Lexington Hills (Lexington Hills is between the Holgate and Pleasant Valley School gages). The large river on the left (west) is the Willamette, draining to the north.

Table 15-2. Runoff Block input for Lexington Hills Pond 3 simulation (Stouder 2003).

```
*  File control and file linkages not needed with PCSWMM
*SW  1       0       9
*MM  9       11      12      13      14      15      16      17      18      19
$ANUM
$RUNOFF
*    Title Lines
A1   'Lexington Hills'
A2   'BMP Pond 3, April 13, 2000'
*=========================================================================
*    Run Control
*=========================================================================
*  METRIC  ISNOW   NRGAG   INFILM  KWALTY  IVAP    NHR     NMN     NDAY    MONTH   IYRSTR  IVCHAN
B1   0       0       1       1       1       0       0       0       13      4       2000    0
*   IPRN1   IPRN2   IPRN3   IRPNGW
B2   0       1       0
*    WET     WETDRY  DRY     LUNIT   LONG
B3   900     900     86400   3       365
*    PCTZER  REGEN
B4   0       0
*==========================================================================
*    ROPT
D1   1
*Rainfall for November 26  2000 through November 29  2000
*Entered from PCSWMM Meteorological module.
*===============================================================================
*    Evaporation Data
*===============================================================================
*    VAP(1)  VAP(2)  VAP(3)  VAP(4)  VAP(5)  VAP(6)  VAP(7)  VAP(8)  VAP(9)  VAP(10) VAP(11) VAP(12)
* Use default of 0.1 in./day
*===============================================================================
*    NAMEG   NGTO    NP      GWIDTH  GLEN    G3      GS1     GS2     G6      DFULL   GDEPTH
*===============================================================================
*    Conduits/Channels
*===============================================================================
* None used in this simulation.
*===============================================================================
*    Subcatchments
*===============================================================================
```

Table 15-2 (Continued. Please note wrap-around on H1 – H3 and J2 lines.)

```
*    JK   NAMEW        NGTO    WIDTH  WAREA  IMPERV  WSLOPE  IMPER_N PERV_N WSTORE1 WSTORE2 SUCT HYDCON
IMD
H1   1   'KC00372#2' 'KC00372' 160    26.98  14      0.06    0.15    0.25    0.03    0.1    9.0  0.4
0.00005
*    NMSUB   NGWGW   ISFPF   ISFGF   BELEV  GRELEV  STG     BC      TW
H2   'KC00372#2' 'KC00372' 0       0       0.0     8.5     8       7       9
*    A1      B1      A2      B2      A3      POR     WP      FC      HKSAT   TH1
H3   0.0014  1.      0.      1.0     0.0     .35     0.1     0.15    0.15    0.15
*    HCO     PCO     CET     DP      DET
H4   0.1     5.      .5      0       1.0
*=================================================================================
*    Water Quality
*=================================================================================
*    IMUL
JJ   0
*    NQS     JLAND   IROS    IROSAD  DRYDAY  CBVOL   DRYBSN  RAINIT  REFFDD  KLNBGN  KLNEND
J1   4       1       0       0       5.00    3.00    5.00    0       0       0       0
*    LNAME   METHOD  JACGUT  DDLIM   DDPOW   DDFACT  CLFREQ  AVSWP   DSLCL
J2   'Residential' 0       1       1.0E04  1.0     1.0     7.0     1.0     0
*    PNAME   PUNIT   NDIM    KALC    KWASH   KACGUT  LINKUP  QFACT1  QFACT2  QFACT3  QFACT4  QFACT5  WASHPO
RCOEF   CBFACT  CONCRN  REFF
J3   'TSS'   'mg/L'  0       1       0       1       0       0       0       0       0       0       2.0
0       0       125     0
J3   'Nitrate' 'mg/L'  0       1       0       1       0       0       0       0       0       0       2.0
0       0       0.23    0
J3   'BOD'   'mg/L'  0       1       0       1       0       0       0       0       0       0       2.0
0       0       15      0
*J3  'TotPh' 'ug/L'  1       1       0       1       0       5       1       0.1     0       0       0.8
5       1.0     0       0
J3   TotZn   'ug/L'  1       1       0       1       0       0       0       0       0       0       2.0
0       0       65.8    0
*    KTO     KFROM   F1
J4   2       1       0.02
*    TSS   Nitrate   BOD       TZn
J5   125     0.23    15      65.8
*    NAMEW   KL      BASINS  GQLEN    NDIM  P(TSS)  P(Nit)   P(Cu)  P(Pb)  P(Zn)
L1   KC00372#2 1       0       0       0       0       0       0       0       0
*====================================================================================
*    Print Control
*====================================================================================
```

Table 15-2 (concluded)

```
*    NPRNT   INTERV
M1   1       8
*    NDET    STARTP1 STOPPR1
M2   1       20000412 20000413
*    IPRNT
M3   KC00372
*    MDEEP   KDEEP
*M4  0       0
*====================================================================================
*ENDPROGRAM
```

The Transport Block was used to simulate the pond since it has the ability to simulate both storage and first-order decay within the storage. Inflow and loads from Runoff occurred at an upstream “manhole” in Transport. The pond was then simulated as a storage unit, draining to another “manhole” for tracking. Bathymetry measurements were obtained from the construction drawings provided by the BES, which showed detailed contour lines (Figure 15-5). These were used to develop stage-area-volume-outflow data that Transport uses for storage-indication flow routing. However, additional effort was needed to develop the outflow rating curve, as described below.

Effluent from the pond exits through a 2.5-in. orifice, which is designed to drain to the orifice level in about 24 hrs. When there is more water than can exit through the orifice, water spills over a broad-crested weir located above the orifice (Figure 15-5). A rating curve was developed by summing the theoretical orifice and weir equations,

$$Q = C_d A_0 [2g(h-h_0)]^{1/2} + C_w L_w (h-h_w)^{1.5} \qquad (15\text{-}1)$$

where

Q = outflow (ft^3/s),
C_d = orifice discharge coefficient, assumed ≈ 0.9,
A_0 = area of orifice = 0.0341 ft^2 for a diameter of 2.5 in.,
g = gravitational acceleration = 32.2 ft/s^2,
h = elevation of water in pond (ft), above pond bottom at elevation ≈ 305.0 MSL,
h_0 = elevation of orifice centerline = 306.0 ft,
C_w = broad crested weir coefficient ($ft^{0.5}/s$), assumed ≈ 3.3,
L_w = length of weir = 4.0 ft,
h_w = weir crest elevation = 307.7 ft.

Surface areas were planimetered and interpolated from the Pond 3 geometric plans, Figure 15-5, yielding the information in Table 15-3 for input into the Transport Block storage element. It should be noted that the outflow rating curve in the stilling well used by BES for monitoring is useful for hydrograph verification but not for modeling, since the stilling well is downstream of the orifice and weir used in the model (Figure 15-5).

Table 15-3. Rating curve development for Pond 3 at Lexington Hills.

Depth, h, ft	Area, ft^2	Volume, ft^3	Q orifice, cfs	Q weir, cfs	Q total, cfs	
0.00	2124	0	0	0	0	
0.50	3152	1319	0	0	0	
1.00	4180	3152	0	0	0	Orifice sill
1.25	4350	4218	0.12	0	0.12	
1.50	4844	5367	0.17	0	0.17	
2.00	5510	7956	0.25	0	0.25	
2.70	6350	12107	0.32	0	0.32	Weir sill
3.00	6850	14087	0.35	2.17	2.52	
3.25	7200	15843	0.37	5.38	5.75	

15.3.2 Initial Modeling

Inflows from the five storm events modeled were compared with actual flow volumes recorded during the sampling events. This is illustrated by comparing recorded and simulated inflows to rainfall, as shown in Figure 15-10.

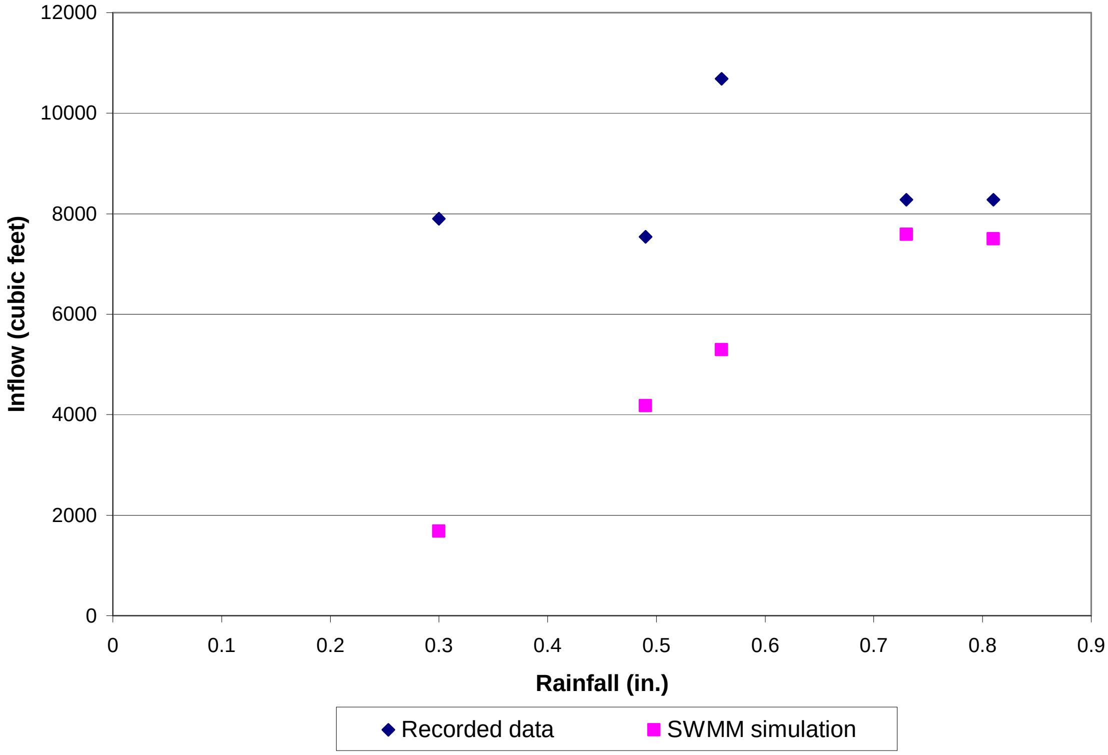

Figure 15-10. Comparison of recorded and simulated flows vs. rainfall for the five simulated events.

It is clear from Figure 15-10 that recorded inflows to the site vary only slightly with rainfall. This leads to questions about the confidence and reliability of the measured data. SWMM simulations indicate a steady increase of inflow with increased rainfall, which is what is expected. Because rainfall amounts were exactly the same for the recorded and SWMM data, it was decided that only the closest two events would be used for further modeling. These were event 1 (April 13, 2000) and event 4 (October 9-10, 2000).

15.3.3 Quantity Modeling Results

Quantity modeling results are summarized in Table 15-4, in which it may be seen that pond influent (catchment runoff) and pond effluent measured and simulated volumes agree well. For the two simulations of the pond, estimates of initial volume were made on the basis of the delay in the starting time of the outflow hydrographs.

Table 15-4. SWMM simulated inflows and outflows versus measured data for Pond 3 at the Lexington Hills BMP site.

Storm Date	Initial Vol. Estimate, ft^3	Measured Inflow, ft^3	Simulated Inflow, ft^3	Measured Outflow, ft^3	Simulated Outflow, ft^3
April 13, 2000	2692	8275	8058	8075	7749
October 9-10, 2000	2629	8276	8208	6815	7339

SWMM predicts slightly higher outflow for the October 2000 storm than was measured. However, the actual measured outflow value was assumed to be low by the BES sampling crew due to clogging of the orifice from debris such as leaves, needles, etc. (Liptan 2001).

15.3.4 Quality Simulations

Modeled constituents were TSS, BOD, NO_3-N, and total zinc, from a much longer list of samples for which EMC values were available from BES monitoring (Liptan 2001). No attempt was made to calibrate Runoff Block nonpoint source water quality values. Instead, Runoff Block concentrations were set equal to measured BES influent concentrations (pond influent = subcatchment effluent) by setting Runoff Block rainfall and groundwater concentrations to the measured influent EMC values. This was done to avoid a time-consuming calibration process when the point of the study was to simulate the BMP, not the watershed. Results are summarized in Table 15-5.

Table 15-5. Measured and simulated quality results for Lexington Hills Pond 3. Concentrations are EMCs. Loads (lb) are computed as product of EMC and measured or simulated volumes (Table 15-4). Pond influent is same as catchment effluent.

	TSS, mg/L	**TSS, lb**	**BOD, mg/L**	**BOD, lb**	**NO_3-N, mg/L**	**NO_3-N, lb**	**Tot. Zn, ug/L**	**Tot. Zn, lb**
Event 1, April 13, 2000								
Influent, measured	125	64.5	15	7.7	0.23	0.119	65.8	0.034
Influent, simulated	125	64.4	15	7.5	0.23	0.118	65.8	0.034
Effluent, measured	35	17.6	8	4.0	0.28	0.141	28.9	0.0146
Effluent, simulated	98.5*	47.6*	15	7.3	0.23	0.111	55.3*	0.027*
Event 4, October 9-10, 2000								
Influent, measured	222	115	5	2.6	0.25	0.128	52.4	0.0271
Influent, simulated	222	114	5	2.6	0.25	0.129	52.4	0.0268
Effluent, measured	114	48.5	6	2.6	0.36	0.153	38.2	0.0162
Effluent, simulated	138*	63.5*	5	2.3	0.25	0.115	35.0*	0.0160*

*See text regarding "decay" used for simulated TSS and total zinc in pond effluent.

Pond effluent EMCs for TSS and total zinc are less than influent EMCs for both storms. The pond would not be expected to remove much, if any, nitrate; in fact, measured effluent nitrate values are higher than influent values. The BES (Liptan 2001) offers no explanation for higher effluent nitrate values, but some could be hypothesized, such as nitrification in standing water during interevent times. The quality of any water standing below the orifice at the start of an event was not measured, so there is no way to determine if the initial pool added mass to the effluent. Influent BOD values are higher than for the effluent for event 1 and about the same for event 2. Although some BOD decay might be expected, the relatively short detention time would not yield much change. Thus, for the simulations, both BOD and NO_3-N were treated as conservative constituents, and it may be seen that inflow concentration is the same as outflow concentration in Table 15-4.

The reduction in TSS and total zinc is expected by sedimentation, of the solids themselves and of the adsorbed zinc. The only decay in the Transport Block is by a first-order process (not including the relatively untested July 2001 modifications to Transport that include a settling velocity). Hence, an equivalent first-order decay coefficient was created from a settling velocity by (see also Equation 6-14),

$$k = v_s/h \tag{15-2}$$

where
k = decay rate (s^{-1}),
v_s = settling velocity (ft/s), and
h = average depth (ft).

The average depth of outflow was determined to be about 1.2 ft from review of the Transport output files. A particle size distribution is provided by BES for the October 9-10, 2000 and later events (not for the April 13, 2000 event). These data are presented in Table 15-6, from which a weighted average diameter can be computed of 25.2 μm, using range midpoints as representative diameters. From this a typical diameter of 25 μm was used in Stokes' law (Equation 6-6).

Table 15-6. Particle size distribution for event of October 9-10, 2000. Counts are in units of particles per 100 mL.

Size range, μm	Range mid-point, μm	Influent Number	Effluent Number
Total particles		75100	29000
5-15	10	32400	26800
15-25	20	4500	1600
25-50	37.5	36900	500
50-100	75	1300	100
>100		n/d*	n/d*

*n/d = not detected

For a measured temperature of 15°C, kinematic viscosity, $\upsilon = 0.0115$ cm^2/s (Chapra 1997). Using g = 981 cm^2/s and a diameter of 25 μm = 0.0025 cm, Equation 6-8 gives a settling velocity of 0.0444 cm/s = 0.00146 ft/s = 126 ft/day. Dividing by an average depth of 1.2 ft gives a first-order decay coefficient of 105 day^{-1}. It may easily be recognized even without any simulation that any constituent with such a large coefficient will be completely removed in just an hour or so, e.g., the half-life,

$$t_{1/2} = \ln 2 / k = 0.0066 \text{ day} = 0.16 \text{ hr} \quad (15\text{-}3)$$

In fact, this value of 105 day^{-1} is in same order of magnitude as values derived in Table 7-4. But not all of the Pond 3 TSS and total zinc is removed.

Another approach would be to use the SWMM S/T Block with a particle size distribution, but the particle counts would have to be converted to concentrations. Logistical constraints prevented this approach (this work was done too late in the project period). In the interest of expediency, overall removal percentages (Liptan 2001) were used to compute approximate first-order decay coefficients for Pond 3. For the seven monitored events, average TSS removal was 57%, average total zinc removal was 45%, average BOD removal was 6%, and average nitrate removal was -11%. BOD and nitrate were treated as conservative, especially since the model has no physically realistic mechanism for increasing the concentration of a pollutant, other than starting with a higher concentration in the permanent pool (discussed earlier). Assuming TSS and total zinc reduction occurs in t = 1 day, an equivalent first-order decay coefficient can be computed from

$$C/C_o = 1 - \text{removal fraction} = e^{-kt} \quad (15\text{-}4)$$

yielding a TSS decay coefficient of 0.84 day^{-1} and a total zinc decay coefficient of 0.60 day^{-1}.

The effluent TSS values predicted by SWMM are above the measured effluent EMCs for both events, although closer for the October event. Simulated total zinc EMCs are higher for the April event and slightly lower for the October event. Refinements could be made, but it would simply amount to a curve-fitting exercise. Instead, the kind of results made possible by use of the somewhat maligned removal fraction (Section 4.3) can be imagined. Constituents treated as conservative reflect this fact through equality of influent and effluent EMCs in Table 15-5.

15.4 SUMMARY

It is difficult to obtain good data sets for testing of BMP simulation, including monitoring of influent and effluent flows and concentrations. The Lexington Hills Pond 3 monitoring conducted by the BES comes close, inasmuch as influent and effluent flows were monitored along with composite quality sampling. However, the flow monitoring itself is somewhat suspicious on the basis of the lack of a relationship of inflow to rainfall (Figure 15-10). A water quality sample from any standing water in the pond would have been useful, as well as noting the depth of the pond at the start of a storm event. Liptan (2001) notes a few other monitoring problems, such as issues of orifice clogging and rating curve generation. Nonetheless, the data set has served a valuable function in evaluating SWMM pond simulation capabilities.

Some conclusions follow:

- Using SWMM Runoff Block catchment data supplied by the BES, seemingly reasonable simulations of runoff hydrographs were obtained. However, the influent monitoring appears to give similar total volumes for all events, which is unexplained. Quality simulations were conducted for the two events for which simulated and measured runoff volumes most closely agreed (Figure 15-10).
- SWMM simulation of the pond hydraulics worked well even while using an outflow rating curve based on theoretical equations for the weir and orifice outlets. This is based on the comparison of simulated and measured total outflow volumes (Table 15-4).
- Approximation of TSS and total zinc losses in a SWMM Transport Block storage unit through an equivalent first-order decay coefficient is crude and serves only to demonstrate that the measured outflow EMCs could be replicated by the model. Better would be to attempt to simulate sedimentation processes in the S/T Block using the limited particle size distribution information provided, but time did not permit this approach.

16 RECOMMENDATIONS FOR SWMM BMP MODELING IMPROVEMENTS

16.1 INTRODUCTION

Evaluating the methods for model BMPs by process and type have much overlap. A breakdown of applicable methods for modeling each BMP is given in Table 16-1, followed by recommendations by type. While nearly all BMPs may be described using removal coefficients or with the effluent probability method (EPM), these methods do not model *processes*, i.e., fundamental unit processes of environmental engineering. Neither would they predict overloaded or undersized BMPs, or describe failing BMPs. This makes these methods less useful for sizing and design of BMPs. Nonetheless, if the EPM is the best representation of BMP performance for an otherwise hopelessly complicated set of processes, it would be a useful option to include in SWMM. This would be implemented by enhancing the statistical analysis capability, currently in the SWMM Statistics (Stat) Block, to provide comparative lognormal plots of influent and effluent EMCs. In addition, an option for specifying a lognormal (or other distribution) for influent and/or effluent EMCs is needed.

A somewhat general CFSTR formulation of fundamental source-sink processes that can be used to represent BMP impacts was provided in Section 4.4, with regard to Equation 4-9, which is repeated here:

$$V\frac{dC}{dt} + C\,\frac{dV}{dt} = (Q_i C^{in} + L\;)(1 - R\;) - QC\; - k\;C\;V - v_s(1 - F\;)C\;A_s \qquad (16\text{-}1)$$

where

C = constituent concentration,
V = volume of conveyance/storage object,
t = time,
Q_i = inflow to object,
C^{in} = inflow concentration,
L = other loadings, e.g., from sediment or precipitation,
R = removal fraction due to BMP,
Q = outflow from object,
k = first-order decay rate,
v_s = settling velocity,
F = fraction of constituent in soluble (non-settleable) form, and
A_s = surface area.

The source-sink terms of this equation can be readily adapted to a one-dimensional advective-dispersion equation, if desired. Of course, the equation is solved immediately after the flow routing equation for the corresponding conveyance/storage object. Flow routing accounts for all inflows and outflows, including "vertical" processes of precipitation, ET, and infiltration. For CFSTR routing, these are reflected in the dV/dt term. If Equation 16-1 were used for a particular settling velocity range, it would be well-suited for treatment-train simulation as well, since particles with higher settling velocities would settle sooner, upstream. The settling velocity formulation could easily be enhanced to something other than a constant v_s as well, as in the current SWMM S/T Block.

16.2 PONDS

SWMM currently does a good job of modeling storage devices based on hydraulic controls, including settling and first–order decay. When searching for improvements to SWMM for pond simulation, one remaining (minor) issue is biological treatment based on second-order reactions. The use of second-order rate reactions used in P8 is novel to SWMM, but much less useful without a default set of reaction constants. The effectiveness of macrophytes on pollutant removal might be simulated with the use of a removal factor similar to the particle scale removal factor, f, in P8 (Section 6.2.3).

Although sedimentation in the SWMM S/T Block can be simulated with reasonable sophistication at the moment, input is not phrased in common environmental engineering terms such as overflow rate. Use of terminology that is more consistent with the profession might be helpful to the model user.

Environmental engineering texts and references often refer to the "tanks in series" model, i.e., a series of N CFSTRs, Equation 6-11, as a useful transition between one extreme of plug flow to the other extreme of complete mixing in a storage device (or chemical "reactor"). This could readily be implemented as an option for treatment in a SWMM5 "object" that includes storage, such as a channel, pipe, or storage unit. However, it will still require user judgment as to whether mixing is "good," "very good," etc., thus retaining the empiricism that seems difficult to avoid (see discussion of Equation 6-11). On the other hand, if tracer data are available, quantitative inferences about the degree of mixing can be made – see discussion in Section 7.8.2 regarding the MUSIC pond algorithms. The MUSIC implementation of Equation 6-11 provides qualitative guidance as to the number of CFSTRs that would be a good start for any SWMM implementation.

SWMM is well equipped to deal with detention and extended detention storage from the standpoint of timing issues. The relationship of storage availability and draw-down to local meteorology and catchment conditions may be analyzed quantitatively via continuous simulation. The principal enhancements for SWMM would be more useful statistics, e.g., analyze statistics of the time series of volume (or depth), in addition to inflows and outflows from storage. This only requires the ability to create additional time series of model state variables.

The hydrology of most extended detention ponds can be modeled in the same way as for wet ponds. However, a missing hydrologic component for both extended detention and for ponds in SWMM is infiltration and evaporation (at better than monthly averages). The vertical water balance may only be simulated crudely, through the input of "negative rainfall."

Table 16-1. Clarification of method applicability to modeling BMPs.

Simulation Practices	Ponds	Bioretention Facilities	Infiltration Trenches	Grassed Swales	Dry Wells	Cisterns	Porous Pavement	Hydrodynamic Devices
Effluent probability method	X	X		X				X
Removal coefficient	X	X		X			X	X
Second-order decay	X	X		X				
Particle scale removal factor	X		X					
Tanks (CFSTRs) in series	X	X						
Fractional volume reduction*			X	X	X	X	X	
Substrate modeling**	X	X						
Ecological and nutrient cycling modeling	X	X						

*As implemented in SLAMM, e.g., Equation 8-1.

**As implemented in VAFSWM.

16.3 WETLANDS AND BIORETENTION FACILITIES

The particle scale removal factor in P8 (usually set to 1) is applied as a calibration parameter for devices. This can be increased to represent better treatment and removal in bio-filtration facilities. Increased removal by macrophytes is documented in the P8 manual wherein the removal factor is set to 2 or 3 based on improved removal rates. But modeling multiple processes can lead to conflicts and difficulties with sequence of treatment. Moreover, the particle scale removal factor is akin to heuristic removal efficiencies and is difficult to represent as a fundamental unit process.

A more conventional alternative within the unit process literature is the tanks-in-series model (Equation 6-11), discussed just above and in Section 6.5 with regard to ponds and Section 7.8 with regard to wetlands. Parameter N, the number of CFSTRs in series, may be estimated somewhat quantitatively from tracer studies (Kadlec and Knight 1996, Persson et al. 1999), or qualitatively on the basis of an estimate of short-circuiting and dead zones (Fair et al. 1968, Pitt and Voorhees 2000, Wong et al. 2001, Minton 2002). Kadlec and Knight's (1996) k'-C* modification of exponential decay used in MUSIC (Section 7.8) is a useful means by which to provide a lower bound (C*) from a CFSTR with first-order decay. The lower bound could also be in the form of a frequency distribution of effluent EMCs.

Certainly wetlands, bioswales, and similar storage areas where there is strong interaction between water quality and biological processes within the area (e.g., vegetation, sediment) are one of the most difficult challenges for BMP simulation. Three models that perform this kind of simulation (WETLAND, PREWET, DMSTA) include complex kinetic formulations and several state variables to represent nutrient cycling. Whether or not this effort is needed or can be supported for SWMM is to be determined. The VAFSWM simplifies the process by simulating only a "substrate" (in addition to the constituent in the water column) consisting of water in sediment and vegetation, and without the linked nutrient kinetics. This concept would be easier to implement in SWMM. A useful enhancement regardless of whether physical and biological process options are updated to better represent wetlands is the ability to link SWMM time series output to other models. For instance, wetland quality processes may be simulated for the most part with the EPA WASP model (Wool et al. 2001), with HSPF being another option (Bicknell et al. 1997). A standard for protocol interfacing is required to facilitate such linkages. That is, SWMM simply may not be the best choice for a "universal treatment model" when it comes to simulation of natural receiving water processes.

16.4 INFILTRATION TRENCHES

One of the biggest needs for SWMM with regard to infiltration trenches and swales (below) is the ability to simulate infiltration from *channels* as well as for overland flow planes. All conveyance and storage options within the model need to include a full vertical water balance that includes precipitation, ET, and infiltration. Within this framework, SWMM conveyance and storage elements should be able to simulate quality processes characteristic of infiltration devices, bio-swales, etc. Notwithstanding the need to be able to infiltrate from channels, infiltration trenches can be simulated with the current SWMM as small overland flow planes, with the ability to receive runoff from an upstream overland flow plane. Depression storage is used to define the depth of the trench, as described in Section 8.3.

Other models perform this function using highly empirical, though often useful, methods. For instance, the methods described for SLAMM produce a fractional volume reduction based on adjusted watershed runoff coefficients that reflect smaller storms, and adjusted percolation rates that account for soil moisture and compaction. Unfortunately, infiltration rates themselves must be known a priori; however, SLAMM's reference base is helpful in this regard, e.g., for disturbed soils.

16.5 GRASS SWALES

Regarding grass swales, filter strips, and similar BMPs that feature flow through vegetation, the same needs for a vertical water balance in conveyance and storage devices just discussed apply here. However, to the extent that flow through swales behaves like a sedimentation device for which performance is a function of hydraulic overflow rate (Minton 2002), swales can be simulated in the current SWMM S/T Block. But a ready means of simulating infiltration from S/T storage is still needed. The simplified infiltration procedures in SLAMM are applicable to grass swales (see infiltration trenches section), but these procedures do not seem to offer many SWMM enhancement opportunities. P8's assumption of adsorption of some dissolved constituents while traveling through swales is an implementation possibility for SWMM, easily simulated through the S/T removal equation.

REMM closely models the N, P, and C cycles in three surface buffer zones (linked to three subsurface layers), which may be applicable to SWMM, although this will require extensive field data to simulate. The most useful information from the REMM model when good documentation becomes available may be the default rate constants used in calculations.

The REMM techniques are certainly applicable in vegetated filter strips, but the REMM methods are complex in the same way as the nutrient cycling simulations are complex in the wetlands models. Moreover, REMM includes a linked surface-subsurface hydrologic model for the riparian zone. This quasi-2D model (three horizontal by three vertical zones) is beyond the capability of SWMM at the moment and likely not worth the effort of implementation given SWMM's common stormwater design applications. Nonetheless, similarly to the wetlands models, there is good opportunity for parameter estimation help from REMM publications, and such a model could be used to calibrate simpler techniques that might be added to SWMM.

What the literature review of this study has not yet uncovered – which is not to say that they are not available – are functional relationships between performance and design parameters for filter strips, although this kind of information does exist for swales (Fletcher et al. 2002). For instance, one expects removal to be proportional to flow path length, type of vegetation, type of soil, etc. (Huber et al. 2000). While SWMM can now simulate load removal associated with infiltration, concentrations are not changed through sedimentation or biological processes in the Runoff Block overland flow routines, although they may be changed using a constant settling velocity, first-order decay rate, or removal fraction. When empirical or theoretical relationships are found with causative parameters – e.g., length, vegetation, soil, slope, and maintenance – they should probably be implemented in SWMM.

16.6 DRY WELLS

Modeling dry wells can involve complex analytical techniques for both quantity and quality for simulation of the groundwater flow regime. Under suitable soil circumstances, though, it may be possible to infiltrate the entire design inflow over the approximate storm duration. SWMM is currently capable of accepting flows into a device that represents a dry well but not capable of simulating the complex flow net that results as water enters the soil. This latter effort is probably not necessary to simulate dry wells as BMPs as long as their capacity can be provided as a function of time or other simple relationship (e.g., a rating curve). If current SWMM infiltration routines are used (from subcatchments), a linkage to groundwater routines can be made to follow the pathway of water further, should that be desired.

16.7 CISTERNS

For cisterns what is needed in SWMM and similar models is an overall water use simulation capability that includes all sources (e.g., cisterns, city water supply) and an irrigation schedule (as in SLAMM) in order to dispose of runoff collected in cisterns. Water stored in cisterns can also be used for gray water supply in buildings and boiler feed water, both of which might require some on-site treatment. The irrigation schedule might in turn be linked to a soil moisture accounting model, leading to greater complexity. Otherwise, SWMM is capable now of simulating the storage associated with cisterns – just not the timed release!

16.8 POROUS PAVEMENT

SLAMM models porous pavement with infiltration rates and rainfall events manipulated to focus on the micro-storm effect. SWMM currently can simulate porous pavement by using its Horton or Green-Ampt infiltration equations, and the resulting infiltrated water can be tracked through use of the groundwater routines, except for water quality. James et al. (2001) demonstrate how the current version of SWMM can be used to simulate roadways and parking lots with porous pavement or permeable pavers.

16.9 HYDRODYNAMIC DEVICES

Because of the variability between devices in this category, the effects may best be simulated with removal equations based on the manufacturer's specifications, or with data from the EPA/ASCE BMP Database. On the other hand, removal as a function of overflow rate may work just as well. Checks need to be established to determine if devices are overloaded or for maximum treatment values. Extran is capable of simulating the hydraulics of some common devices through proper use of orifice settings in particular, but water quality is not tracked.

16.10 LID AND OTHER RELATED NEEDS

Almost all BMP design is linked to a description of the influent stormwater. A characterization of solids is especially important, i.e., *treatability* data on settling velocities and/or particle size and specific gravity distributions. The SWMM S/T Block requires this information to perform sedimentation computations, but this is entered only for the S/T Block and as a constant for the inflowing stormwater or combined sewage. Upstream blocks should supply this time series – a very complicated process since it relates to erosion, scour, deposition, and sediment transport, all of which are poorly represented in SWMM and in most any alternative model. Nevertheless, typical particle size distributions are available from several sources (e.g., Minton 2002) that can be provided as a default, in lieu of site-specific data. Within the rest of SWMM, particle sizes can be tracked by treating a size range as a separate constituent subject to a constant settling velocity. This is not too bad an assumption in the current Runoff and Transport Block routines for which settling in essence can be simulated as a function of residence time, V/Q. Even without improved scour and deposition routines, simply linking the constituents grouped by particle size (or settling velocity) range to the S/T Block would be a huge enhancement to the overall SWMM simulation capability. If SWMM5 can perform this linkage, then it will immediately provide an improved BMP simulation capability.

Several of the models discussed in this report include heuristic parameters (e.g., efficiency ratio, particle scale removal factor). While these parameters do not represent fundamental unit processes and are difficult to evaluate *a priori*, if they gain favor in the profession, SWMM might include them as options for simulation of BMPs. The N tanks-in-series model (Equation 6-11) would be relatively easy to implement as a treatment option within any device that incorporates storage, including any conveyance element.

Regarding the suitability of LID simulation, the ability to route flows from one overland flow plane to another is useful (e.g., see Section 4.7) but insufficient. Infiltration and ET simulation are also required for open channels and for ponds and storage devices. The vertical water balance is critical in infiltration devices such as swales as well as in wetlands. SWMM currently has only limited capabilities in these areas. Another useful LID simulation procedure would be to provide a way in which stored water could be distributed seasonally, as from a cistern. An irrigation schedule would be useful here, and coupling with other components of the urban water cycle (e.g., water supply, wastewater removal) should be provided.

SWMM does not simulate water quality in the subsurface zone, i.e., in the Runoff Block groundwater routines. Examples of models with this capability include HSPF and REMM. This would require a major renovation of model capabilities, but it would be useful for simulatinf all water budget components in an area subject to infiltration. However, alternative models, e.g., HSPF and REMM, already provide this capability.

Another useful addition to SWMM would be the ability to prescribe a lognormal (or other) frequency distribution to represent EMCs in surface runoff and/or effluent quality from BMPs, e.g., as characterized by the effluent probability method. The modeling challenge would be in part to ensure conservation of constituent mass if EMCs and hence loads are inserted into the effluent (or nonpoint source runoff) from a prescribed frequency distribution.

Finally, SWMM is unlikely ever to perform all the functions that a user might need. Particularly for complex nutrient dynamics and subsurface water quality, alternative models such as WASP, HSPF, REMM or WETLAND should be considered. In order to interface one model with another, standards for time series transfer are needed. Such standards would facilitate many multi-media modeling needs within the environmental community in general and Environmental Protection Agency in particular.

16.11 FINAL SUMMARY OF SWMM BMP SIMULATION NEEDS

For the convenience of the reader, proposed BMP/LID simulation enhancements are summarized in Table 16-2. A summary commentary is provided in Chapter 17. The priority assigned in the table is purely the opinion of the authors of this report. In their opinion, probably the highest priority enhancement should be item 10 of the table: inclusion of the vertical water balance (precipitation, ET, infiltration) for all flow objects, especially channels. This is urgently needed for proper simulation of any porous conveyance or storage, such as bioswales and wetlands.

Table 16-2. Summary of proposed SWMM BMP/LID simulation enhancements. These are listed generally in order of discussion in Chapter 16. Priority: H = high, M = medium, L = low.

	Simulation proposal	**Comment**	**Priority**
1	Emphasize fundamental unit processes.	Most flexible regarding “real world” settings.	M
2	Provide for automated evaluation of inflow and outflow EMCs by effluent probability method.	Will facilitate comparisons with published evaluations using this method, often recommended for BMP performance evaluation.	M
3	Provide for lognormal and other frequency distributions of watershed runoff EMCs and BMP effluent EMCs.	BMP influent and effluent EMCs are often characterized by a lognormal distribution and performance may sometimes best be characterized simply by the effluent distribution, rather than, for example, a removal fraction.	M
4	Provide a generalized source-sink formulation of the type given in Equation 16-1 for conveyance and storage objects.	This formulation provides for first-order decay, settling, and generalized “removal.”	H
5	Second-order decay is useful for some constituents.	A significant problem with implementation is lack of data.	L
6	Particle removal scale factors as used in P8 can help with calibration.	These scale factors amount to a linear increase in settling and first-order decay terms. This effect can also be provided for in other ways.	L
7	The “tanks in series” model provides a useful bridge between simulation of conveyance and storage objects as well-mixed (worst performance) and plug flow (best performance).	Adapting this formulation, often used in environmental engineering practice and implemented in the MUSIC model, might be one of the most useful enhancements to SWMM.	H
8	A lower bound on exponential decay through a first-order process in a CFSTR can be provided with the k’-C* model.	The lower bound, C*, could be specified in a numerical solution. The lower bound could also be in the form of a frequency distribution.	M
9	Upgrade SWMM statistical analysis options.	Related to item 1, graphical, regression, and other statistical evaluation options would make a model such as SWMM5 even more powerful as an analysis tool.	H
10	The “vertical” water budget must be available for all flow objects, including overland flow on subcatchments, conveyance channels, and storage objects. This implies inclusion of precipitation, ET, and infiltration for all flow objects.	Implementation of this vertical water budget is essential for simulation of conveyances such as swales, bioswales, and wetlands. Load reduction associated with infiltration, for instance, cannot be ignored.	H

Table 16-2. (Continued)

11	A ready method with which to interface SWMM to other models is urgently needed, including a standard format for time series files.	This will facilitate linkage of SWMM to models better suited to certain kinds of receiving water and subsurface water analysis, such as WASP and WETLAND. SWMM cannot "do everything," and the ability to link SWMM's watershed processes with the receiving water simulation strengths of other models will facilitate use of the right model for the right task.	H
12	Settling velocity ranges should be tracked through the runoff-transport-treatment pathways.	This will facilitate proper simulation of "treatment trains."	M
13	Related to item 12, even though stormwater treatability data are relatively uncommon, SWMM should be adapted to use such information when available.	The model must be able to track particle ranges based on settling velocities (or size – specific gravity ranges) throughout the whole model: sources, transport, and treatment.	H
14	Improved mechanisms for simulation of sediment, including scour and deposition, will facilitate item 13.	This is one of the most difficult set of physical processes to simulate in the context of urban hydrology.	M
15	Although dry wells can be simulated as an outflow or storage within SWMM, a link to groundwater routines would provide additional continuity.	Water leaving the surface through dry wells could be tracked similarly to infiltration from subcatchments.	L
16	Storage of water in cisterns can be simulated with the addition of a water distribution (water use) algorithm or input table.	Stored water can be used for irrigation, gray water, boiler feed water, etc. A schedule is needed for such distributions.	L
17	Data and parameters from other models should be adapted for use in SWMM to the extent possible.	Other models reviewed in this report have different ways of conceptualizing processes such as infiltration and porous pavement, but parameter values from such models may be very useful in SWMM simulations. The effect of disturbed soils on infiltration parameters is particularly important.	M
18	Additional information is needed on hydrodynamic devices to determine how well they can be simulated using fundamental processes. Simpler removal fractions might be used in the meantime.	Proprietary devices are especially difficult to evaluate simply on the basis of manufacturers' information.	M
19	Simulation of subsurface water quality would be useful for continuity of constituent loads as well as flow.	But this can be very difficult, with the need to account for sorption, etc. Linkage to groundwater quality models is another option.	L

17 CONCLUSIONS

The EPA Storm Water Management Model was developed in 1968-71 to simulate combined sewer systems, including evaluation of management strategies. The user community immediately, and logically, applied the model to separate stormwater systems as well as combined sewer systems and eventually into the entire spectrum of urban and non-urban watershed response. All of this was facilitated by EPA support for continuous enhancement and development of the model in the intervening years. The Storage/Treatment Block within the original model included algorithms that allowed it to mimic removal for a few specified pollutants and for an array of CSO controls available at that time; these algorithms were generalized and enhanced to permit broader simulation of series-parallel arrangements of S/T units beginning with SWMM III (Nix et al. 1978). In 2004, municipalities and their consultants are increasingly under pressure to implement BMPs for control of both stormwater quantity and quality. As new information and data are collected about the performance of BMPs, SWMM and similar models must be updated further to reflect this new technology. LID (hydrologic source control) offers yet another opportunity for stormwater management through distributed and localized control options, with emphasis upon infiltration, evapotranspiration, and water reuse. This report evaluates a number of alternatives for simulation of conventional BMP and LID options, with emphasis upon incorporation of fundamental unit processes into the algorithms wherever possible. Several ways in which SWMM can successfully accomplish these simulation goals are summarized below, along with several more areas in which improvements are needed.

When modeling BMPs, the ability of SWMM to redirect flow from impervious areas to pervious (simulating the effects of downspout disconnection) greatly enhances the model's ability to simulate LID. This has been shown in the literature (Lee 2003) and is demonstrated in an extensive case study for Portland (Chapter 14). However, for the model to be able to characterize treatment based on solids settling, improvements in the overall ability of SWMM to erode, transport, deposit, and scour sediment (i.e., to incorporate scarce treatability data) need to be provided, as well as the ability to simulate infiltration in channels and ponds. The vertical water balance must be computed for all conveyances and storages, not just for overland flow planes.

The Runoff Block is very stable with regard to size of subcatchments that are simulated. SWMM can model very small parcels (and consequently BMPs and LID facilities on an individual lot scale or smaller). An immediate consequence of simulation of areas on the order of a fraction of an acre is the need for a smaller Runoff Block (or SWMM5 subcatchment object) time step (e.g., ≤ 1 minute), compared to the more typical 5-minute value routinely employed. For a process model like SWMM, the issue of temporal and spatial variability is largely an issue of data availability and the amount of detail desired in the simulation run. For long-term (continuous) simulations, aggregated subcatchment schematizations can work just about as well as highly discretized schematizations, as demonstrated in the Portland case study presented in Chapter 14. However, this advantage becomes less relevant as computer speeds continue to increase.

Regarding BMP simulation, the S/T Block has the most capabilities for fundamental unit process simulation but is unsophisticated hydraulically and hydrologically. Insertion of fundamental processes into any conveyance or storage "object" in SWMM5 should make the model much more useful, especially if quality routing can be performed when dynamic routing (as in the current Extran) is being used for pipes and channels. The current Transport Block provides simpler but somewhat flexible mechanisms for simulation of storage BMPs, as demonstrated for a Portland detention pond in Chapter 15. The Transport Block also permits simulation of decay, settling, and removal within any conveyance or storage element, as described in Section 4.4. The general form of source-sink terms used in this formulation (Equation 4-9) is suitable for SWMM5 conveyance and storage objects.

When considering the options for modeling BMP performance that can be integrated into SWMM, the most applicable improvements would be infiltration from swales and trenches, storage and reuse in cisterns, storage and infiltration in dry wells, and inclusion of infiltration from porous pavements (although current SWMM infiltration and groundwater routines can be made to simulate porous pavement satisfactorily – Section 12.3). The SLAMM documentation has an extensive review of case studies and many default parameters, which may be useful when modeling these BMPs. P8's use of the particle scale removal factor may be a simple method for calibrating BMPs to actual performance data. The tanks-in-series model from environmental engineering practice appears to supply a similar qualitative parameter in N, the number of CFSTRs. REMM has the potential for useful algorithms and default parameters when modeling nutrient cycling in buffers. However, models such as REMM, WETLAND and DMSTA likely contain more complex nutrient cycling descriptions than need to be supported in SWMM and for which parameter estimates would be even more difficult than they are now. In this case, linkage to "downstream" receiving water models or groundwater models should be considered. Standards for transfer of time series from one model to another will facilitate such linkages and are urgently needed.

Modeling multiple BMPs in a "treatment train" (several BMPs in series) is difficult unless solids removal is represented through fundamental sedimentation processes – and separate solids "ranges" (i.e., characterized by settling velocity or else particle size and specific gravity) are carried from one part of the model to another, e.g., from the current Runoff to the Transport to the S/T Block. Difficulties with using removal rates in series can easily lead to over prediction of removal (if the first BMP removes 90% TSS, it is unlikely that the second BMP in series will remove 90% of the remaining suspended solids). Often, instead, the output may be relatively constant compared to influent concentrations. Some BMPs may increase effluent concentration (such as BOD or nitrate in biological systems) if influent concentrations are low. None of these were considered in the models surveyed in this report. Nor would the current SWMM be able to simulate this effect in other than "work-arounds." One option to handle this effect would be to input a frequency distribution of effluent EMCs.

Another issue is the malfunctioning or maintenance issues associated with BMPs. Reduced removal efficiencies are likely without regular maintenance, and effects of clogging or other reduction in proper performance should be included in any modeling procedure.

The United States Environmental Protection Agency can be proud of the current state of stormwater modeling using SWMM. Of the models surveyed in this study, SWMM has the most extensive and versatile capabilities for simulation of BMPs. Implementation of SWMM5, the "next generation" version of SWMM, should enhance the model's overall status for use by practitioners in stormwater and wet-weather flow management.

APPENDIX: USING REMM TO PREDICT RIPARIAN BUFFER PERFORMANCE

RIPARIAN ECOSYSTEM MANAGEMENT MODEL (REMM)

This appendix is intended to supplement the information on the Riparian Ecosystem Management Model (REMM) provided in Chapter 9. Source material is taken primarily from USDA-ARS (1999) and Inamdar et al. (1998a,b).

REMM has been developed for natural resource agencies and researchers as a tool that can help quantify the water quality benefits of riparian buffers. REMM simulates: (a) the movement of surface and subsurface water; (b) sediment transport and deposition; (c) transport, sequestration, and cycling of nutrients; and (d) vegetative growth.

When looking at the processes REMM is capable of modeling and applying those processes to all applicable BMPs, the strengths of the REMM model are the ability to deal with movement of subsurface water and fate and transport of nutrients, neither of which are currently available in SWMM. The applicable wet-weather control (WWC) and program suitability are listed in Table A-1.

REMM can be applied to:

- Quantify nitrogen and phosphorus trapping in riparian buffer zones and determine buffer width for a given set of riparian conditions and upland loadings.
- Determine buffer effectiveness under increased loads.
- Evaluate influence of vegetation type on buffer effectiveness.
- Determine impacts of harvesting on buffer effectiveness.
- Investigate long term fate of nutrients in riparian zones, sequestration in vegetation, or loss to atmosphere (denitrification in case of N) investigate N / P saturation in riparian buffers.

Table A-1. REMM wet-weather controls program suitability.

WWC Option	REMM
Source Controls	Modeled as reduced input from adjacent lands.
Overland Flow, Swales, Infiltration, Porous Pavement	Useful for grass swales. Infiltration simulated with a modified Green-Ampt equation, vertical unsaturated conductivity with Campbell's equation. Simulates surface and subsurface water, sediment transport and deposition, transport, sequestration and cycling of nutrients. Capable of modeling hydrology budget with losses to seepage, transpiration and interception. Porous pavement is not modeled.
Major Benefits	Detailed analysis on nutrient cycling in buffer strips including grass filter strips. Can determine effects of vegetation type on buffer effectiveness.
Major Drawbacks	While REMM does model complex nutrient cycling for grass and forested buffers, it is most applicable to rural areas.

BUFFER HYDROLOGY

The riparian system is characterized in the model as consisting of three zones parallel to the stream and representing increasing levels of management away from the stream. These zones include a narrow, undisturbed forest area adjacent to the stream for protecting the stream bank and aquatic environment, an area with managed woody vegetation for sequestering sediment and nutrients from upland runoff, and a grass strip for dispersal of incoming upland surface runoff and sediment and nutrient deposition.

The soil is characterized in three layers through which vertical and lateral movement of water and associated dissolved nutrients are simulated. Water movement and storage are characterized by processes of interception, evapotranspiration, infiltration, vertical drainage, surface runoff, subsurface lateral flow, upward flux from the water table in response to evapotranspiration losses, and return flow. These processes are simulated for each of the three zones. The storage and movement of water between the zones is based on a combination of mass balance and rate controlled approaches.

Each of these processes is simulated on a daily basis and described briefly in the following paragraphs. For a more complete description of the processes and the equations used, the reader is referred to Inamdar et al. (1999a).

Canopy interception is an exponential function of the canopy storage capacity and the amount of daily rainfall and is simulated using a modified form of the Thomas and Beasley (1986) equation. Potential rates of leaf evaporation and transpiration are both computed using a modified form of the Penman Monteith equation (Running and Coughlan 1988). Unsaturated soil hydraulic conductivity is described by Campbell's (1974) equation. Soil evaporation is computed in two stages (Gardner and Hillel 1962).

Infiltration in the model is simulated using a modified form of the Green-Ampt equation (Stone et al. 1994). Surface runoff entering the riparian area is routed downslope using a simplified procedure based on the depth of runoff and flow velocity.

Evapotranspiration flux is determined using the Darcy Buckingham equation as described by Skaggs (1978). Actual transpiration loss is limited by the availability of moisture in the soil and competition among the roots of the various plant types present. As the soil dries, water extraction depends on both the root distribution and the rate at which water can move to the roots. The maximum rate of water uptake from a layer is limited by its soil hydraulic conductivity. Vertical unsaturated conductivity is simulated as a function of the soil moisture content using Campbell's equation (Campbell 1974). This allows any excess demand that is not realized by a layer to be transferred to the layer below.

Subsurface lateral movement is assumed to occur when a water table builds up over the restricting soil layer. The lateral movement of the water is simulated using Darcy's equation. In the model, saturated lateral soil conductivity is assumed the same as vertical saturated conductivity. Down-slope subsurface flow between the component zones is driven by the gradient of the water table. The potential hydraulic gradient that determines the subsurface movement from zone 1 to the stream is assumed equal to the smaller of the surface slope of zone 1 and the gradient from the water table elevation from the mid point of zone 1 to the stream thalweg. Stream thalweg is a user-defined input.

Vegetation and associated litter material provide physical barriers to water and sediment transport over the ground surface. Deposition of organic matter by plants provides a substrate supporting important biological transformations of chemicals in the soil. Plants also sequester nutrients such as nitrogen and phosphorus that contribute to water pollution. The zone immediately adjacent to the stream helps to protect the stream bank and aquatic habitat.

The litter layer is important as the locus for the mixing of surface water with the soil surface. This mixing process results in an equilibrium of dissolved and adsorbed chemical concentrations, which determines amounts of chemicals that are subsequently leached, deposited on the ground surface, or carried along in surface runoff. Concentrations of dissolved and adsorbed chemicals are recalculated as water moves through each of the other soil layers.

Climate parameters required are the following: rainfall amount and duration, solar radiation, maximum and minimum air temperatures, dew point temperatures. If actual measured data are not available the model uses a subroutine, CLIGEN, to generate climate data. The model operates on a daily time step.

EROSION AND SEDIMENT

Erosion and sediment is calculated separately for each of the three riparian zones. The Universal Soil Loss Equation (USLE) is used to predict erosion for each storm event (Wischmeier and Smith 1978). Parameterization of the USLE for forested area is accomplished using guidelines presented by Dissmeyer and Foster (1984). The method used for sediment routing uses equations developed by Foster et al. (1981) and Lane (1982) and is as applied in the AGNPS model (Young et al. 1989). The effective transport capacity is computed using a modification of the Bagnold stream power equations (Bagnold 1966). A detailed presentation is made in Young et al. (1989).

NUTRIENT DYNAMICS

REMM is also capable of modeling nutrient dynamics. Simulation of the carbon dynamics is based on the Century Model (Parton et al. 1984, Inamdar et al. 1999b). Stoichiometric relationships are assumed among C, N, and P in organic matter. N and P are released and immobilized in proportion to transformations of C. The decomposition rates of the organic matter pools area calculated according to first-order rate equations modified by temperature, moisture and C:N and C:P ratios (Inamdar et al. 1999b).

Nitrification is calculated with a first-order rate equation modified by temperature, moisture and pH. Rate coefficients are determined following the approach of Reuss and Innis (1977) and Godwin and Jones (1991) based upon a Michaelis-Menten (Monod) function.

Phosphorous is simulated using the EIPC model (Jones et al. 1984) with two pools of inorganic P unavailable to plant uptake, and a labile form that may be dissolved or adsorbed according to a partitioning coefficient (Williams et al. 1984).

REMM rate constants may be a good source of K values for process modeling. Sources for upland data may be from site monitoring or from the use of an upland model such as GLEAMS (Knisel et al. 1993).

Output from REMM includes predicted sediment yields, depth to groundwater, and predicted C-N-P distribution throughout the system in many forms. BMP effectiveness may be evaluated on the basis of comparison of incoming and outgoing loads.

RECOMMENDATIONS

The REMM model is only model reviewed that integrates subsurface flow and groundwater interaction when simulating buffers. REMM also closely models the N, P, and C cycles in the three buffer zones, although this requires extensive field data to simulate. The most useful information from the REMM model may be the default rate constants used in these calculations. Its overall structure is likely beyond what is required in SWMM.

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