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The Application of Monitoring and Modeling in Watershed Management

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showing City of Elmhurst employees recovering a dissolved oxygen probe from Salt Creek in Illinois as part of a stream dissolved oxygen feasibility study.

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From the Editor's Desk

This issue of the Bulletin focuses on the practical application of monitoring and modeling in stormwater and watershed management. Monitoring and modeling are invaluable for assessing watersheds, identifying problems, analyzing alternatives, or measuring progress toward a desired goal or endpoint; however, these tools are also quite challenging technically. Because implementing a monitoring program or running a model often requires specialized expertise, these activities are traditionally left to researchers at universities or at state or federal agencies. Yet monitoring and modeling have become necessary and integral components of local watershed and stormwater management programs because of an increasing focus on producing measurable results with which to justify program budgets.

Local watershed and stormwater managers must carefully weigh which monitoring and modeling approaches are most appropriate for the task at hand. For example, important considerations include, data availability and quality, funding and staff required, time period for which data are needed, the need for technical expertise, and available equipment and software. Integral to the successful implementation of any approach is a clear definition and understanding of what information is needed and the most effective and efficient way to obtain the data. For monitoring, this includes the selection of the most appropriate sampling methods and parameters. Ideally, monitoring programs should also identify, in advance, the number of samples needed to generate a statistically valid data set. However, in some instances, collecting just enough samples for characterization purposes may be sufficient to help better define the problem and to generate more sophisticated questions as part of "next step" efforts.

The use of models is pervasive—and attractive—as a way to readily obtain answers to difficult questions from the comfort of one's own workstation. However, because all models require the oversimplification of complex processes, careful model selection is paramount. Many options are available, such as lumped or spatially distributed models, statistical or physically based models, and models that simulate annual versus daily time steps. In choosing a model, one also must consider the type of data needed to run the model, data quality, and whether the model's spatial and temporal scale is appropriate for the task. It's true that a model is only as good as the data used for the simulations.

In this issue, the *Bulletin* highlights how monitoring and modeling can be effectively transferred from research-based efforts to practical tools that help managers address everyday decision-making related to watershed and stormwater management. This collection of research articles and vignettes presents a wide array of monitoring and modeling applications that reflect the versatility of these tools to address watershed and stormwater program needs and issues. Despite the range of topics, some common threads are apparent. For example, several authors explain that cooperation among various agencies, organizations, and individual stakeholders was an integral part of a project's success. This theme is also repeated in our Ask the Experts section, where researchers and representatives from federal, state, and local government agencies share their perspectives on the practical applications of monitoring and modeling for watershed assessment and how these tools are used to inform decision making to protect or improve watershed health.

Lilly and others present the stormwater version of "Dirty Jobs" as they use monitoring techniques to investigate the presence and magnitude of bacterial and nutrient loads from illicit discharges in urban watersheds in Maryland. The results uncover a potentially significant pollution source that has typically not been accounted for in watershed-based pollutant load estimates. This article emphasizes the need for greater municipal coordination to find and eliminate these pollutant sources. The authors demonstrate how a relatively low-cost monitoring effort could lead to cost-effective pollutant load reductions and argue that pollution source tracking is paramount to identifying effective pollutant reduction strategies.

Similarly, in Salt Creek and the East Branch DuPage River in Illinois, **McCracken and Huff** ask what pollution sources are contributing to the dissolved oxygen (DO) total maximum daily load (TMDL). McCracken and Huff describe a process of cooperation among local stakeholders, the regulated community, and the Illinois Environmental Protection Agency to address objections to TMDL reductions that focused on expensive upgrades to wastewater treatment plants. After questioning the information used to develop the TMDL, stakeholders developed a local partnership that revisited and revised the monitoring network and the data used to calibrate and validate the models. The successful use of monitoring and modeling in this case has resulted in newly defined, soon-to-be implemented projects that will more effectively address the DO TMDL.

Hawley and others further demonstrate the ability to change a course of action by using the right tools. Sanitation District No. 1 of Northern Kentucky and its project partners use monitoring and modeling as part of an adaptive watershed management strategy to plan and design system improvements. In this article, the authors describe the infiltration and inflow (I/I) problem encountered in many developed watersheds and how innovative monitoring and modeling efforts helped them find a more sustainable solution. Hawley and others describe a collaborative project that addressed the I/I problem while also finding cost-effective solutions to protect infrastructure from downstream erosion and improve water quality and habitat conditions. The detailed quantitative methods and illustrative results and metrics in this research article showcase how monitoring and modeling can highlight new solutions to old problems.

Whereas Hawley and others focus on project-specific monitoring, **Morris and Johnson** describe a comprehensive monitoring program that uses the monitoring results to understand baseline conditions and temporal trends for the San Gabriel River watershed in California. This article illustrates the basics of a "textbook" monitoring program, in which interested stakeholders work together to define and implement a monitoring program that uses multiple types of indicators to better define and answer management questions at the local, regional, and state levels.

The vignette, **Local Monitoring Data Used To Support Watershed-Based Hydrologic Modeling of Downscaled Climate Model Output,** further illustrates the usefulness of long-term monitoring programs for local decision making. This vignette on water supply management and anticipated climate and land use changes demonstrates how existing monitoring networks and modeling efforts can provide essential information supporting the incorporation of nontraditional, but increasingly relevant, climate change issues into local decisions about future water supply issues.

In the final research article and the second vignette of this issue, authors use models to better define and understand local conditions. For example, **Yagow and others** offer a new approach for developing targeted pollutant load reduction strategies to meet TMDL requirements in smaller, nested subwatersheds with no quantifiable sediment water quality standards. The authors use an empirical modeling approach to develop the *disaggregate method* as a way to help watersheds in Virginia meet sediment load reductions for the protection of aquatic life.

The vignette, **Locally Derived Water Balance Method To Evaluate Realistic Outcomes for Runoff Reduction in St. Louis, Missouri**, describes the use of a model to evaluate how future criteria for evaluating the effectiveness of stormwater management controls are applicable to local conditions in Missouri.

I hope that this issue illustrates the many uses—and the usefulness—of monitoring and modeling. The successful application of these tools in the projects described here can be attributed, in part, to their well-defined uses and data collection needs and to the collaborative nature of the projects. Many of the authors also see such modeling and monitoring efforts as stepping stones from which to reach future goals. So strap on your waders, pull out your algorithms, and enjoy this issue of the *Bulletin*.

Neely L. Law, PhD, Editor-in-Chief

Pollution Loading from Illicit Sewage Discharges in Two Mid-Atlantic Subwatersheds and Implications for Nutrient and Bacterial Total Maximum Daily Loads

Lori A. Lilly,[a]* Bill P. Stack,[b] and Deb S. Caraco[c]

Abstract

The Center for Watershed Protection (Center) collaborated with local jurisdictions to comprehensively detect and quantify the nutrient and bacterial loads from nonstormwater discharges in two Mid-Atlantic subwatersheds. Water quality analyses indicate that the discharges are probably from sewage sources and appear to be a significant, yet unaccounted for, source of pollution to the Chesapeake Bay and its tributaries. The discharges represent a controllable source of pollution whose systematic elimination could result in significant progress toward meeting nutrient and bacterial total maximum daily load (TMDL) reduction requirements.

The Center followed a comprehensive procedure for detecting, tracking, and eliminating pollution sources that included (1) using threshold criteria, such as ammonia and bacteria to determine the presence of illicit sewage discharges; (2) estimating instantaneous pollutant loadings from the dry weather flowing outfalls; and (3) comparing the illicit sewage discharge pollutant load to the watershed load, as estimated from grab samples taken from a downstream, instream location. This analysis shows that the elimination of illicit sewage discharges has the potential to achieve up to 21% of the estimated TMDL phosphorus reduction, 43% of the estimated TMDL nitrogen reduction, and 51% of the estimated TMDL bacterial reduction in one of the study subwatersheds.

Improvements in illicit discharge detection and elimination programs may help communities achieve their targeted pollutant load reductions and can be an important first step for addressing water quality impairments in urban watersheds. Detecting and tracking illicit discharge sources can be a labor-intensive process for government staff that can potentially be offset through collaborative efforts with watershed organizations and volunteer water quality monitoring programs.

Introduction

Studies have shown that dry weather flows from the storm drain system may contribute more than wet weather stormwater flows to the annual discharge mass for some pollutants (US Environmental Protection Agency [USEPA] 1983b; Duke 1997; Pitt and McLean 1986). McPherson et al. (2005) found that dry weather flow in the Ballona Creek watershed in Los Angeles, California, contributed more than 40% of the pollutant loading for each of the following constituents: nitrate-nitrogen, nitrite-nitrogen, ammonia-nitrogen, total Kjeldahl nitrogen, and total phosphorus (TP). Dry weather flows can stem from car wash discharges, water main breaks, and illicit sewage discharges, among other sources.

In particular, the cumulative illicit sewage discharges into a storm drain system can have a significant water quality impact by introducing high nutrient and bacterial loads with toxic and pathogenic effects. They are often missed by ineffective and/or inefficiently implemented municipal separate storm sewer system (MS4) illicit discharge detection and elimination (IDDE) programs because such programs target larger storm drain or sewer issues (e.g., by limiting illicit discharge monitoring to pipes greater than 36 inches[1] [91.4 cm] in diameter). Furthermore, sewage discharges are relatively small, but persistent, problems that are often not considered part of the large capital improvement projects required under USEPA consent decrees to manage sanitary sewer overflows (SSOs). Finally, although a part of each National Pollutant Discharge Elimination System (NPDES) MS4 permit requires an IDDE program, incentives for implementing *effective* IDDE programs are lacking. For example, the USEPA Chesapeake Bay Program does not currently have a system for crediting local governments that fix illicit discharges through the total maximum daily load (TMDL) process. And in some instances, regulators developing TMDLs assume that

[1] English units have been used throughout this paper due to their common use in engineering and infrastructure applications. Metric equivalents or example conversions are provided.

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loadings from sewage discharges will be addressed through actions such as consent decrees. For this reason, they do not "count" loadings from illicit discharges or include them in the background waste load from urban runoff. Therefore, a best management practice (BMP) that eliminates this source, such as IDDE, cannot be credited as part of nutrient load reductions. Local governments find themselves scrambling to undertake the enormous task of nutrient accounting for practices and programs in highly urban landscapes, where substantial benefit could be achieved through the investment of resources into sewage discharge elimination. Benefits could be seen in terms of water quality improvements as well as agency credit for eliminating pollution sources. When federal and state regulatory agencies either fail to understand the importance of the issue or lack the resources to adequately address it, program implementation at the local level can become more of a "check the box" strategy rather than an actual tool to be used for improving water quality.

The purpose of this paper is to present data from two case studies showing that water quality goals in some watersheds may be achieved only if dry weather illicit sewage discharges are addressed within the overall watershed restoration framework. By quantifying dry weather pollutant loading from illicit sewage discharges in two subwatersheds, this paper illustrates the pervasiveness and cumulative impact of dry weather illicit sewage discharges along with the potential value of IDDE as a BMP for achieving goals set forth in TMDLs for impaired waters. Furthermore, this paper presents watershed management implications and recommendations related to sewage discharge elimination based on results from the case studies. In particular, we recommend increasing the priority of sewage discharge elimination within the overall strategy for watershed restoration.

Regulatory Background

Uncontrolled or unpermitted sewage leaks and discharges come under the broad regulatory heading of "illicit discharge." The NPDES Program defines an illicit discharge as "any discharge to a municipal separate storm sewer that is not composed entirely of stormwater, except discharges pursuant to a NPDES permit and discharges resulting from fire-fighting activities." 40 CFR 122.26(b)(2) (1999). NPDES permits may also authorize discharges as long as permit requirements, such as established effluent limits, are being met.

Each Phase I and Phase II MS4 is required to develop and implement a stormwater management program to reduce contamination of stormwater runoff and prohibit illicit discharges. The stormwater management program must include an IDDE program with three primary components—detection, tracking, and elimination of illicit discharges. As part of its IDDE program, each Phase I and Phase II MS4 should have an outfall screening program, education measures, a local ordinance prohibiting illicit discharges, and measurable goals. The programs of Phase I versus Phase II MS4s differ in two main ways. First, Phase I MS4s are explicitly required to screen "major" outfalls—that is, those greater than 36 inches (91.4 cm) in diameter, whereas Phase II MS4s do not have this requirement. Second, Phase I MS4s must use a very prescriptive set of water quality parameters for screening, whereas, in many states (e.g., Maryland, Pennsylvania, and Illinois), Phase II MS4s are not required to conduct water quality testing as part of the screening program.

Stormwater and Wastewater History

Understanding the potential impact of illicit sewage discharges on receiving water quality requires an awareness of the nexus between the sanitary sewer and stormwater pipe networks. Sewer systems are either separate or combined. Combined sanitary systems (CSS) are pipe networks that convey stormwater and sewage together. The comingled flow is transported to a wastewater treatment plant except when large storm flows exceed the capacity of the conveyance system or pipe network. In such cases, the excess untreated sewage–stormwater mixture is diverted to a nearby water course; this is referred to as a combined sewer overflow (CSO).

In separate sanitary sewage (SSS) systems, sewage and stormwater are conveyed in separate pipe networks. Sewage is collected from homes, businesses, and industries and conveyed to a wastewater treatment plant—without mixing with stormwater, at least in theory. In the early 1900s, SSS replaced the CSS as the predominant type of conveyance system in the United States. While more advanced than CSS, decades of neglect have resulted in systemic deterioration of SSS that allows groundwater and stormwater to enter these systems through breaks and leaks in the pipe network. As a result of inflow and infiltration, large storm events, and other causes, USEPA estimates at least 23,000–75,000 sanitary sewer overflows (SSOs) per year (not including sewage backups into buildings; USEPA 2004). Most large SSS communities with SSO issues are regulated by state agencies and/or USEPA under consent actions that require structural repairs and proactive maintenance. Receiving waters served by SSS are plagued by small leaks, breaks, and maintenance-related discharges (clogging with grease) that are easily overlooked by sewer evaluations. Sewer evaluations also can overlook direct discharges of sewage into the storm drainage system from individual homes and businesses. These discharges—which can be easily identified through IDDE programs—are a major source of bacterial and nutrient impairment.

Communities across the United States are spending billions of dollars to address CSOs and SSOs through repairs or sewer capacity expansions intended to reduce major overflows that occur primarily during storm events. However, recent studies by Kaushal et al. (2011) and the Center (CWP 2011) cast doubt on whether such efforts are adequate to address all sewage-related impacts to water quality. These studies were conducted in the City of Baltimore, Maryland, a community served by an SSS. In response to a consent decree, Baltimore has spent millions of dollars on wet weather repairs to address SSOs (City of Baltimore 2010), but both studies indicate that these repairs have had little impact on *dry weather* discharges. Specifically, Kaushal et al. (2011) studied six urban tributaries in the Baltimore region. Using stable isotopic techniques, they found that sewage was the predominant source of nitrogen load during baseflow conditions, even after repairs to the wastewater system were complete. Similarly, a restoration plan for Baltimore Harbor found little or no improvement in nutrient or bacterial loading after years of sewer system repairs in Baltimore that targeted infrastructure limitations causing wet weather SSOs (CWP 2011). The authors determined that this finding was due to the underlying persistent pollution loads from dry weather sewage sources.

The persistence of water course impairments, despite substantial investments in infrastructure repair, is due, in part, to scale. Municipal programs that aim to eliminate CSOs and SSOs predominantly target wet weather events through the repair and replacement of pipes and pumping stations. Although dry weather occurrences are addressed through proactive operation and maintenance protocols, as specified in consent decrees, widespread small sewage leaks continue to discharge to surface waters. This is the case especially for sewer laterals—that is, private connections to homes and businesses that are often connected to the municipal system by private contractors with limited public oversight.

Case Study Descriptions

This study included estimates of illicit sewage discharge pollution loads for two Chesapeake Bay subwatersheds: Western Run, a 5.4-square-mile (mi^2; 12.9-km^2) subwatershed in northwestern Baltimore City, and Sligo Creek, a 9.6-mi^2 (24.9-km^2) subwatershed in Montgomery County, Maryland, just north of the District of Columbia (Figure 1). Both watersheds drain low- to medium-density residential land uses (Table 1) on the outskirts of major metropolitan areas. These watersheds are typical of many urban streams with limited floodplain connectivity, armored banks, channel incision, and impaired water quality (in terms of bacteria, sediment, and nutrients; see Table 2). Each subwatershed is within a Phase I MS4 jurisdiction and therefore regulated for illicit discharges. The City of Baltimore and Montgomery County implement IDDE programs in Western Run and Sligo Creek, respectively.

Figure 1. Two subwatershed case studies: Western Run and Sligo Creek. Image courtesy of Google Maps.

Table 1. Size and land use distribution of sampled watersheds.

Subwatershed	Area (mi^2)	Impervious Cover (%)	Percentage Land Use in Watershed					
			Commercial	High-Density Residential	Medium-Density Residential	Low-Density Residential	Open Space	Other
Western Run	5.4	~33.0	3.9	7.1	27.0	41.5	2.2	18.3
Sligo Creek	9.6[a]	33.6	6.5	11.8	60.6	—	—	21.1

[a] This area reflects only the Montgomery County portion of the watershed.
Note: 1 mi^2 = 2.6 km^2

Table 2. Water quality impairments in sampled watersheds.

TMDL	Anacostia (Sligo Creek)	Jones Falls (Western Run)
Bacteria	X	X
Sediment	X	X
Nutrients	X	X
PCBs	X	
Trash	X	
Zinc		X
Copper and Lead		X

Note: PCBs, polychlorinated biphenyls.

Methods

The methods included collecting flow and water quality data from storm drain outfalls and instream locations in each subwatershed over several days within a two-week period. Fieldwork took place in June 2010 for Western Run and January 2011 for Sligo Creek. Teams of three to four Center and local government staff, along with watershed group volunteers visited all outfalls in the subwatershed by walking entire stream reaches. Using the outfall reconnaissance inventory technique described by Brown et al. (2004), the teams investigated outfalls with dry weather flow and screened them for a number of illicit discharge indicators, including physical indicators, such as pipe benthic growth, odor, flow lines, and cracking or spalling (flaking or chipping) of the pipe; bacteria; and chemical indicators, specifically ammonia, detergents, potassium, and fluoride. The teams collected three samples from each flowing outfall and analyzed them as indicated in Table 3. Sample collection methods included conditioning the sample bottle with dry weather flow (i.e., rinsing the sample bottle several times with sample water before collection) and then directly filling a single bottle by holding it under the discharge from each drain until the bottle was full. We also collected instream grab samples on only one day and analyzed them for total nitrogen (TN) and TP.

The teams took a flow measurement at each outfall using either a timed volumetric method, cross-section–velocity method, or weir equation[2] (depending on the conditions at a given location). Teams also collected instream flow measurements in the upper, middle, and lower regions of each watershed using a pygmy meter. Standard conversions and assumptions for outfalls (i.e., that flow remained constant over the entire day) allowed for daily flow estimates.

We adjusted the grab sample concentrations by subtracting a background surface water concentration of TN (1.0 mg/L) and TP (0.02 mg/L) for each grab sample to provide a conservative estimate of pollutant load. The background nutrient concentrations are based on data collected by the US Geological Survey's National Water-Quality Assessment program in natural watersheds (average TN = 0.26 mg/L; average TP = 0.022 mg/L; Clark et al. 2000) as well as data collected by Center staff from "clean" outfalls—that is, those that did not exceed illicit discharge screening parameter thresholds—in Baltimore, Maryland (average TN = 2.0964 mg/L; average TP = 0.0539 mg/L; Lilly and Sturm 2010). We used the adjusted concentrations to estimate an annual load with the assumption that the illicit discharge flow rate remained constant over an entire year. The diurnal and weekly variations in outfall discharges, however, may skew the estimates of the cumulative outfall discharge, in contrast to the estimates from the instream grab samples. Likewise, temporal and seasonal differences, as well as differences in land cover and riparian characteristics of the subwatersheds, probably contributed to differences observed between the subwatersheds. Further sampling could address these issues. Although one should use extrapolated estimates with caution, they are useful for estimating the potential contribution of the sewage discharge to the total loading. The limited budget of this project could not accommodate a more frequent and regular monitoring program that would have allowed for more accurate quantification of seasonal/diurnal variability and refined annual load rates.

We used a variation of the flow chart method (Brown et al. 2004) to distinguish among three major types of discharges: wastewater, wash water, and tap water (Figure 2). Subsequently, teams tracked these discharges to their sources when possible. When the threshold levels were not exceeded, we assumed that the source was groundwater and was not composed of sewage, wash water, or tap water. The flow chart method helped determine the presence of a potential illicit discharge and loading from suspect outfalls. Wastewater (sewage) discharges include sanitary wastes, as indicated by the presence of detergents or other surfactants and high ammonia concentrations. Wash water discharges can include domestic wash water (e.g., from a cross-connected washing machine) as well as a wide range of industrial process waters. Detergents are typically present in wash water, but the ratio of ammonia to potassium is generally lower than that found in wastewater. Tap water

[2] Flow = [3.1 × wetted width (feet) × depth (feet)]1.5. This method was used only with a free-flowing outfall and when the depth of flow was relatively uniform.

Table 3. Water quality sample analysis.

Sample	Parameter Analyzed	Equipment	Method	Location	Specifications	Notes
Field Measurement	Ammonia	Hannah HI 93715 or Milwaukee MI405	Adaptation of the Nessler method (USEPA 1979, method no. 350.2)	Field	Range: 0.1–9.99 mg/L Accuracy: ± 0.1 mg/L	Meter zeroed with sample water before each measurement
Sample 1	Fluoride	Hannah HI 93729 Low-Range Photometer	Adaptation of the SPADNS method (USEPA 1979, method no. 340.1)	Baltimore City's Ashburton lab, Baltimore, MD, or Maryland National Capital Park and Planning Commission lab	Range: 0–2.00 mg/L Resolution: 0.01 mg/L Precision: ± 0.03 mg/L at 1.00 mg/L	Meter zeroed before each reading using a standard created with distilled water reacted with reagent
	Anionic surfactants	Chemetrics Detergent Kit	USEPA (1983a) method no. 425.1		Range: 0–3 ppm	
	Potassium	Horiba Cardy Compact Ion Meter C-131	As per manufacturer: nitrate ion electrode method		Range: 0–99 · 100 ppm; Resolution: 1.0 ppm (0–99 ppm), 100 ppm (10–99 ·10 ppm), and 100 ppm (10–99 · 100 ppm)	Two-point calibration conducted before each set of sample readings, where the meter was standardized first to 20 x 100 ppm and then to 15 x 10 ppm
Sample 2	TN	—	USEPA (1979) method no. 353.2	Contracted to Chesapeake Bay lab (Solomons, MD) and Horns Point lab (Cambridge, MD) for analysis	Labs undergo a blind audit; average percentage difference of the analysis compared to the prepared reference concentration, which is between 5% and 10%	Samples frozen at end of field day and mailed on ice to the lab
	TP	—	USEPA (1979) method no. 353.2			
Sample 3	*E. coli* and total coliform	3M Petrifilm plates	As per manufacturer: Incubated at approximately 35°C for 24 hours ± 1 hour; red and blue colonies with gas enumerated manually or with a 3M Plate Reader	Center office in Ellicott City, MD		100 mL of sample collected in a sterile bottle and plated no more than six hours after collection; a 1-mL subsample plated to grow *E. coli* as "blue" colonies and total coliform as "red" colonies; the colonies of each are counted, multiplied by 100 and reported as colony forming units, or CFUs, per 100 mL

Notes: Hanna Instruments, Smithfield, RI; Chemtrics, Inc, Midland, VA; Horiba Instruments Inc, Irvine, CA; 3M Microbiology Products, St Paul, MN.

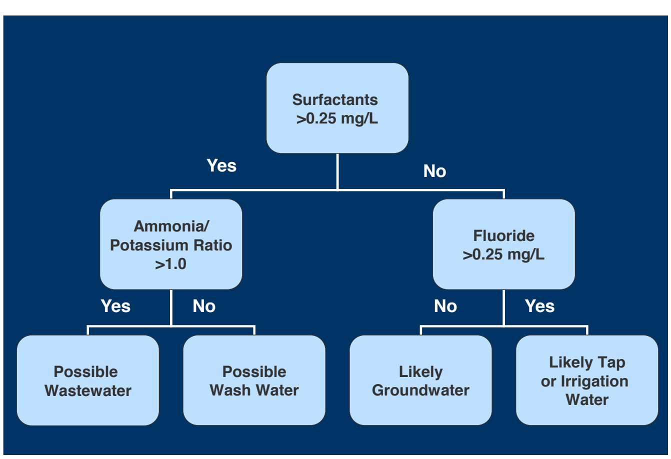

Figure 2. Flow chart method used to distinguish among potential illicit discharges. Source: Brown et al. 2004.

(which includes no detergents) often originates from a broken water line and, although not illicit, is often a target repair for a community.

Threshold levels for illicit discharge screening parameters, defined in Table 4, stem primarily from research conducted for the preparation of an IDDE guidance manual for Phase II MS4s (Brown et al. 2004). If an illicit discharge was suspected based on the initial sampling, typically one team (designated the "tracking team") would immediately leave the stream and attempt to track the source of the contaminated flow to the source. The team would conduct visual screening and chemical monitoring in the upstream storm drain network to attempt to confirm the source of the illicit discharge.

Table 4. Threshold levels for screening parameters used in outfall screening.

Parameter	Threshold	Source
Ammonia	>0.1 mg/L	Brown et al. (2004)
E. coli	235 CFU/100 mL (grab sample)	USEPA (1986)
Total coliform	10,000 CFU/100 mL (grab sample)	California state standard (Dorfman and Rosselot 2011)
Fluoride	0.25 mg/L	Brown et al. (2004)
Detergents	0.25 mg/L	Brown et al. (2004)
Potassium	5 ppm	Guidance extrapolated from Lilly and Sturm (2010)

Measuring TN and TP concentrations at the outfall, along with flow, allowed for a quantification of the nutrient load from individual outfalls suspected of having sewage contamination. For example, outfall CCA8 from Sligo Creek had an ammonia concentration of 3.62 mg/L, a detergent concentration of 0.75 mg/L, and 15,000 colony forming units (CFU) of *Escherichia coli* per 100 mL. These concentrations are much higher than one would find in ambient stream or groundwater conditions and are most likely due to the presence of sewage. TN measured at this outfall was 6.5 mg/L and flow was 0.05 cubic feet per second (cfs) (0.0014 m^3/s). A conservative nitrogen load estimate, made by subtracting 1.0 mg/L from the original concentration, gives a final estimated annual load, using standard conversions[3], of 539 pounds/year (244 kg/year).

Case Study Results: Western Run and Sligo Creek

Illicit sewage discharges were pervasive in the two case study watersheds. Of the 313 outfalls assessed, 103 (33%) had dry weather flow (Table 5). Of the outfalls with dry weather flow, 78% exceeded water quality parameters that indicate the presence of illicit discharges. Ammonia, the primary wastewater indicator, was present in half of the discharges investigated. Approximately 40% of the discharges contained fluoride, a potable (i.e., tap) water indicator. Detergents, indicators of wash water or wastewater, were present in one-third of the discharges. More than one-third of all discharges had *E. coli* concentrations above the USEPA (1986) threshold for contact recreation, and half of the flowing outfalls in Western Run exceeded *E. coli* thresholds.

Discharge

The cumulative discharge from all suspected storm drain outfalls in Sligo Creek was approximately 1.35 million gallons/day (5,110 m^3/day)—approximately equal to the total instream discharge (1.26 million gallons/day [4,770 m^3/day]). In contrast, the stormwater outfall discharge in Western Run (0.25 million gallons/day [946 m^3/day]) was only 9% of the total instream discharge (2.77 million gallons/day [10,486 m^3 /day]).

Nutrients

Based on the downstream instream flow and nutrient sample collection in each subwatershed, the estimated daily nitrogen load was 24–31 pounds/day (10.9–14.1 kg/day) and the daily phosphorus load was 0.15–1.0 pounds/day (0.068-0.45 kg/day); (Table 6). In Sligo Creek, the TN load from outfalls suspected of having illicit discharges made up 97% of the instream load, and phosphorus loadings from suspected discharges composed more than 500% of the

Table 5. Outfall summary.

	Sligo Creek	Western Run	Sum
Total outfalls assessed	213	100	313
Outfalls with dry weather flow	58 (27%)	45 (45%)	103 (33%)
No. of discharges exceeding threshold levels for ammonia, fluoride, or detergents	47 (80%)	33 (73%)	80 (78%)
No. of discharges with potential wastewater or other discharge of unknown origin (ammonia >0.1 mg/L)	35 (60%)	16 (36%)	51 (50%)
No. of potential tap water discharges (fluoride >0.25 mg/L)	17 (29%)	23 (51%)	40 (39%)
No. of potential wash water discharges (anionic surfactants >0.25 mg/L)	24 (41%)	11 (24%)	35 (34%)
Outfalls with *E. coli* above USEPA threshold for contact recreation (>235 CFU/100 mL)	14 (24%)	24 (53%)	38 (37%)

[3] Pounds per cubic foot = (nitrogen concentration × 28.317)/453,592; pounds per year = pounds per cubic foot × cfs × 31,557,600. 1 pound = 0.454 kg and 1 cubic foot = 0.028 cubic meters.

instream load. In Western Run, the TN load from outfalls suspected of having illicit discharges made up 17% of the instream load, and phosphorus loadings from suspected discharges composed 58% of the instream load. Instream flow measurements in each subwatershed were collected only on day 1 of the outfall screening. In each subwatershed, outfall screening took place on multiple field days over an approximately two-week period. The refinements needed in sampling methods for calculating load estimates may overcome the limitations of this study that resulted in the phosphorus outfall load exceeding the instream load in Sligo Creek.

Table 6. Instream sample (farthest downstream point).

	Sligo Creek	Western Run
Ammonia (mg/L)	N/A	0.13
E. coli (CFU/100 mL)	100	20,000
Discharge (cfs)	1.9	4.3
TN (mg/L)	2.4	1.3
TN Load (pounds/day)	2.4	31.0
TP (mg/L)	0.02	0.04
TP Load (pounds/day)	0.2	1.0

Bacteria

The downstream instream bacterial concentration was much higher in Western Run (20,000 CFU/100 mL) than in Sligo Creek (100 CFU/100 mL), probably because of a large sewer line break found upstream of the instream monitoring location during the sampling in Western Run. Average *E. coli* concentrations from outfalls were high in both subwatersheds: 1,345 CFU/100 mL in Sligo Creek and 321,462 CFU/100 mL in Western Run. The majority of outfall *E. coli* came from those outfalls that exceeded illicit discharge parameter thresholds. For example, 96% of the *E. coli* from outfalls in Sligo Creek and 87% of the *E. coli* from outfalls in Western Run came from those outfalls that were suspected of having illicit discharges.

Tracking Sources

Tracking the source of illicit discharges may be straightforward and even obvious in some cases; however, in other cases, a lot of detective work is required. Many of the illicit discharges in Western Run were tracked to specific sources. In one instance, dye testing confirmed that a sewage discharge resulted from leakage from the sanitary system into the storm drain system. In another instance, sewage discharge was confirmed from a broken sanitary pipe. In Sligo Creek, a handful of the 47 potential discharges initially found through field screening have been successfully tracked to a source. One investigation required approximately 55 total staff hours; the effort was complicated by the fact that the source was a blend of at least four different sewage sources. Several source investigations are ongoing.

Management Implications and Recommendations

The elimination of a watershed's illicit discharges may have significant cost and management implications if considered as part of watershed-wide pollutant load reductions. The results of this study suggest that (1) IDDE can play a significant role in meeting TMDL requirements; (2) IDDE, although labor-intensive, is a cost-effective way to meet pollutant load targets; (3) detection and load estimation methods must be refined; (4) municipalities can work with the volunteer monitoring community to find illicit discharges; and (5) finding and removing sources requires significant coordination and persistence among local agencies.

IDDE Can Play a Significant Role in Meeting TMDL Requirements

IDDE is a tool that can be used to identify sewage discharges and meet both bacterial and nutrient TMDLs in local waterways. For example, although Western Run itself has no specific nutrient impairment, the City of Baltimore will have to meet jurisdiction-wide nutrient load reduction targets (18% for TN and 34% for TP) as part of the State of Maryland's strategy to address the Chesapeake Bay nutrient TMDL (Maryland Phase I Watershed Implementation Plan 2010).[4] Since Western Run is a subwatershed of the Jones Falls watershed, reduction targets were applied to loading estimates from the Lower Jones Falls small watershed action plan (CWP 2006). Comparing the load reductions of 3,015 pounds/year (1,368 kg/year) for TN and 1,025 pounds/year (465 kg/year) for TP to the loadings measured from the illicit discharges, the illicit discharge load for TP, based on the Center's field screening, was 217 pounds/year (98 kg/year) —approximately 21% of the reduction needed for Western Run (Figure 3). The illicit discharge load for TN was 1,897 pounds/year (860 kg/year)—approximately 43% of the reduction needed for the subwatershed (Figure 4). In a similar analysis for Sligo Creek, we found that the illicit discharge load represented 17% of the TN and 6% of the

[4] More refined jurisdiction-wide targets were issued in October, but not in time to be incorporated into this paper.

TP TMDL reduction. The analysis was based on the nutrient TMDL developed for the nontidal Anacostia watershed (Maryland Department of the Environment [MDE] and District of Columbia Department of the Environment 2008), which required an 80% reduction in TP and a 79% reduction in TN.

MDE developed a fecal coliform TMDL for the Jones Falls watershed in 2006. A baseline load for a subwatershed of the Jones Falls (i.e., subwatershed JON0039) is an estimated 9,152 billion most probable number (MPN)/day. MPN refers to serial dilution tests that measure the concentration of a target microbe in a sample (MDE 2006a). The TMDL allocation for the subwatershed is 430 billion MPN/day—a reduction of 8,722 billion MPN/day. Assuming that the load allocation for Western Run (with an area of 3,478 acres [14 km^2]) is proportional to that of the 7,546-acre (30.5-km^2) TMDL subwatersed, the baseline load for Western Run would be 46% of the baseline load, or 4,210 billion MPN/day. The estimated TMDL allocation for Western Run is therefore 20 billion MPN/day, or a reduction of 4,190 billion MPN/day. The illicit discharge load for bacteria estimated from Center staff field screening is 2,056 billion MPN/day, or 51% of the required bacterial reduction (Figure 5). We conducted a similar analysis for Sligo Creek and the illicit discharge

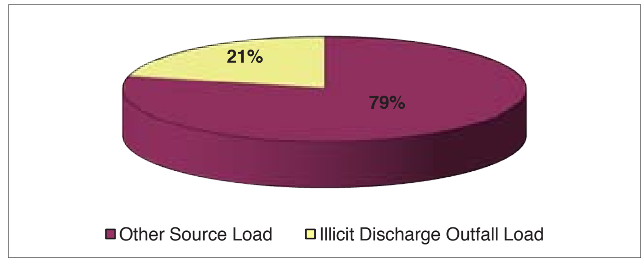

Figure 3. Illicit discharge load as a percentage of TP reduction for Western Run.

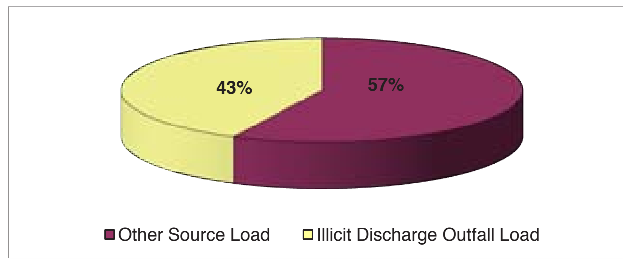

Figure 4. Illicit discharge load as a percentage of TN reduction for Western Run.

load represented 21% of the bacterial TMDL reduction. The analysis was based on a fecal coliform TMDL developed for the Anacostia watershed (MDE 2006b), which required a 93% reduction for the watershed[5].

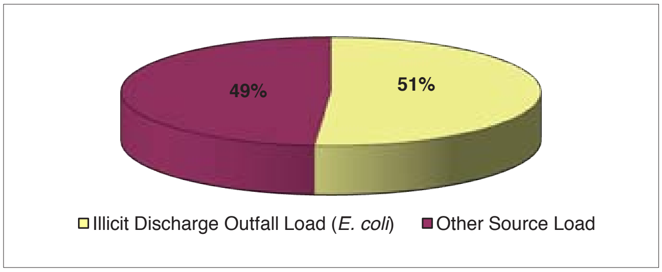

Figure 5. Illicit discharge load as a percentage of total bacterial reduction for Western Run.

This analysis suggests that some pollutant loads may be missed if the right "accounting tools" are not used to identify sources. Consequently, watershed managers and regulating agencies may be misled about the real pollutant load and the stormwater practices and programs that will most effectively reduce the pollutant load. Kaushal et al. (2011) estimated that, although highly variable, approximately 13.5% of the TN load in Baltimore area streams is from sewage sources. Some modelers perceive that pollutant loadings from sewage discharges are intermittent in nature; therefore, such discharges may be considered inconsequential to the total annual stormwater load and not incorporated as a significant source in simulation models. However, the present case studies found that illicit sewage discharges are more widespread and of much longer duration than previously thought. The state of Maryland's SSO database reports that the SSO volume in Sligo Creek from 2005 to 2010 was 224,021 gallons (848 m^3) from blockages and wet weather events. Just one of the illicit discharge flows found through this study had an estimated flow of 32,344 gallons per day (122 m^3) for a total of >9 million gallons (34,069 m^3) in a ten-month period. This is one of more than 40 illicit discharges detected in the field. The cumulative impact of many such problems to receiving waters is noteworthy. More broadly, because the illicit sewage discharge as a source has not been previously accounted for in inputs to the Chesapeake Bay Watershed Model, the actions and strategies needed to address the issue have not been a priority. In an age of pollutant accounting, local governments should be offered incentives for more comprehensively implementing their IDDE programs.

[5] Although the TMDL was developed using fecal coliform as the indicator organism, the State revised the criteria such that it is now based on water column limits for either *E.coli* or enterococci.

IDDE Is an Inexpensive Way To Meet Pollutant Load Targets

The cost of fixing illicit discharges is much less expensive per pound of nutrient reduced than other methods that treat the same load at the end of the pipe. For example, removing the annual nitrogen load associated with potential illicit discharges found in Sligo Creek would conceivably cost 18 times more if done via a practice such as a dry swale (Figure 6). IDDE can be costly in terms of staff time to track down problems, but the water quality benefit that can be achieved outweighs the upfront cost. In addition, as illustrated by Pennington et al. (2003, 1040), "communities are ill advised to rely exclusively on structural BMPs to address their water quality concerns." A holistic approach that effectively integrates both structural and nonstructural practices will be needed to address the many water quality impairments in the United States.

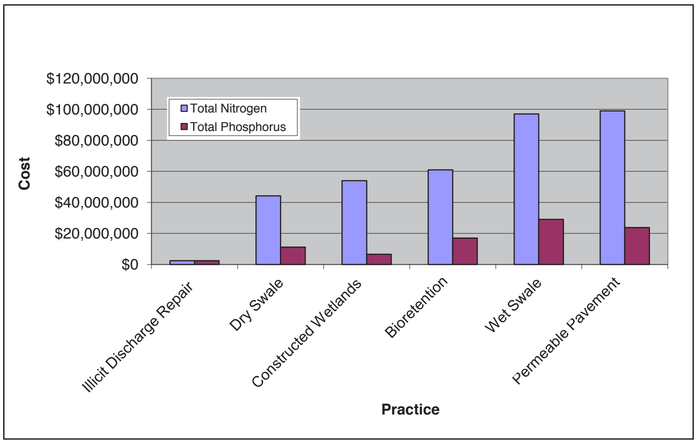

Figure 6. Costs of various practices to treat an equivalent annual load estimated from illicit discharges in Sligo Creek. The estimate for the cost of the illicit discharge repair assumes that each repair will cost $50,000. The estimates for the cost of constructed wetlands, bioretention, and permeable pavement assume that 100% of the water quality volume is provided to treat 1 inch of rainfall.

To Successfully Identify Discharges, the Detection Methods Are Extremely Important and Need To Be Refined.

As indicated by the results of this study, monitoring for the right parameters is important. Many Phase I communities, in particular, do not monitor for ammonia, one of the best indicators of sewage discharges. Typical monitoring indicators for Phase I communities include pH, temperature, conductivity, chlorine, phenols, and copper. Simply adding ammonia to this list of parameters would go a long way toward identifying more discharges.

In Sligo Creek, teams consisted of at least one Center staff person and one Montgomery County staff person. Staff from each organization used their own illicit discharge monitoring protocols at each outfall (Table 7); this enabled a comparison of the protocols. Use of the Center's monitoring protocols resulted in a significant benefit in terms of the number of discharges found: 22% more additional discharges were detected in Sligo Creek using the Center's protocol compared to that used by the local government. The Center's protocol uncovered approximately 70% more discharges than would have been found using the "standard" Phase 1 set of water quality parameters (which include all of the county's parameters except detergents).

In addition, although physical indicators are important, particularly for identifying the worst discharges, one cannot always rely on physical indicators alone. In other words, actually monitoring suspect flows is a critical first step for virtually all outfalls flowing during dry weather.

Table 7. Illicit discharge monitoring parameter comparison.

CWP	Montgomery County	"Standard" Phase I Jurisdiction
Ammonia Fluoride Detergents Potassium Bacteria	Detergents pH Temperature Copper Phenols Chlorine	pH Temperature Copper Phenols Chlorine

Municipalities Can Work with the Volunteer Monitoring Community To Find These Discharges

Increasingly, citizens are interested in protecting their waterways. The volunteer monitors who worked with Center staff on this project added tremendous value in terms of watershed knowledge and enthusiasm. Although quality control issues can sometimes make it difficult to use regular instream volunteer monitoring, the use of more accessible field and laboratory techniques can be used to guide immediate management decisions. To make this work, the local government must establish good working relationships with local watershed groups so that the government agency can focus limited resources on tracking discharges and removing the source of discharges from suspect outfalls.

Using watershed group staff and/or volunteer monitors as part of the sewage discharge detection process will take training on protocols, methods, and safety, but the challenges are far from insurmountable. Given the sheer number of outfalls in urban areas, the potential breadth of the problem, and the fact that the methods would meet both the MS4 permit requirements and watershed advocacy goals, IDDE partnerships between local governments and watershed groups could go a long way toward finding and fixing sewage discharge problems.

Actually Finding the Source of Discharges Requires Effort and Persistence

The elimination of illicit discharges can be the most challenging goal, and one that needs ongoing commitment. To achieve this goal, communities need to establish an accurate storm drain network map for pipes and outfalls and continue to update it as new geographic information becomes available through monitoring and investigations. Some of the most challenging discharges to find were those from outfalls that did not exist on the stormwater map but carried a discharge. Further, one can often find a disconnect between local wastewater and stormwater agencies; the establishment of a good working relationship between these two agencies will go a long way toward elimination. Increased coordination and sharing of resources (e.g., a sewer camera) between local agencies, such as public works and wastewater utilities, would facilitate efforts to track the sources of illicit discharges.

Conclusions

Illicit sewage discharges into storm drain systems can be a major source of bacteria and nutrients entering urban waterways, despite system-wide improvements to rehabilitate the sewerage system. An investigation in Western Run in the City of Baltimore showed that the elimination of illicit discharges in this subwatershed could potentially meet 21% of the TP, 43% of the TN, and 46% of the bacteria TMDL goals. For Sligo Creek in Montgomery County, a similar analysis showed that the elimination of illicit discharges could potentially meet 6% of the TP, 17% of the TN, and 21% of the bacteria TMDL goals. Although this assessment was based on limited sampling data, the sheer magnitude of the potential load reductions is compelling, especially in light of the potential cost savings apparent from a comparison of load reductions through illicit discharge elimination versus green infrastructure practices for Sligo Creek. More research is needed, especially in estimating flow rates, to better quantify the load reduction potential from illicit sewage discharges.

Regulatory agencies should consider widespread programmatic changes to ensure that MS4 permits require the use of basic tracking tools. As a first task, agencies should develop a comprehensive geographic information system that identifies all storm drains regardless of size. This should be followed by the development of a systematic screening program that monitors all dry weather flows from storm drain outfalls for indicators of sewage, including ammonia and bacteria. Finally, the elimination of sewage discharges into the storm drain system should be the collective responsibility of MS4 permit programs as well as programs addressing SSOs. Staff resources have the potential to be high but may be offset by engaging local watershed groups in the initial screening process where feasible.

Acknowledgments

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Stream Dissolved Oxygen Improvement Feasibility Study—Salt Creek and East Branch DuPage River

Stephen McCracken[a] * and James Huff[b]

Abstract

In 2004, the Illinois Environmental Protection Agency (IEPA) developed dissolved oxygen (DO) total maximum daily loads (TMDLs) for several mainstem reaches of Salt Creek and the East Branch DuPage River in Illinois. The TMDLs recommended steep reductions in effluent concentrations of five-day carbonaceous biochemical oxygen demand and ammonia-nitrogen at the 17 wastewater treatment plants (WWTPs) that discharge into the two basins. Members of the local regulated community objected to the TMDLs, partially because of costs, but also on the grounds that the TMDLs' modeling lacked empirical data and overemphasized wastewater input contributions to the impairments. Local environmental groups also voiced skepticism about the ability of the TMDLs to improve the local aquatic environment. IEPA and the local regulated community reached an agreement that allowed local partners time to examine a number of scenarios by which to achieve compliance with the DO water quality standards. A group of local stakeholders rebuilt the models used in the original TMDLs and populated them with data from a newly implemented network of stream monitoring stations and actual WWTP loading information. The result was two calibrated and validated models that were accepted by the regulated community, local environmental groups, and IEPA. Stakeholders then used the models to project the impacts of a number of possible alternatives, including effluent loading reductions, instream aeration, and dam removal. The first wave of project implementation based on the model output is currently underway.

Introduction

In 2004, the Illinois Environmental Protection Agency (IEPA) completed dissolved oxygen (DO) total maximum daily load (TMDL) studies for several mainstem reaches of Salt Creek and East Branch DuPage River (CH2M HILL 2004a,b). To achieve the Illinois DO standards (Table 1), the TMDLs recommended further reductions in five-day carbonaceous biochemical oxygen demand ($CBOD_5$) and ammonia-nitrogen concentrations in the effluents of area wastewater treatment plants (WWTPs), based on outputs from QUAL2E models developed for each waterway. The TMDL studies noted that dam removal might abate the need for waste load reductions for oxygen-demanding pollutants, and indicated that this option could be further evaluated (CH2M HILL 2004 a,b). Dams have been observed to lower DO in their impoundments by creating conditions for excessive algae growth, decreasing re-aeration rates and increasing detention times and sediment oxygen demand (SOD) (Butts and Evans 1978).

Reactions to the TMDLs were uniformly unfavorable. WWTP operators pointed to the large costs associated with reducing wastewater loadings; the Illinois Association of Wastewater Agencies (2003) estimated compliance costs at $48 million for Salt Creek alone. They also had reservations about model accuracy. Among other things, they noted that the models used design average flow, as opposed to actual flows; the loadings were the National Pollutant Discharge Elimination System permit limits, rather than the actual discharge loadings; the data were more than seven years old; and neither model had been validated. The regulated community was skeptical that the reductions would improve aquatic biology. Environmental advocacy groups noted that the TMDL reports themselves placed low confidence in the models. According to the implementation plan of the Salt Creek report (CH2M HILL 2004b, 13), "[discharge monitoring report] data for WWTPs … show that average summer values for $CBOD_5$ and ammonia are below the proposed limits…. Thus it may be possible that these [waste load allocations] can be met with little or no additional treatment." As such, the environmental advocacy groups were also skeptical that the reductions would improve conditions for aquatic life. The WWTP community argued that this language ignored the elimination of the margin of safety needed to consistently meet recommended limits.

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Table 1. Illinois Pollution Control Board DO standards.

Measurement Interval	DO Water Quality Standard	
	August–February	March–July
At any time	3.5 mg/L	5.0 mg/L
7-day average	4.0 mg/L daily min average	6.0 mg/L daily mean
30-day average	5.5 mg/L daily mean	N/A

Source: 35 Illinois Administrative Code 302.206 (January 24, 2008).

Given the opposition, the stage was set for years of contentious implementation efforts. In 2005, IEPA came to an agreement with local stakeholders, now incorporated as the DuPage River Salt Creek Workgroup (DRSCW or Workgroup), to delay implementation of the TMDL recommendations while the DRSCW developed a plan to address DO and other impairments. Stakeholders immediately set about rebuilding the DO models. The first DRSCW project, summarized in this paper, assessed the feasibility of stream DO improvement for the East Branch DuPage River and Salt Creek. DRSCW set the following objectives for the Stream DO Improvement Feasibility Study:

- identify the principle low-flow DO sags in both waterways
- evaluate the impacts of decreasing oxygen-demanding loadings from WWTPs on the low-flow ambient DO concentrations
- evaluate the impacts of five existing dams on DO and, where significant, identify alternatives for specific dam sites (e.g., complete removal, "bridging," or some other modification that meets project goals while addressing applicable concerns)
- identify criteria and sites where stream aeration could be used to improve DO levels during low-flow conditions
- determine financial impacts, including project capital costs (e.g., for sediment removal and disposal), operation and maintenance needs, and costs associated with stream improvement projects (life cycle costs)

At all steps of this process, a diverse group of DRSCW stakeholders—representing WWTPs, municipalities, Forest Preserve Districts, and environmental groups—worked collaboratively to plan, manage, and collect data for the project. By early 2006, DRSCW had contracted with a team consisting of HDR Inc (water quality modeling), Huff & Huff Inc (water quality analysis), and Inter-Fluve Inc (stream restoration and dam evaluation) to work on the project.

Model Selection

To model DO impairments, the DRSCW chose the QUAL2K model. The fundamental utility of QUAL2E and QUAL2K is essentially the same: they are one-dimensional, steady-state models for the prediction of DO and associated water quality constituents in rivers and streams. Steady-state modeling assumes that stream conditions, such as flow, point-source discharge, and loadings, are constant in time. However, QUAL2K is capable of diurnally varying headwater and meteorological input data and includes a full sediment diagnosis model to compute SOD and nutrient fluxes between the bottom sediment and the water column. In addition, the QUAL2K model offers more options for decay functions of water quality constituents, re-aeration rate equations, heat exchange, and photosynthetically available solar radiation calculations (Chapra et al. 2005).

Given the similarities between the two models, the first step in preparing the QUAL2K model was to input data previously used in QUAL2E to produce QUAL2K outputs that could be compared to the results in the TMDL reports. The QUAL2K initial model set-up closely followed the input files from the QUAL2E model. DRSCW did not initially use the more refined features in QUAL2K, described above. DRSCW compared the QUAL2K model outputs for DO, $CBOD_5$, and ammonia-nitrogen to the QUAL2E outputs reported in CH2M HILL (2004a,b). After some manipulation of internal coefficients, QUAL2K satisfactorily reproduced the general trend of DO profiles previously generated with QUAL2E.

DRSCW modified river reach lengths in QUAL2K based on updated geographic information system (GIS) data developed as part of this project. In contrast, the QUAL2E model used US Geological Survey (USGS) river mile information. The reach lengths used in the two models differed by as much as 2.4 miles (3.9 km) in the upstream reaches of Salt Creek. DRSCW collected detailed bathymetric data from major impoundments on both rivers and adjusted the geometric files accordingly. DRSCW revised main channel slopes using the digital elevation model (DEM) developed by USGS for Salt Creek, which is publicly available in a GIS format. DRSCW extracted elevation information for the end points of each reach segment from the overlay of the DEM and reach end points set up in QUAL2K. The model proved sensitive to both geometry and SOD inputs.

DRSCW also completed sensitivity analysis for cloud cover variation. QUAL2K calculated stream velocity and depth except for impounded reaches, for which survey data were used. Changes to the stream geometry meant that reaction rate coefficients would also change. DRSCW modified CBOD, nitrification, and the settling rates of various water quality constituents using stream characteristics reported by Chapra (1997), Thomann and Mueller (1987), and the US Environmental Protection Agency (1987). Because the QUAL2K model did not simulate suspended solids in the stream or the light extinction caused by elevated suspended solids, DRSCW used a higher background light extinction rate compared to that used for QUAL2E inputs, effectively reducing the diurnal DO flux in the model.

Data Collection

A major criticism of the original DO model was its lack of quantitative data. Although data were available on streamflow, wastewater flow, and effluent quality, very limited data existed on stream quality. Gathering such information became an immediate priority. In spring 2006, DRSCW set up a system of "continuous" DO monitoring stations, which collected hourly DO, water temperature, conductivity, and pH data. The short sample interval was selected to account for the expected variability of ambient DO concentrations. The stations recorded data from May through September (warm-weather months) at six sites on Salt Creek and five on the East Branch DuPage River. The density of the sites proved critical when calibrating the model because, at various times, DO probes were inoperable or recorded data outside of quality assurance guidelines. Additionally, because QUAL2K is a steady-state model, calibration and validation required that multiple monitoring stations capture some period of steady-state ambient conditions. The continuous DO monitoring stations also supplied data with the necessary resolution to gauge compliance with the Illinois DO water quality 7-day and 30-day standards (Table 1).

Sites were selected based on stream reconnaissance carried out in early spring. DRSCW consultants identified stretches of stream where warm-weather DO sags seemed likely, including areas upstream of dams and wide, sluggish areas of river without canopy cover. The DRSCW placed the DO probes at identified monitoring stations, using casings affixed to bridges and instream mobile casings for sites where no spanning structure was available (Figure 1). Although the instream casings were more flexible in terms of placement, they also required more staff time for deployment, data retrieval, and probe maintenance. DRSCW collected all DO data according to the quality assurance project plan (QAPP) approved by IEPA. For other parameters, the probes were calibrated according to the manufacturer's recommendations and the QAPP. Continuous DO data collected on the East Branch DuPage River from 2006 and 2007 indicated that DO concentrations upstream of the Churchill Woods dam, dropped to below 2.0 mg/L and exhibited a diurnal swing of greater than 12 mg/L. This was an expected result because of the longer retention time, lack of canopy cover, higher SOD, and higher water temperatures associated with the impoundment.

Figure 1. A technician from the City of Elmhurst, an agency member of the DuPage River Salt Creek Workgroup, retrieves a data logger from an instream casing on Salt Creek.

DRSCW consultants also collected SOD data at 16 sites using the in situ method described by Murphy and Hicks (1986) concurrently with the continuous DO monitoring. The SOD survey was completed in mid-summer to minimize temperature adjustments. SOD had been entered into the QUAL2E models as a uniform assumed value. The SOD survey found that the value was in fact highly variable. Individual SOD measurements at ambient temperatures in the East Branch DuPage River ranged from a minimum of 67 g/m^2/day [1] to a maximum of 9.53 g/m^2/day. Multiple samples from each location were collected to allow for averaging across each stretch of the river. The temperature in the model runs used station-averaged 20°C SOD values, which ranged from 1.13 to 3.61 g/m^2/day.

[1] SI units are industry standard for SOD measurements.

All WWTPs in the basins cooperated in the re-modeling exercise and supplied discharge monitoring report (DMR) data to the modeling team. These data included daily values for flow, $CBOD_5$, ammonia-nitrogen, total suspended solids, and pH. DRSCW collected field coordinates for all WWTP outfalls in the two basins to ensure accurate spatial placement of the data. USGS records provided additional data on river flow.

Calibrating the Model

Unless otherwise stated, the model referred to here is the East Branch DuPage River model. DRSCW calibrated and validated the model for Salt Creek using the same methodology, except that the modeling team completed a set of additional runs when initial results proved unsatisfactory. DRSCW changed model input to simulate the period of DO data collection in August 2006. In particular, the modeling team modified the characteristics of the Churchill Woods dam impoundment based on the bathymetric survey performed in 2006. The model also used more recent streamflow, stream geometry, and actual wastewater effluent water quality and flow data as inputs. The modeling team plotted a calibration run of the model, completed for August 20, 2006, against the continuous DO measurements taken during field sampling for the same date. This comparison demonstrated excellent agreement, with the exception of the diurnal pattern at Hidden Lake (River Kilometer 23, QUAL2K output is in kilometers), which was greater than the model predicted. The modeling team repeated the calibration exercise for August 13–17, 2006, and again compared the results with observed data for that period from the continuous DO stations. These results were also satisfactory. Based on the comparison between the computed and observed results, DRSCW revised the model and completed a third model run for validation. That validation run (for the period June 19–21, 2006; Figure 2), shows the computed DO against the ambient DO concentrations observed for that period. The relative size of each green triangle shown along the top of Figure 2, representing the locations of WWTPs discharging to the East Branch DuPage River, is representative of the quantity of discharge supplied by the plant during the modeling period. (In other figures, the WWTPs are shown only as locations.)

To help identify low-flow DO sags, the modeling team had to use the calibrated and validated model to predict ambient conditions under seven-day, ten-year, low-flow (7Q10) warm-weather conditions. Historical data sets compiled by the Metropolitan Water Reclamation District

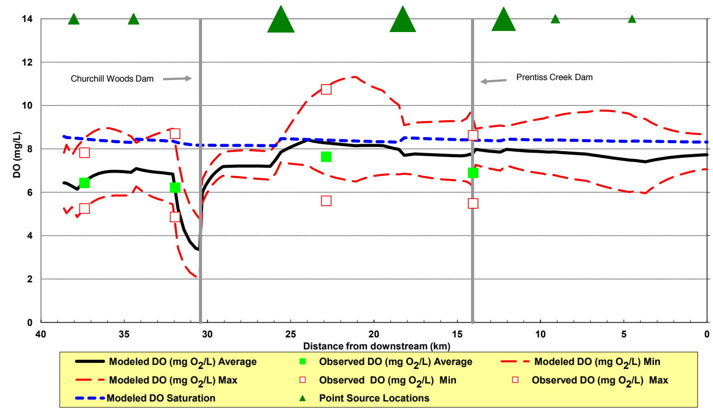

Figure 2. Observed and computed DO in the East Branch DuPage River from the QUAL2K validation run for the period June 19–21, 2006. (1 km is equivalent to ~ 0.62 miles.)

of Greater Chicago showed that, over the last 30 years, the highest recorded stream temperature was 3°C higher than the highest temperatures recorded during the validation and calibration periods.

The baseline model used the highest recorded historical temperature, the average $CBOD_5$ and ammonia-nitrogen levels discharged during summer months from WWTPs (based on DMR data from summer 2005, a period that approached the 7Q10 condition), and 7Q10 flow for the East Branch DuPage River (Singh and Ramanurthy 1993). This model run was intended to reflect worst-case conditions. The baseline output (shown graphically as Figure 3) showed that, upstream of the Churchill Woods dam, the minimum and daily mean DO levels were predicted to drop to 0 mg/L and 1.5 mg/L,

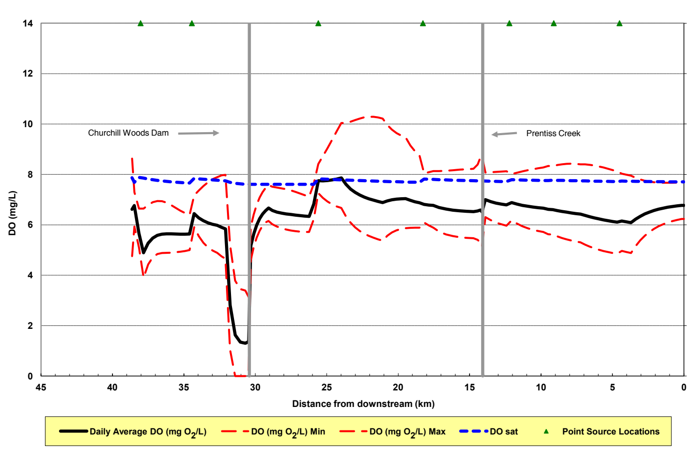

Figure 3. Computed DO for East Branch DuPage River mainstem. QUAL2K baseline model calculated using monthly average of June 2005 DMR conditions but with 3°C increased plant discharge and air temperature to simulate worst-case scenario. (1 km is equivalent to ~ 0.62 miles.)

respectively. The computed values suggested that other DO sags along the East Branch DuPage River were minor compared to the DO impact from the Churchill Woods dam.

Modeling Alternatives

The DRSCW worked with project consultants to develop, evaluate, and rank a number of aeration alternatives and to assess area dams. The group evaluated five dams according to their importance in flood control and the pros and cons of removal (ownership, sediment management, gradient at site). DRSCW removed one through-flow dam (Prentiss Creek dam) on the East Branch DuPage River from the study because modeling had not identified it as a cause of impairment and it was part of a local flood control system.

At this point, the DRSCW was ready to use the baseline model run to project the impacts of various remediation efforts on ambient DO concentrations. Initially, DRSCW evaluated riffles and various forms of instream aeration, including air and high-purity oxygen. However, the oxygen deficit above the Churchill Woods dam was so severe and the SOD so high, that only high-purity oxygen had the potential to achieve the DO water quality standard. In addition, the biological studies clearly showed a loss of aquatic biological integrity above the dam, something aeration would not ameliorate. In short, the Churchill Woods dam was clearly shown to be such a significant ecological problem that removal became the primary focus. The group selected the following alternatives for modeling on the East Branch DuPage River:

- lower WWTP loadings to zero while maintaining flow (strictly a theoretical exercise deemed necessary to demonstrate the effectiveness of such an approach)
- full removal of the Churchill Woods dam

The zero-loading model run for the East Branch DuPage River showed that, absent any pollutant loading from the WWTPs, the DO impairment would still exist at the site. The dam removal alternative model (shown in Figure 4) projected that daily average DO concentrations at the site would be in compliance following full removal, and that the higher DO levels would continue downstream. Given that the zero-loading model was projected to cost up to $67 million[2] for just the two WWTPs above the Churchill

[2] Cost estimates were based on plant design average flow, the addition of a membrane bioreactor, and the use of granular activated carbon to treat that volume of flow. Maintenance and operation costs were not included.

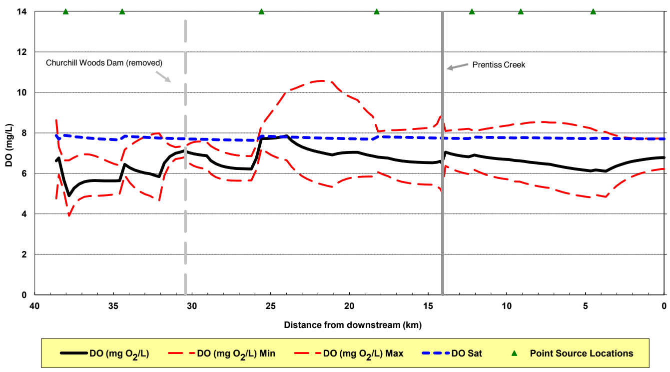

Figure 4. Modeled DO in the East Branch DuPage River. Baseline model showing removal of the Churchill Woods dam. (1 km is equivalent to ~ 0.62 miles.)

Woods dam, the DRSCW was confident that it could make a compelling case for full dam removal at that location (at a cost of $1.7 million, inclusive of engineering and permitting costs).

For the Salt Creek alternatives, modeling was more complex. The principle DO sag identified lay immediately upstream of the Fullersburg Woods dam, a local landmark. Given the nature of that site, the DRSCW devoted more resources to modeling alternatives in this waterway than on the East Branch DuPage River. The Workgroup selected the following alternatives:

- lower WWTP loadings to zero while maintaining flow (strictly a theoretical exercise)
- full removal of the Fullersburg Woods dam
- partial breach of the dam
- incremental lowering of the crest of the dam
- instream aeration with air or high-purity oxygen in the dam impoundment

As with the East Branch DuPage River, biological assessments on Salt Creek found a significant drop in aquatic biological integrity upstream of the dam. Again, modification of the dam, which served multiple purposes, became the preferred option. Cost also pointed clearly to dam removal, with estimates for upgrading the ten upstream WWTPs estimated at greater than $388 million, while dam modification costs lay in the region of $1.1 to $2.5 million.

Project Implementation

In the second half of 2008, a team consisting of the Forest Preserve District of DuPage County (the property owner), DuPage County Division of Stormwater Management, the regional stormwater authority, and the DRSCW began investigating funding options to remove the Churchill Woods dam on the East Branch DuPage River. The team hired V3 Consultants and Huff & Huff Inc in early 2009 following a number of public meetings. Engineering plans and permits for the dam removal were completed in late 2010, and the dam was removed in March 2011 (Figure 5). The project was complicated by the presence of culverts immediately downstream of the dam, which set the post-project stream floor elevation higher than that used in the QUAL2K model, eroding some of the potential DO improvements. However, the elevation of the culvert inverts also prevented the mobilization of sediments during drawdown of the impoundment, a common issue in dam removal projects. Continued monitoring at the site will confirm whether project DO goals are achieved.

The Salt Creek recommendations have not yet been implemented. The DRSCW hosted and participated in several community stakeholder meetings prior to the release of the modeling report. Many of the dam impoundment's neighbors were resistant to any modification of

Figure 5. Removal of the Churchill Woods dam gets underway.

the structure. Given the enormous cost disparity between options and the high probability of biological improvement under a dam modification scenario, partial breech and full removal remain the DRSCW's preferred options. Several dams, including Churchill Woods, have been removed in the watersheds during the last two years. The DRSCW is optimistic that data and post-project conditions at these sites will help convince community stakeholders to work for a compromise on modifying the Fullersburg Woods dam.

Conclusions

The Stream DO Improvement Feasibility Study has proven to be a very successful project. It allowed local stakeholders to organize around a joint project and build an objective decision-making process using empirical data that all parties accepted. The graphical outputs from the models made the analysis accessible to lay audiences—notably, the dam owners and those with abutting properties. All discussions emphasized the amount of empirical data involved in calibrating and validating the model. The modeling alternatives to predict the effects of reducing WWTP pollutant loading to zero clearly demonstrated that such actions were unlikely to eliminate DO violations under low-flow conditions. For both sites, modeling predicted that, compared to a WWTP loadings reduction strategy, dam removal would more effectively improve DO and would do so at lower cost. Dam removal holds the extra value of directly and beneficially impacting aquatic biology and riparian and instream habitat: pre- and post-project fish surveys of the Churchill Woods site have shown that, post-project, five species not previously found in the area have moved into the location of the former impoundment.

Acknowledgments

The DRSCW would like to thank the consultants who worked on the Stream DO Improvement Feasibility Study: HDR Inc, Huff & Huff Inc, and Inter-Fluve Inc. The DRSCW would also like to thank IEPA for its support through the 319 program and the Metropolitan Water Reclamation District of Greater Chicago for supplying data to the study.

More information on the project and on the DRSCW can be found at http://www.drscw.org/dissolvedoxygen.html.

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Integrating Stormwater Controls Designed for Channel Protection, Water Quality, and Inflow/Infiltration Mitigation in Two Pilot Watersheds To Restore a More Natural Flow Regime in Urban Streams

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Abstract

Reducing sanitary sewer overflows (SSOs) is important. But many inflow and infiltration (I/I) mitigation projects simply separate stormwater from the sanitary system and send it downstream without any treatment, causing additional channel erosion in already unstable urban streams. This is unsustainable management of water resources—in terms not only of ecological integrity, but also of public infrastructure, because unstable streams in urban settings impact adjacent sewers and roadways. In a more holistic approach to SSO mitigation, we added goals of water quality and channel protection to two otherwise routine I/I projects. Collecting fluvial geomorphic field data allowed for more accurate estimation of storage volumes required to create a less erosive flow regime in the downstream channel networks. Using continuous simulations over 57 years, we optimized stormwater controls, reducing the total duration of disturbance events and the cumulative sediment transport capacity as close to predevelopment conditions as possible, while meeting the cost criteria of the Sanitation District No. 1 of Northern Kentucky ($0.03/gallon of water treated in a typical year). These collaborative projects demonstrate the benefits of treating I/I mitigation as an opportunity, not only to renew sewer infrastructure in the project area, but also to protect downstream infrastructure from channel erosion, improve water quality by addressing both point and nonpoint source pollution, and benefit aquatic biota by restoring a more natural flow regime. In this setting, stream restoration via flow regime restoration has the potential to be more cost-effective and more beneficial to aquatic biota than approaches that rely exclusively on instream structures, which can be prone to failure in urban and suburban environments.

Introduction

Urban streams face numerous stressors, including altered flow regimes (Poff et al. 2006), physical modifications or burial (Roy et al. 2009), fragmentation (Chin and Gregory 2001), and loss of riparian area or quality (Coles et al. 2010). This degrades the richness and abundance of aquatic resources (Walsh et al. 2005). The mechanisms by which aquatic biota are impacted include chemical (toxicity), physical (habitat), and hydrologic (flow regime) pathways.

The mitigation of chemical stressors from both point sources (e.g., sanitary sewer overflows [SSOs]) and nonpoint sources (e.g., stormwater runoff) is increasing—many communities are investing hundreds of millions to billions of dollars for sewer system upgrades intended to reduce direct overflows of both combined and sanitary sewers as part of enforcement actions (e.g., see US Environmental Protection Agency [USEPA n.d.] for a complete list of enforcement cases). In some communities, these efforts have also included directives to improve the quality of stormwater runoff by, for example, installing best management practices (BMPs) or green infrastructure (GI) in addition to building sewer system capacity. Recognizing the importance of habitat to aquatic communities, some USEPA consent decrees have also included directives to conduct stream restoration projects in addition to more traditional sewer system investments. For this and other reasons, stream restoration expenditures have increased substantially during the last several decades (Bernhardt et al. 2005). But despite large investments in both water quality and habitat improvements, little postconstruction monitoring has occurred, especially in terms of aquatic biota recovery (Bernhardt et al. 2005; Bernhardt and Palmer 2007). Independent lines of evidence suggest that improved water quality and habitat may not be sufficient for preserving/restoring full ecosystem function because many native species depend on features of the natural

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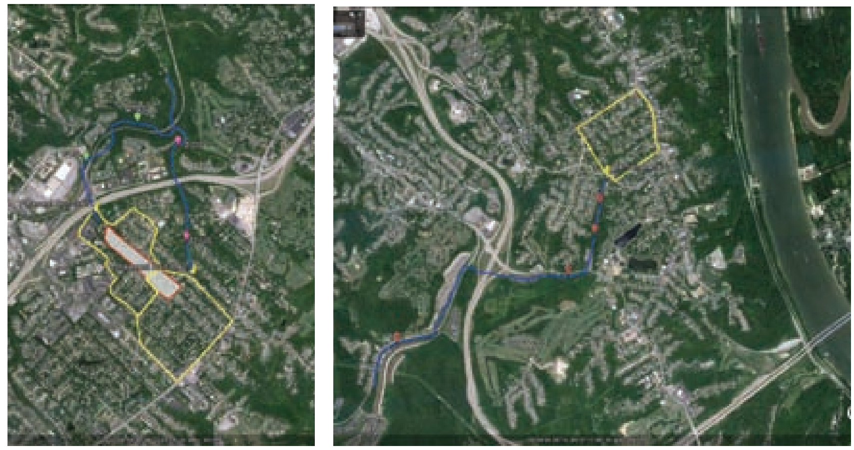
flow regime, such as the frequency and timing of disturbance events (Poff et al. 1997). Thus, a minimum level of hydrologic or watershed restoration might be necessary if functional aquatic communities are a primary goal of such investments (Palmer 2009).

Moreover, the erosive power of the urban flow regime often creates channel instabilities (Bledsoe and Watson 2001; Booth 1990; Hawley et al. forthcoming) that can impact urban infrastructure. In the three Kentucky counties of the greater metropolitan area of Cincinnati, Ohio, channel incision and bank failure have led to the closure and emergency repair of state highways and the complete replacement of main trunk sewers. This sequence, in which poor stormwater management causes channel erosion, which in turn causes damage to urban infrastructure, is highly unsustainable. Recently, the cost of replacing just one exposed sewer crossing on a small stream (~10 feet (ft)[1] [3.0 m] wide) was $100,000. Furthermore, arresting unstable channels with stream restoration that relies heavily on engineered structures, such as cross vanes, is expensive (e.g., $1.25 million for ~600 ft [182.9 m] on a recent project) and can be prone to failure in the urban or suburban setting; dozens of such structures in this area have failed within a few years of construction.



Vernon Lane *Pleasant Run*

Figure 1. Drainage areas (yellow) to project outfalls (push pins), flow paths (blue), and field sites (balloons). I/I project area on Pleasant Run (polygon with white fill, ~32 acres [12.9 ha]) was smaller than project outfall drainage areas (DA1, ~80 acres [32.4 ha]; DA2, ~192 acres [77.7 ha]). I/I project area and drainage area to outfall in Vernon Lane were essentially overlaid (~86 acres [34.8 ha] and ~96 acres [38.8 ha], respectively). North is up. Image courtesy of Google Earth.

In an effort to circumvent this trend, Sanitation District No.1 of Northern Kentucky (SD1) has conducted stream channel stability monitoring, in addition to water chemistry, habitat, hydrologic, and aquatic biota monitoring, as part of its adaptive watershed management strategy when planning and designing system improvements for its combined and separate sewer service areas. In recognition of the interdisciplinary needs of holistic watershed management, this strategy attempts to address multiple sources of pollution that affect water quality, rather than concentrate efforts exclusively on sewer system capacity and overflow reduction.

One common problem in aging sanitary sewer systems is inflow and infiltration (I/I) from nonsanitary sources, such as downspout connections and groundwater infiltration. I/I can become problematic during heavy rains when excess stormwater can overload the sanitary sewer system and cause direct overflows of untreated sanitary waste into receiving streams. Because such untreated waste is considered a human health risk and a water quality pollutant, the Clean Water Act requires that regional sewer agencies ultimately eliminate such SSOs.

This paper describes two recent pilot projects in residential sewersheds with I/I-induced SSOs (Vernon Lane, ~86 acres [34.8 ha], ~29% impervious cover; Pleasant Run, ~32 acres [12.9 ha], ~40% impervious cover; Figure 1) in which SD1 addressed water quality and channel stability design criteria in addition to I/I removal. Water quality goals included a reduction in bacterial loads from both SSOs and stormwater runoff. The channel stability goal was to create a less erosive flow regime in the receiving channels, matching both the peaks and durations of the erosive portion of the predevelopment flow regime to the extent practicable. Our expectation was that a more natural flow regime of high water quality would lead to measurable improvements in downstream aquatic communities.

A central issue in designing stormwater controls for channel protection is the fact that durations of erosive

[1] This paper primarily uses English units because of their dominant use by stormwater professionals in our study area. In some cases, however, industry standards require the use of metric/SI units

flows are typically much longer in the postdevelopment flow regime (e.g., Hawley and Bledsoe 2011), and stormwater controls focused on matching pre development flow durations tend to be more difficult to design than controls focused exclusively on peak flow matching. Even so, Santa Clara, California, requires new developments to match the entire hydrograph, such that postdevelopment flow magnitudes and durations match the predevelopment regime (Santa Clara Valley Urban Runoff Pollution Prevention Program 2004). A similar but simplified strategy in Knox County, Tennessee, uses *centroid-to-centroid* matching of the predevelopment and postdevelopment storm hydrograph for the one-year, 24-hour event (Knox County, Tennessee Department of Engineering and Public Works 2008). This approach could be achieved by controlling and releasing the predevelopment runoff volume for a given storm using primary controls to match the predevelopment hydrograph (i.e., exactly following the blue curve in Figure 2), while storing, infiltrating, and/or evapotranspirating the excess runoff volume using secondary controls. This is desirable for receiving streams because it results in the least hydrologic alteration relative to predevelopment conditions. However, the required footprint of stormwater controls—particularly in areas of poorly drained native soils, such as northern Kentucky—may make the approach difficult to achieve.

A potentially more attainable method for our region currently is erosion control detention (Bledsoe 2002; Figure 2). In this approach, stormwater controls are designed to overcompensate for the excess erosion potential of moderate- and high-frequency storms (i.e., the one- to two-year flows, which are generally considered the flows that most strongly influence channel form [Wolman and Miller 1960]), with the understanding that excess channel erosion may occur during the largest and most infrequent events. We define the flow magnitude where channel erosion begins to occur as the critical flow ($Q_{critical}$). Erosion control detention attempts to match the cumulative erosion potential of the predevelopment flow regime to the extent practicable, without necessarily matching the exact hydrograph of every storm. In other words, the cumulative channel erosion that occurs following development should be similar to the magnitude of channel erosion that would have occurred under predevelopment conditions.

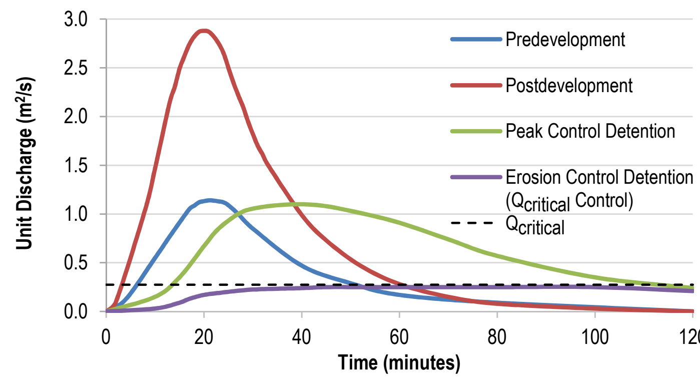


Figure 2. Example of $Q_{critical}$ control erosion control detention in Fort Collins, Colorado, for the two-year, two-hour event (adapted from Bledsoe [2002]), where the two-year storm is overcontrolled such that the cumulative erosion potential of all postdevelopment events more closely matches the predevelopment erosion potential. Peak control detention is defined as detention that is designed to match the predevelopment peak flow magnitude with prolonged duration.

This study explored the potential use of $Q_{critical}$ control as a means to restore more natural flow and disturbance regimes in two receiving streams with channel instabilities largely attributable to upstream urbanization. With limited space in two built-out watersheds, $Q_{critical}$ control focuses on mitigating the erosive portion of the urban flow regime, acknowledging that full hydrologic restoration would probably be cost prohibitive in this case.

Methods

This study used both monitoring and modeling data to evaluate the effectiveness of stormwater controls for reducing downstream erosion impacts in the two pilot project areas, while also improving the biological and water quality condition of the streams. We present a description of geomorphic and biological assessments, along with hydraulic and hydrologic analyses of pre- and postdevelopment flow regimes.

Field Data Collection

This paper evaluated four sites within each project drainage area for biological and geomorphic conditions (Figure 1). Because the I/I project area in Pleasant Run drained to two separate basins (drainage area [DA]1 and DA2 in Figure 1), we divided field sites evenly among the two downstream reaches. We collected preconstruction biological and habitat data according to USEPA rapid bioassessment protocols (Barbour et al. 1999), with regional adaptations by the

Kentucky Division of Water (KDOW 2008). We assessed biological communities using the Kentucky macroinvertebrate biotic index (MBI; Pond et. al 2003).

We conducted fluvial geomorphic field assessments over several stream reaches on project receiving streams to assess channel stability and select suitable sites for data collection. Selected sites were (1) representative of the respective reach and (2) removed from the potential influence of fluvial constrictions, backwater, and channel hardpoints to the extent possible. The latter point was of particular importance because Hawley et al. (forthcoming) documented an increasing risk of channel incision moving upstream from artificial grade control and natural bedrock. In each pilot watershed, we collected cross-section, profile, and bed material data at four sites according to Harrelson et al. (1994) and Bunte and Abt (2001a; 2001b).

Estimating $Q_{critical}$

We estimated $Q_{critical}$ for the median bed material particle size (d_{50}) at each site using the dimensionless shear stress and Manning's equations. We estimated Manning 's *n* using the Cowan method (Chow 1959) and the Shields parameter ($\boldsymbol{\tau}_{*c}$) per Julien (1998). Because both empirical parameters have considerable variability, and limited literature is available on the Shields parameter for embedded clasts of broken limestone bedrock, we populated a range of probable values for both Manning's *n* (e.g., 0.048–0.132) and the Shields parameter (e.g., 0.03–0.54). This produced a range of $Q_{critical}$ estimates, which we summarized by their means and associated 95% confidence intervals.

Estimating Q_2 and Scaling to Project Outfalls

Although we developed detailed hydrologic models of the sewersheds for each project, budgetary constraints did not allow for the extension of those models to the downstream channel locations, except in DA2 of Pleasant Run, where the design site (DA2-upstream [US]) was relatively close to the project outfall. Therefore, for cross-comparison and to enable scaling of $Q_{critical}$ estimates from field sites to project outfalls, we expressed the $Q_{critical}$ estimates as functions of the predevelopment two-year instantaneous peak flow (Q_2) after Watson et al. (1997), using the US Geological Survey (USGS) regional regression equation, which was developed using gage sites with drainage areas as small as ~100 acres (40.5 ha; Hodgkins and Martin 2003):

$$Q_2 = 312 \times DA^{0.673} \qquad \textit{(Eq. 1)}$$

where Q_2 = predevelopment instantaneous peak flow with a recurrence interval of two years, in cubic feet per second (cfs), and *DA* = contributing drainage area in square miles (mi²).

Sediment Transport Modeling

Hydraulic modeling is a prerequisite to sediment transport modeling because sediment transport equations ultimately depend on hydraulic properties, such as depth, hydraulic radius, and cross-sectional area. Assuming normal depth, we used the Manning's equation to model reach hydraulics, with site-specific hydraulic–geometry relationships after Buhman et al. (2002). We modeled the stream's sediment transport capacity using the Meyer-Peter and Müller (1948) equation as presented by Julien (1998), with corrected parameters from Wong and Parker (2006):

$$q_{bv} = 3.97 \times (\boldsymbol{\tau}_* - \boldsymbol{\tau}_{*c})^{1.6} \times \{(G-1)gd_s^{\ 3}\}^{0.5} \qquad \textit{(Eq. 2)}$$

where q_{bv} = unit bedload discharge by volume (m^2/s), which must then be integrated over the top width for the respective flow to determine volumetric bedload (m^3/s); $\boldsymbol{\tau}_*$ = dimensionless shear stress, approximated for gradually varied flow as $\boldsymbol{\tau}_* = RS_f/\{(G-1) \times d_s\}$, where R = hydraulic radius and S_f is approximated by the bed slope; $\boldsymbol{\tau}_{*c}$ = Shields parameter; G = specific gravity of sediment (2.65); g = acceleration of gravity (9.81 m/s^2); and d_s = sediment particle diameter, d_{50} in this application. The equation is presented in SI form for consistency with the referenced presentation in Julien (1998).

Modeling Storm Sewer Hydrology

We developed independent storm sewer models for the Pleasant Run and Vernon Lane project areas using the Storm Water Management Model and Infoworks, respectively, from a combination of field survey, geographic information system data, and connectivity data. We calibrated base models of the existing systems with flow monitoring data, collected over several months, from multiple locations within the respective sewersheds. We then modified these base models to reflect predevelopment and proposed condition scenarios. We took an additional step on the Pleasant Run project to calibrate the predevelopment model to expected peak flows using the rational method. We ran long-term (1950–2007) continuous simulations based on hourly rainfall data from the Covington, Kentucky, airport gage (see National Oceanic and Atmospheric Administration n.d.). Because the time of runoff concentration can be less than one hour on small watersheds, we disaggregated the rain data into five-minute increments for the Pleasant Run model after Ormsbee (1989).

Water Quality Design Parameters

In addition to reducing direct SSOs, a central goal in SD1's watershed plans is to achieve a reduction of at least 50% in nonpoint source bacterial loadings from the first 0.8 inches of stormwater runoff. Moreover, SD1 attempts to achieve these reductions as close to the source as possible. In both pilot watersheds, essentially no stormwater treatment or detention existed in the project areas prior to these projects.

Alternatives Evaluation

Using the detailed hydrologic models, we developed design alternatives to minimize $Q_{critical}$ exceedances and match the sediment transport capacity of predevelopment conditions to the extent practicable, while also meeting the point and nonpoint source water quality treatment goals. The design alternatives included above ground and below ground multi-stage detention and retention options to reduce erosive flows, coupled with GI to prolong network travel time and reduce nonpoint source bacteria concentrations. GI included down-spout disconnections, curb and walk filter media, curbside or backyard bio-swales and infiltration trenches, pervious pavement, and underground storage in streets. We developed estimates of probable construction cost independently for each project based on regional construction costs.

Results

Channel Condition

The receiving streams on both projects had varying degrees of instability. Similar to the findings of Hawley et al. (forthcoming), reaches immediately upstream of hardpoints, such as intact bedrock or exposed pipe crossings, were relatively stable, whereas reaches that lacked the protective capacity of channel hardpoints showed greater instability (Figure 3). This is evident in their cross-sectional forms (Figure 4), where Vernon (VRN)-D, DA2-downstream (DS), and DA1-DS were the farthest removed from hardpoints and tended to have the highest and steepest banks. In contrast, the erosional impacts at VRN-C were minimal because of the protective effects of an exposed pipe crossing (e.g., a hardpoint) at a relatively short distance downstream. The grade-controlling effects of the exposed pipe crossing were also evident at VRN-C by its finer bed material gradation compared to other sites. For example, Figure 5 shows that 50% of the particles were smaller than 30 mm at VRN-C, whereas only ~20% of the particles at the other Vernon sites were

Figure 3. Looking upstream at the DA2-DS site in Pleasant Run (note failure of left bank).

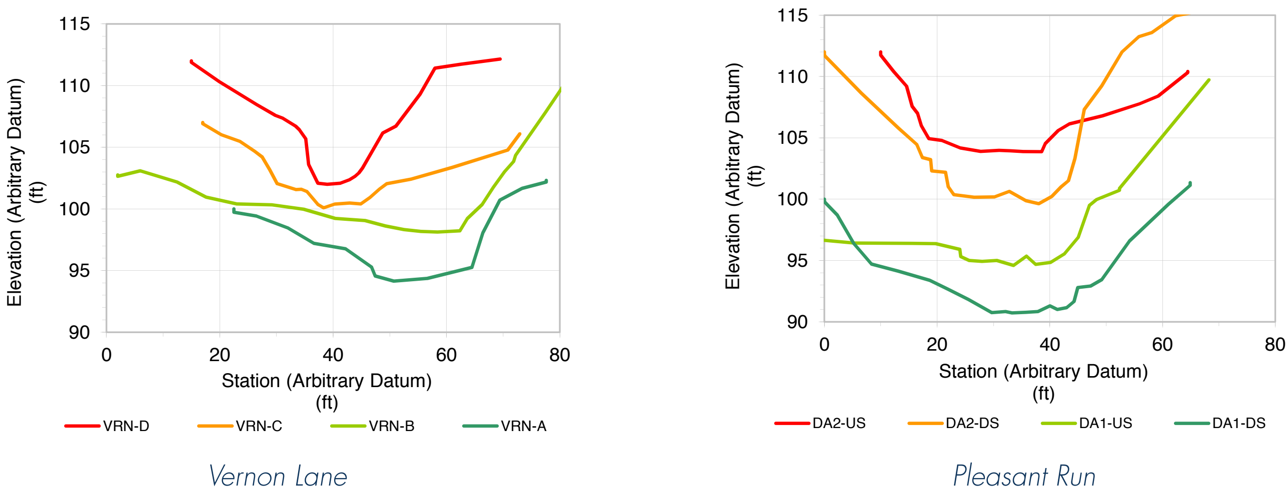


Figure 4. Superimposed cross-sections from representative sites (facing downstream, ~3.28 ft = 1 m).

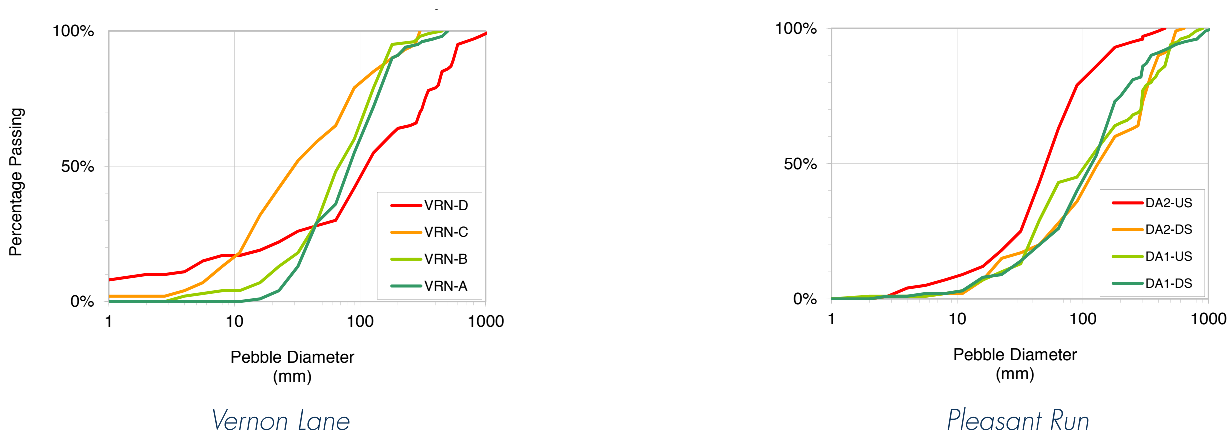


Figure 5. Superimposed bed material pebble counts from representative sites.

Table 1. Select properties of field sites and mean estimates of $Q_{critical}$.

Site	Drainage Area (mi²)	Q_2 (cfs)	Slope (%)	d_{50} (mm)	Bankfull Width (ft)	Bankfull Depth (ft)	Critical Depth (ft)	Critical Depth (%BF)	Mean $Q_{critical}$ (cfs)	Mean $Q_{critical}$ (%Q_2)
VRN-D	0.25	122	2.36	113	14	4.17	2.02	48	55.3	54
VRN-C	0.29	136	1.66	30[a]	19	1.96	0.85	43	6.1	5
VRN-B	0.57	214	1.51	68	31	1.86	1.80	97	93.9	50
VRN-A	1.64[b]	435	1.90	83	23	2.63	3.28	125	63.5	14
DA2-US	0.30	139	1.37	52	24	1.73	1.30	75	48.0	35
DA2-DS	0.54	206	3.98[c]	133	22	2.56	1.40	55	43.3	21
DA1-US	0.67	239	1.20	109	19	1.31	4.27[d]	326	793.6	332
DA1-DS	0.79	267	2.71	119	24	2.08	1.61	77	86.0	32

Note: BF, bankfull, ~2.6 km² = 1 mi²; ~35 cfs = 1 m³/s.
[a] Bed material composition at VRN-C was influenced by a proximate downstream hardpoint (unavoidable in this reach), which induced deposition and caused the bed material to become finer.
[b] VRN-A was less transferable to the project because of the large differences in drainage areas (1.6 mi² vs. 0.15 mi² project area).
[c] Slope at DA2-DS was possibly over-steepened as a result of active headcutting, despite several attempts to install artificial grade control using cross vanes that were undergoing failure via headcutting and flanking.
[d] Critical depth at site DA1-US was influenced by an atypically wide (40-ft) and flat terrace accessed at a depth of only 1.3 ft.

smaller than 30 mm; this indicates that the flatter bed slope upstream of the pipe crossing had induced sediment deposition at VRN-C. Table 1 summarizes select metrics.

Preconstruction Habitat and Biological Conditions

Biological conditions of each stream, based on habitat assessments (Table 2) and macroinvertebrate communities (Table 3), indicated generally degraded conditions in receiving streams of both project areas. Habitat was designated, after KDOW (2008), as "nonsupporting" of aquatic life at all sites. Macroinvertebrate communities, again after KDOW (2008), were designated as "poor" at VRN-A, VRN-B, and VRN-D, and VRN-C was "very poor;" in the Pleasant Run project area, site DA2-DS was rated as "poor," and all three remaining sites were "very poor."

Table 2. Habitat assessment scores.

Site	ES	EMB	VDR	SD	CFS	CA	FOR	BS		VP		RZW		Score	Classification
								Left	Right	Left	Right	Left	Right		
VRN-D	11	7	12	9	12	9	15	5	3	5	3	7	2	100	Nonsupport
VRN-C	9	9	9	6	8	8	16	7	5	6	3	9	2	97	Nonsupport
VRN-B	13	13	10	7	10	11	13	6	4	6	3	6	1	103	Nonsupport
VRN-A	17	12	12	11	15	13	7	4	3	4	3	1	1	103	Nonsupport
DA2-US	10	10	9	8	12	11	14	6	7	6	6	4	3	106	Nonsupport
DA2-DS	10	16	11	5	9	13	16	5	3	9	5	8	2	112	Nonsupport
DA1-US	10	8	8	3	7	12	13	7	2	4	4	8	2	88	Nonsupport
DA1-DS	10	7	9	7	10	15	15	7	5	7	5	8	5	110	Nonsupport

Notes: ES, epifaunal substrate; EMB, embeddedness; VDR, velocity/depth regime; SD, sediment deposition; CFS, channel flow status; CA, channel alteration; FOR, frequency of riffles; BS, bank stability; VP, vegetative protection; RZW, riparian zone width.

Table 3. Kentucky macroinvertebrate metric and index scores.

Site	G-TR	G-EPT	mHBI	%Ephem*	m%EPT	%C+O	%CLING[a]	MBI	Classification
VRN-D	6	0	7.48	N/A	0	12.5	0.24	18.69	Poor
VRN-C	10	0	7.88	N/A	0	22.5	0.5	17.54	Very Poor
VRN-B	18	2	7.58	N/A	0.5	16.9	3.2	22.34	Poor
VRN-A	15	2	7.71	1	1	8	2.5	22.54	Poor
DA2-US	14	0	7.97	N/A	0	82	3	10.25	Very Poor
DA2-DS	11	2	7.82	N/A	1.2	23.3	0.6	18.91	Poor
DA1-US	9	1	7.08	N/A	0	94.7	2.6	9.23	Very Poor
DA1-DS	15	1	6.38	N/A	4.81	88.7	6.5	14.35	Very Poor

Notes: G-TR, genus-level taxa richness; G-EPT, genus-level Ephemeroptera, Plecoptera, and Trichoptera taxa richness; mHBI, modified Hilsenhoff biotic index; %Ephem, relative abundance of mayflies, only used in headwater stream assessments; m%EPT, relative abundance of EPT individuals, minus the genus Cheumatopsyche; %C+O, relative abundance of Chironomidae and Oligochaeta; %CLING, relative abundance of clingers.*
[a] Note the particularly low abundance of clingers, a habitat type that is indicative of the relative stability of the channel.

Estimates of $Q_{critical}$

Based on a range of probable estimates for the empirical parameters of Manning's *n* and the Shields parameter, we produced a range of $Q_{critical}$ estimates with the mean values shown in Table 1. Because each site had different contributing drainage areas, we expressed each $Q_{critical}$ estimate as a percentage of Q_2 for greater comparability among estimates (see Table 1, far right column). As discussed above, VRN-A, VRN-C, DA1-US, and DA2-DS were all influenced by factors that could artificially bias the $Q_{critical}$ estimate (see Table 1, notes). As such, VRN-D and VRN-B were most representative for design on the Vernon Lane project, with mean estimates of ~50% of Q_2. That is, the $Q_{critical}$ values corresponded to approximately half of the predevelopment, two-year peak flow magnitude (Q_2). In Pleasant Run, sites DA1-DS and DA2-US were most representative, with mean estimates of ~30% of Q_2.

We then scaled the respective $Q_{critical}$ values of ~40 cfs (1.13 m^3/s) at VRN-D, ~66 cfs (1.87 m^3/s) at DA1-DS, and ~37 cfs (1.05 m^3/s) at DA2-US upstream to the respective project outfalls to develop project design values using the USGS regional equation for Q_2 (Hodgkins and Martin 2003) after Watson et al. (1997; Table 4).

Table 4. Design $Q_{critical}$ values scaled to project outfalls via $(DA_{project}/DA_{stream})^{0.67}$.

Stream Site	Stream Drainage Area (mi^2)	Stream Design $Q_{critical}$ (cfs)	Project Drainage Area (mi^2)	Project Design $Q_{critical}$ (cfs)
VRN-D	0.25	40	0.15	28
DA1-DS	0.79	66	0.13	20
DA2-US	0.30	37	0.30[a]	37

Note: 1 mi^2 ≈ 2.6 km^2; 1 m^3/s ≈ 35 cfs.
[a] Because of the close proximity of DA2-US to the Pleasant Run project outfall, the detailed hydrologic model was extended downstream to encompass the entire drainage area of DA2-US, requiring no flow scaling in this case.

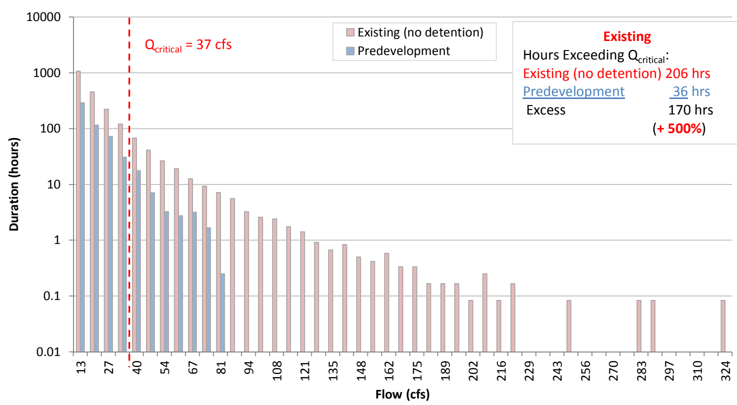


Figure 6. Magnitude and duration of $Q_{critical}$ exceedances under existing and predevelopment conditions in DA2 of Pleasant Run over 57 years of rainfall, ~35 cfs = 1 m^3/s.

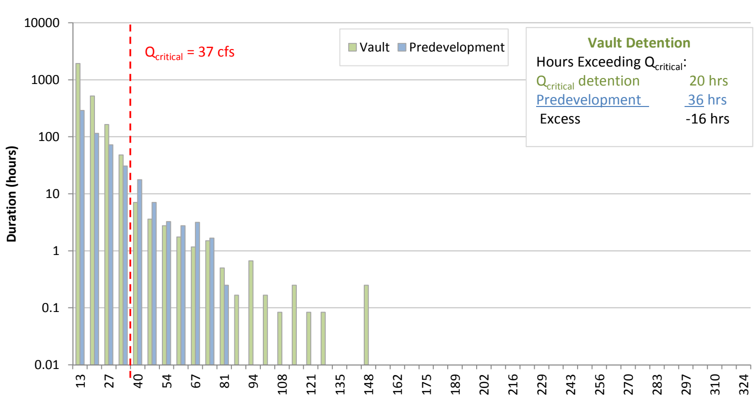


Figure 7. Magnitude and duration of $Q_{critical}$ exceedances under vault detention and predevelopment conditions in DA2 of Pleasant Run over 57 years of rainfall, ~35 cfs = 1 m^3/s.

Hydrologic Simulations

We modeled 57-year simulations of predevelopment, existing (postdevelopment with no flow control), and several proposed stormwater control scenarios to determine their performance in minimizing cumulative $Q_{critical}$ exceedances (Table 5). Despite differences in modeling platforms and rainfall resolution, both projects showed substantial imbalances between existing and predevelopment conditions. DA2 of Pleasant Run (~40% imperviousness) had 206 hours of $Q_{critical}$ exceedances under existing conditions compared to 36 hours under predevelopment conditions, for an excess of 170 hours, or 500% (Figure 6). In DA1 of Pleasant Run (~40% imperviousness), the values for existing and predevelopment conditions were 275 hours and 25 hours, respectively, for an excess of 250 hours. In Vernon Lane (~29% imperviousness), the values were 95 hours compared to 0 hours, for an excess of 95 hours.

Given the magnitude of the existing hydrologic alteration, it seemed impractical, in some cases, to control stormwater to predevelopment conditions (i.e., by installing controls such that, above 37 cfs (1.05 m^3/s), the red bars would match the blue bars in Figure 6). However, the purpose of this exercise was to see what level of control (and associated costs) would be required to achieve a more natural flow regime. Because of the heavily urban nature of the project areas, large footprints were not readily available to fit more cost-effective detention structures. For example, in DA2 of Pleasant Run, a traditional detention basin augmented with subsurface vaults was required to nearly match predevelopment flow conditions (Figure 7; note that the green bars come much closer to matching the blue bars above 37 cfs). But perhaps an equally valuable consideration when assessing the performance of various design scenarios is the improvement relative to existing conditions, especially given that these channels have been adjusting to altered flow regimes for more than 50 years. For example, even the smallest detention alternative in DA2 of Pleasant Run (i.e., graded detention in Table 5) reduces the duration of $Q_{critical}$ exceedances by more than 60% (or 75 hours) relative to existing conditions (206 hours).

Table 5. $Q_{critical}$ exceedances, cumulative sediment transport capacity, and estimated costs of competing design scenarios for DA2 in Pleasant Run, modeled over 57 years of rainfall.

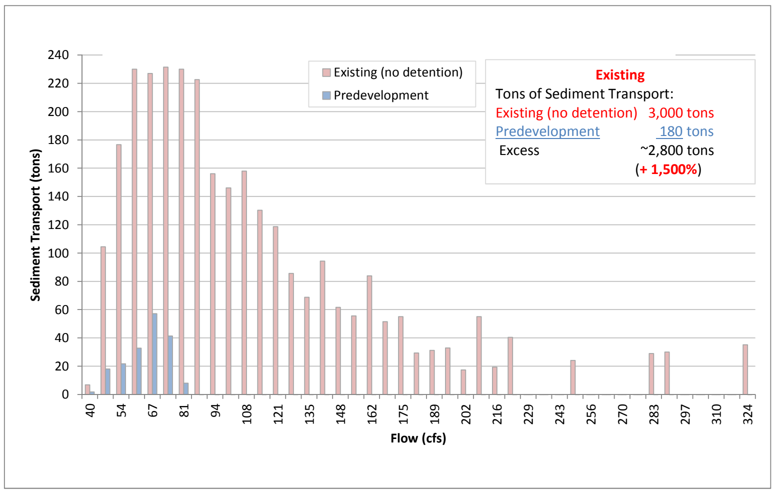
Model Scenario[a]		Storage Volume	Cost		$Q_{critical}$ Exceedance		Sediment Transport	
Name	Description	(Thousands of ft³)	Total Cost ($k)	Mean Annual Cost per Gallon Stored[b] ($/gal)	Duration (hours)	Relative to Predvlp.	Total (tons)	Relative to Predvlp.
Predevelopment	Predevelopment conditions		—	—	36	—	180	—
Existing	Existing conditions (no detention)		—	—	206	+500%	3,000	+1,500%
Graded Detention	Detention basin with graded side slopes	49	140	0.002	75	+100%	1,400	+650%
Graded Detention with Inline Basin	Graded basin with downstream inline basin	79	170	0.002	53	+50%	930	+400%
Wall and Graded Detention	Graded basin augmented with retaining wall	95	200	0.003	40	+10%	660	+265%
Wall Detention with Inline Basin	Retaining wall basin with downstream inline basin	125	230	0.003	30	−15%	450	+150%
Vault Detention	Detention basin with subsurface vaults	292	2,000	0.030	20	−45%	240	+33%

Note: ~35 ft³ = 1 m³; ~0.264 gallons (gal) = 1 L; ~1.1 ton = 1 metric ton.
[a] Implicit in each design are water quality features (e.g., bio-infiltration) to achieve the water quality criteria for nonpoint source pollution of removing 50% of bacterial loads from runoff induced by the first 0.8 inches (~2 cm) of precipitation.
[b] Mean annual cost per gal stored during a typical year of precipitation (i.e., 1970 rainfall record).

Cumulative Sediment Transport

Evaluating design alternatives based exclusively on durations of $Q_{critical}$ exceedances can mask potentially disproportionate increases in erosive power at the highest flow events. For example, the 15 minutes of flows at 148 cfs (4.19 m^3/s) in the vault design (Figure 7) could do nearly four times the damage of 15 minutes of flows at 81 cfs (2.3 m^3/s) under the predevelopment scenario. Indeed, sediment transport modeling showed that the flows at 148 cfs (4.2 m^3/s) could transport 31 tons (28.1 metric tons) of sediment, whereas the same 15 minutes at 81 cfs (2.3 m^3/s) could transport only about 8 tons (7.3 metric tons) of sediment. Designing controls to match the cumulative sediment transport capacity of predevelopment conditions may be more appropriate than matching only the duration of $Q_{critical}$ exceedances because it may be a better surrogate for channel stability, and would more effectively match the natural habitat disturbance regime of the predevelopment setting.

For example, when integrating over the 57-year simulation, Table 5 indicates that the wall and graded detention alternative in DA2 comes within 10% of matching the total number of hours of $Q_{critical}$ exceedances in predevelopment conditions. However, it still has the potential to transport 265% more sediment than in predevelopment conditions. Although the design is a vast improvement over existing conditions (in which sediment transport capacity is 1,500% more than in predevelopment conditions; Figure 8a), it exemplifies the importance of considering cumulative sediment transport in addition to $Q_{critical}$ exceedances. (See Figure 8b for the vault detention alternative.)



(a) Existing and Predevelopment

(b) Vault Detention and Predevelopment

Figure 8. Cumulative sediment transport capacity in DA2 of Pleasant Run over 57 years of rainfall.

Cost–Benefit Analysis

Based on previous evaluations of alternative approaches for meeting its water quality goal for nonpoint bacterial pollution (50% reduction from the first 0.8 inches [~ 2 cm] of precipitation) in its separate sewer service area, SD1 has a watershed planning goal of keeping the capital costs of stormwater controls associated with both peak flow reduction and water quality improvement below \$0.03/gallon of runoff[2] treated per typical year (compared to \$0.50/gallon in the combined sewer service area). We had limited cost criteria data from other communities; however, our water quality alternatives evaluation identified this target of \$0.03/gallon treated as the *knee of the curve*, in that unit costs of associated BMPs increased at much faster rates above the \$0.03/gallon value, whereas BMPs below the \$0.03/gallon value tended to have similar cost-effectiveness in the separate sewer service area. We estimated the cost-effectiveness by running a continuous simulation of the typical-year rainfall (i.e., 1970), and determining how many total gallons would be effectively routed through stormwater controls. All design scenarios on all projects achieved the \$0.03/gallon criterion; however, the projects had considerable variability because of site constraints. For example, a graded basin augmented with a retaining wall could effectively match

[2]~0.264 gallons = 1 liter.

predevelopment $Q_{critical}$ exceedances for $200,000 ($0.003/gallon), but it would take a $2 million ($0.030/gallon) basin with subsurface vaults to come within 33% of matching the predevelopment sediment transport capacity (Figure 9).

Discussion

The project areas were developed primarily in the 1950s and 1960s with no stormwater detention. This led to large increases in the magnitudes and durations of erosive flows and much higher sediment transport capacity, causing severe instabilities in receiving stream reaches that lack the protective capacity of grade control. System-wide instability was so severe that several reaches with recently installed cross vane grade-control structures were already being undermined by headcutting and/or flanking at the start of this project.

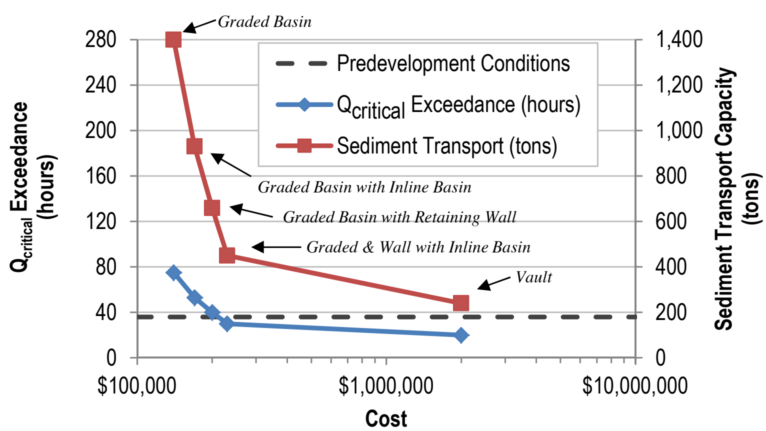


Figure 9. Performance vs. cost of detention alternatives in DA2 of Pleasant Run.

As a part of its I/I mitigation projects, SD1 looked for opportunities to install stormwater controls that could help arrest the downstream channel instability by restoring a less erosive flow regime. The storage requirements for detention that could result in a predevelopment-like sediment transport regime were relatively large (e.g., ca. 300,000 ft³ [8,495 m³] in DA2 of Pleasant Run), and SD1 found very few opportunities to retroactively fit controls of such scale. We considered an array of distributed and centralized controls, such as pervious pavement, swales, and underground storage, but multistage detention was typically the only control that could store the required volume at SD1's cost criterion of $0.03/gallon. DA2 of Pleasant Run and Vernon Lane included just enough open space for surface detention that could be optimized for $Q_{critical}$ control and augmented with bio-infiltration to meet our water quality design criteria for nonpoint source pollution (removal of 50% of bacterial loads from runoff induced by the first 0.8 inches of precipitation).

Because DA1 of Pleasant Run included no open space, the only locations that could hold the required volume of ca. 250,000 ft³ (7,079.2 m³) were in open-channel sections. We were uncertain how the potential benefits for downstream water quality, habitat, and channel stability would be received by the permitting authorities at the US Army Corps of Engineers and KDOW, given their general resistance to inline storage basins. A request for the consideration of inline storage seemed to be warranted in this case because of the heavily degraded and intermittent state of these few hundred feet of channels that were not otherwise buried during the original construction in the 1950s relative to the potential system-wide benefits. If this aspect of the project is not permitted, our sites downstream of DA1 will serve as controls relative to DA2 and Vernon Lane, where designs are less dependent on permitting considerations.

Beyond the $0.03/gallon criterion, we also considered the relative cost-effectiveness of the various basin designs. For example, adding subsurface vaults in DA2 would bring the sediment transport regime to within 33% of predevelopment conditions, but the costs were an order of magnitude higher than the next best alternative that controlled to within 150% of the predevelopment regime. Given that the existing conditions were 1,500% more erosive than the predevelopment regime, the knee of the cost curve ($220,000) in Figure 9 seemed to be a reasonable selection.

Conclusions

Numerous studies have demonstrated that watershed urbanization directly alters the quality, habitat, and stability of receiving streams, a finding further supported by our study. However, by attempting to mitigate these impacts as a part of I/I mitigation projects, our approach may be novel. Sanitary sewer systems and stormwater quantity and quality have traditionally been approached as separate design problems requiring different engineering teams. But we did more than simply consider design criteria from all three fields—our stormwater controls are actually calibrated to their respective receiving streams. By collecting fluvial geomorphic data, we were able to more accurately estimate how much volume we needed to control to promote downstream channel stability. And rather than engineering the stream channel with expensive grade-control structures, we are promoting the more holistic restoration of the fluvial geomorphic process by designing to a flow regime that better matches the natural disturbance regime and is of high water quality.

In future work, we expect to quantify improvements in channel stability and macroinvertebrate communities with our planned postconstruction monitoring and will revisit the metrics summarized in Tables 2 and 3.

Acknowledgments

We thank Dr. Brian Bledsoe (Colorado State University), who pioneered the erosion control detention (i.e., $Q_{critical}$) approach. We are grateful to SD1 staff, including Sean Blake, Samantha Brown, Jim Gibson, and the field crews. Andrew Dodson (Knox County, Tennessee) shared insights on Knox County's centroid-to-centroid approach. Kelly Kuhbander, John Lyons, and Chris Rust (Strand Associates); Eric Strecker (Geosyntec); Kristen Benick (AECOM); and Rob Keller (Bayer Becker) have all contributed to this or related work on regional channel stability. Finally, we thank Dr. Neely Law and several anonymous reviewers for their excellent reviews and suggestions, which strengthened the manuscript.

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The State of the San Gabriel River Watershed:
Using Multiple Indicators To Assess Watershed Health

Kristy Morris[a] and Scott Johnson[b]*

Abstract

The San Gabriel River Regional Monitoring Program (SGRRMP), developed by a stakeholder workgroup to provide a multilevel monitoring framework combining probabilistic and targeted sampling of watershed-scale water quality, toxicity, bioassessment, and physical habitat condition, was the first such monitoring effort in California. To assess the condition of streams in the watershed, SGRRMP sampled 69 unique sites from 2005 through 2009 using multiple lines of evidence, including indictors for aquatic chemistry, toxicity, bioassessment, and physical habitat conditions. Results demonstrated that stream conditions, particularly water quality and physical habitat, were less degraded in the upper (undeveloped) portion of the watershed compared to the lower, developed watershed, which includes the concrete-lined mainstem. To assess whether conditions at sites of unique interest are getting better or worse, SGRRMP annually monitored eight sites upstream of confluence points in the upper and lower watershed to assess temporal trends. After five years of monitoring, it has not been possible to discern temporal trends in aquatic chemistry, toxicity, and physical habitat conditions. Index of biological integrity scores were consistently above the impairment threshold for confluence sites in the upper watershed and below reference conditions in the lower watershed. Results from SGRRMP are directly comparable to regional and statewide programs and have led to several collaborative special studies. SGRRMP has successfully shown that a combination of probabilistic and targeted sampling can address watershed-scale management questions and can provide a context for answering essential management questions on a watershed, regional, and statewide scale.

Introduction

To assess the condition of surface waters in their respective regions, many local, regional, and state government agencies have developed ambient water quality monitoring programs. Data from such programs are valuable for answering questions specific to particular watersheds. However, these programs do not enable comparisons among watersheds or data sharing across agencies because they do not share a common monitoring design framework and they lack procedural, geographic, and temporal coordination.

Monitoring in the San Gabriel River watershed prior to 2005 was largely uncoordinated, with numerous agencies independently collecting data from defined portions of the watershed—mostly around major discharges for permit compliance purposes—while much of the watershed was left unmonitored. The large inconsistencies among programs in relation to the constituents sampled and the frequency of measurement resulted in limited data comparability, redundancies among monitoring programs, and major data gaps. Realization of these deficiencies led to the development of a coordinated watershed monitoring program that integrates permit-mandated and ambient monitoring.

The San Gabriel River Regional Monitoring Program (SGRRMP), developed by a multistakeholder workgroup in 2004 to provide a framework for watershed-scale monitoring, is the first such program in California. It provides coordinated, multilevel, watershed-wide monitoring by expanding the monitoring of ambient conditions, improving coordination and cost-effectiveness of disparate monitoring efforts, and providing a framework for periodic and comprehensive assessments of watershed conditions.

The development of the monitoring design brought together watershed stakeholders consisting of representatives from state and federal water regulatory agencies, key permittees in the watershed, other resource management agencies, nonprofit organizations, and citizen monitoring groups. In the first steps in SGRRMP's development, the workgroup created a list of core monitoring questions and assessed the ability of current monitoring efforts in the watershed to answer these questions. The workgroup then recommended monitoring designs to effectively and efficiently answer these questions and achieve multiple objectives. The resulting program is a multilevel monitoring framework that combines probabilistic and targeted sampling for water quality, toxicity, bioassessment, and physical habitat condition (Figure 1).

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Watershed Monitoring Program Approach

Bring together watershed stakeholders

↓

Compile an inventory of existing effort

↓

Develop list of monitoring questions

↓

Assess current ability to answer questions

↓

Design monitoring program

Figure 1. Approach for developing the San Gabriel River Regional Monitoring Program.

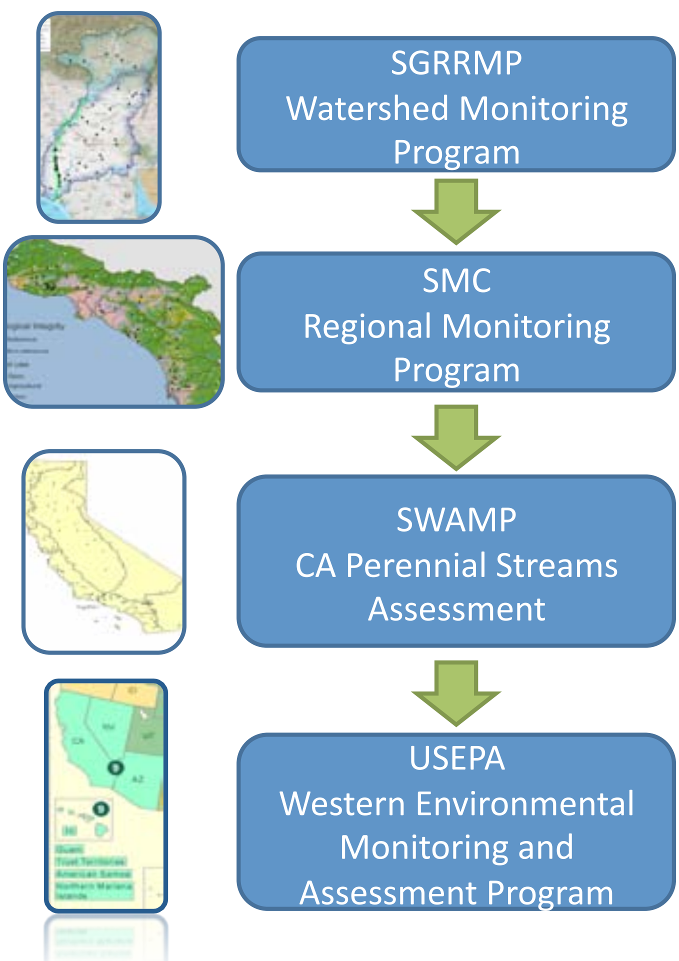

Figure 2. Integration of watershed monitoring programs.

The overall program design addresses each of the following five key management questions:

1. What is the condition of streams in the watershed?
2. Are conditions at areas of unique interest getting better or worse?
3. Are receiving waters near discharges meeting water quality objectives?
4. Is it safe to swim?
5. Are locally caught fish safe to eat?

These questions provide the rationale for the design approach, selection of monitoring indicators, sampling frequencies, and appropriate data products. The monitoring is focused on collecting data that help managers make scientifically informed decisions. The monitoring design is also intended to be adaptive, in terms of its ability both to initiate follow-up studies as needed and to make necessary changes based on monitoring findings.

Finally, SGRRMP was developed to complement, coordinate, and integrate with existing larger-scale monitoring efforts that address similar questions and concerns at the regional, state, and national levels. For example, the monitoring design to assess question 1, regarding the ambient condition of streams, can be seen as a watershed-scale counterpart to the Stormwater Monitoring Coalition's (SMC) Southern California Regional Monitoring Program, the state's Surface Water Ambient Monitoring Program (SWAMP), and the US Environmental Protection Agency's (USEPA) Western Environmental Monitoring and Assessment Program (Figure 2). These programs are embedded, one within the other, as a result of their shared probabilistic monitoring designs. This feature allows managers to compare the findings from their own watersheds to those of other watersheds in the region, the state, and the western United States. Other benefits of this program include the integration, coordination, and standardization of sampling protocols, laboratory methods, quality assurance programs, and data management efforts.

This paper describes the utility of integrated watershed monitoring programs for informing watershed managers, regulators, scientists, and the public regarding the current state of their watersheds. The results from five years of monitoring by SGRRMP provide an example of how this type of monitoring approach can address a wide range of management questions and improve monitoring efficiencies. The goals of this paper are to (1) provide a summary of the monitoring results for questions 1 and 2 for

SGRRMP's first five years (2005 to 2009), (2) show how these results have informed management decisions, and (3) describe how special studies are being designed to answer questions that arise as a result of this effort.

Methods

Study Area

The San Gabriel River watershed, located in coastal Southern California, is semi-arid with a Mediterranean climate (Figure 3). It is bounded by the San Gabriel Mountains to the north, the San Bernardino Mountains to the east, the watershed divide with the Los Angeles River to the west, and the Pacific Ocean to the south. Approximately half of the 1,785-km^2 watershed consists of extensive areas of undisturbed riparian, chaparral, and woodland habitats within the Angeles National Forest in the upper watershed. The heavily urbanized lower watershed is home to more than 2.3 million people. This part of the river and its major tributaries flow primarily in concrete-lined or heavily shored, soft-bottom channels. The river finally passes through the San Gabriel River Estuary, a shored, soft-bottom channel that discharges to the Pacific Ocean in the city of Long Beach.

Figure 3. The San Gabriel River watershed and San Gabriel River Regional Monitoring Program sampling sites, 2005–2009.

Sampling

To assess question 1, regarding the condition of streams in the watershed, SGRRMP sampled a total of 69 sites from 2005 through 2009 (Figure 3). SGRRMP determined sampling locations using a "master list" approach to integrate sampling efforts by multiple agencies and to facilitate collaboration with other monitoring programs (Larsen et al. 2008). Between 2005 and 2008, USEPA randomly selected sites for SGRRMP using a spatially balanced, generalized, random-tessellation design (Stevens and Olsen 2004). Sites were drawn with the entire watershed representing a single stratum, but weighted so that an even number of sites were drawn from each of three distinct watershed subregions: the upper watershed, lower watershed, and mainstem channel (Figure 3). Starting in 2009, SGRRMP integrated into the newly developed and larger regionwide SMC program, which uses a master list of more than 50,000 sites that are randomly distributed across the stream network of the entire Southern California region using the same spatially balanced, generalized, random-tessellation design. SGRRMP then assigned sites to the watershed using a geographic information system. SGRRMP classified sites by (1) Strahler stream order, using the National Hydrography Dataset Plus and (2) land use, based on the designation of the stream segment. SGRRMP excluded streams below second order from the survey because these sites are typically non perennial or inaccessible in mountainous regions.

SGRRMP employed a monitoring approach using multiple lines of evidence to assess stream conditions, including measurements for chemical, toxicological, biological, and physical habitat (Figure 4).

SGRRMP used bioassessment, a measure of the structure of one or more components of the instream biological community, to assess the ecological status of instream benthic macroinvertebrate (BMI) communities. The field protocols and assessment procedures followed the California SWAMP (2007) stream bioassessment protocol. SGRRMP identified BMIs to level II (generally, the species level), as specified by the standard taxonomic effort list of the Southwest Association of Freshwater

Invertebrate Taxonomists (Richards and Rogers 2006). Using BMI data collected from perennial streams, SGRRMP calculated biological metrics including diversity, average tolerance scores, and functional feeding groups. SMC's Regional Monitoring Program defines perennial steams for Southern California as those flowing through September 30 because of the highly intermittent nature of stream flow in the region. From these metrics, SGRRMP calculated the multimetric Southern California index of biological integrity (IBI) for each site (Ode et al. 2005). The IBI score derived for each site allows for a comparison of that site's biological community with that of "undisturbed" reference sites in Southern California. The sampling index period for surveys of all components of stream condition was May through July.

SGRRMP assessed physical habitat conditions using two methods. The first is a method originally developed by USEPA and modified by SWAMP (2007) for use in California. This method focuses on the habitat conditions found in the streambed and riparian corridor, including streambed morphology (e.g., width, depth, and bankfull width), vegetative density and canopy cover, substrate composition, sedimentation, human influences, and flow regimes. The second measure, the California Rapid Assessment Method (CRAM), more broadly characterizes the overall function and quality of the riparian and buffer zone system (Collins et al. 2008). The CRAM score includes the hydrologic, physical, biological, and buffer zone conditions of the habitat out to 500 m on either side of the streambed. The maximum possible score represents the best condition likely to be achieved for the type of wetland being assessed. The overall score for a site therefore indicates how it is doing relative to the best achievable conditions for that wetland type in the state.

SGRRMP used a target-site approach to address question 2, which asks whether conditions at sites of unique interest are getting better or worse. This target-site approach differs from the random sampling design used to assess ambient stream conditions for question 1 because target sites are revisited annually as opposed to only once. SGRRMP monitored stream confluences and four wetland sites to determine how the chemical, toxicological, biological, and physical habitat conditions might be improving or declining over time.

SGRRMP selected the four wetland sites because of their relatively natural state in otherwise heavily urbanized areas. Assessing the baseline condition of these sites and following them over time will inform either restorative or protective management actions. The four sites included one estuarine habitat, Los Cerritos wetland in Long Beach, and three riverine wetlands: Santa Fe dam scrub habitat in Irwindale, Walnut Creek County Park in San Dimas, and a localized wetland area at Whittier Narrows. SGRRMP performed CRAM assessments annually (in 2008 and 2009) at the three riverine wetlands, and visited Los Cerritos wetland on three separate occasions in 2008 only.

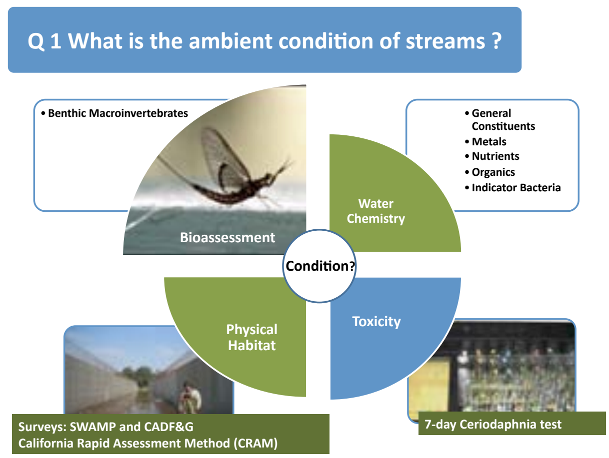

Figure 4. Multiple lines of evidence used to assess stream condition. CADF&G, California Department of Fish and Game.

To assess temporal trends at the sub watershed level, SGRRMP monitored eight sites upstream of confluence points in the upper and lower watersheds. SGRRMP has collected a total of 40 samples from the eight target sampling locations—1 sample per site for five years, from 2005 to 2009. SGRRMP analyzed target site samples for aquatic chemistry, toxicity, biota, and physical habitat condition as described above for question1.

Laboratory Analysis

Table 1 lists the chemical constituents measured at each site and the method's detection limits. The analytical methods for each chemical constituent, as well as data quality objectives for each group of constituents, can be found in the SGRRMP quality assurance project plan (Los Angeles & San Gabriel Rivers Watershed Council and Aquatic Bioassay & Consulting Laboratories 2010). SGRRMP performed toxicity testing on 100% stream water using the water flea (*Ceriodaphnia dubia*) seven-day survival and reproduction test (USEPA 2002).

Table 1. Methods and minimum detection limits for measured water quality parameters in freshwater.

Analyte	Method	Minimum Detection Limit
Ammonia as N	SM 4500-NH3 D[b]	0.03–0.05 mg/L
Dissolved Organic Carbon	EPA 415.1[c]	0.013 mg/L
Nitrate as N	EPA 300.0[c]	0.013 mg/L
Nitrite as N	EPA 300.0 [c]	0.01 mg/L
Alkalinity as $CaCO_3$	SM 2320B[b]	1–1.2 mg/L
Hardness as $CaCO_3$	SM 2340B[b]	0.089–1 (mg $CaCO_3$/L)
Total Nitrogen	Calculated	Calculated (mg/L)
Total Kjeldahl Nitrogen	EPA 351.3 [c]	0.74 mg/L
Total Organic Carbon	EPA 415.1 [c]	0.13–0.32 mg/L
Orthophosphate as P	SM 4500-P E[b]	0.00083–0.01 mg/L
Phosphorus as P	SM 4500-P C[b]	No value[a]
Total Suspended Solids	SM 2540D[b]	0.5–5 mg/L
Trace Metals (total and dissolved)	EPA 200.8 [c]	0.008–0.6 µg/L
Mercury (total and dissolved)	EPA 1631 [c]	0.0005–0.0039 µg/L
Toxicity	(*Ceriodaphnia dubia*) test [d]	

Note: $CaCO_3$, calcium carbonate; N, nitrogen; P, phosphorus.
[a] No minimum detection limit reported; reporting limit range = 0.1–0.5 mg/L.
[b] American Public Health Association (2005)
[c] USEPA (n.d.[a])
[d] USEPA (2002)

Data Analysis

SGRRMP characterized aquatic chemistry and physical habitat data from each of the three subregions using descriptive statistics, including the means, standard deviations, medians, and ranges of concentrations (R-CRAN statistical software). Where applicable, SGRRMP compared aquatic chemistry values to numeric regulatory thresholds, such as those specified in the Los Angeles basin plan objectives (Los Angeles Regional Water Quality Control Board 1994) and the California Toxics Rule (CTR; USEPA n.d.[b]), to determine the number of times they exceeded these values.

To assess the biological condition of streams, SGRRMP compared area-weighted IBI scores against reference site conditions in Southern California. IBI scores below 39 (on a scale of 100) represent communities that are below reference conditions, and those 39 and above represent sites where biological conditions are similar to reference site conditions in the region.

The determination of toxic endpoints for the water flea seven-day survival and reproduction test was based on (1) a statistically significant difference in either survival or reproduction between water fleas held in 100% stream water and those held in laboratory control water and (2) a response of less than 80% for either survival or reproduction.

SGRRMP calculated an overall CRAM score for each site from four main attribute scores and their metrics: landscape context and buffer, hydrology, physical structure, and biotic structure. No regulatory thresholds are described for CRAM scores; the lowest CRAM score possible for these sites is 27, and the maximum score is 100. SGRRMP compared the overall scores across sites and years to determine temporal and spatial trends.

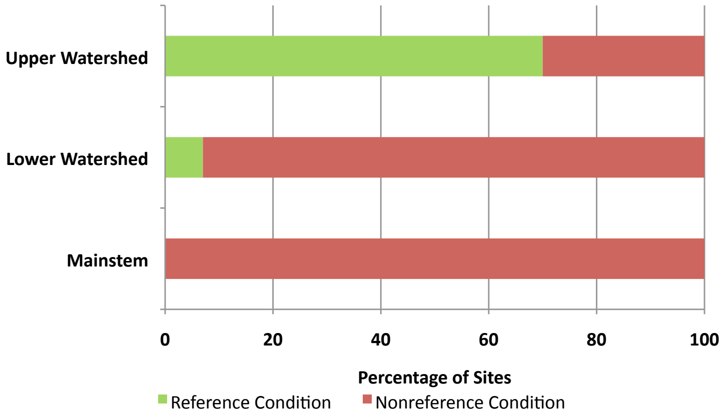

Figure 5. Biological condition (IBI scores) for different watershed subregions.

Results

Q1: What is the condition of streams in the watershed?

SGRRMP collected and assessed aquatic chemistry, toxicity, bioassessment, and physical habitat data from 69 randomly selected sites throughout the San Gabriel River watershed from 2005 through 2009. During this five-year period, BMI communities in the upper watershed had IBI scores greater than 39, indicating that BMI communities there were similar to those found at reference sites throughout Southern California (Figure 5, Table 2). Only 30% of stream miles in the watershed had IBI scores similar to those of reference sites. When evaluated by subregion, 70% of upper watershed sites were in good condition, whereas only 7% of the lower watershed tributaries and none of the mainstem sites were in good biological condition. Interestingly, several upper watershed sites that appeared to have good water quality and physical habitat conditions had biological communities that were impaired relative to reference sites. This has triggered follow-up studies to investigate the source of the impairment.

Biological communities in the upper watershed exhibited a wide range of feeding strategies and were characterized by pollution-sensitive organisms (Figure 6). Collector species dominated this subregion, but a wide range of other groups, including grazers, filterers, and predators, made up a combined 20% of the population. The upper watershed was the only subregion where highly sensitive species were found, such as stoneflies (*Calineuria californica*, *Makenka sp.*, and *Sweltsa sp.*), mayflies (*Drunella sp.*, *Ephemerella sp.*, and *Epeorus sp.*), and caddisflies *Micrasema sp.*, *Parapsyche sp.*, *Rhyacophila sp.*, and *Lepidostoma sp.*). In contrast, the biological communities in the lower watershed were more degraded, as evidenced by lower IBI scores (below 39); less diverse feeding strategies, such as fewer predator and collector taxa; and the dominance of organisms more tolerant of pollution, such as Oligochaetes, Ostracoda, *Hyalella sp.* (Amphipoda), and gastropods (*Physa sp.*).

A comparison of chemical constituents revealed differences in concentrations in the upper watershed, lower watershed, and mainstem (Table 2). Nutrient and metal concentrations were consistently lower at upper watershed sites than in the lower tributaries and the mainstem. Nutrients were greatest in the mainstem, whereas most metals were greatest in the lower tributaries. An exception to this was dissolved zinc, which was much greater in the mainstem compared to the other subregions. Aquatic chemistry concentrations rarely exceeded numeric regulatory thresholds during the five-year period. Nitrate and ammonia were well below toxicity thresholds, and SGRRMP found no exceedances of the hardness-adjusted CTR threshold for any dissolved metal. Nearly all organic constituents, including organophosphorus and pyrethroid pesticides, were always below the limits of detection.

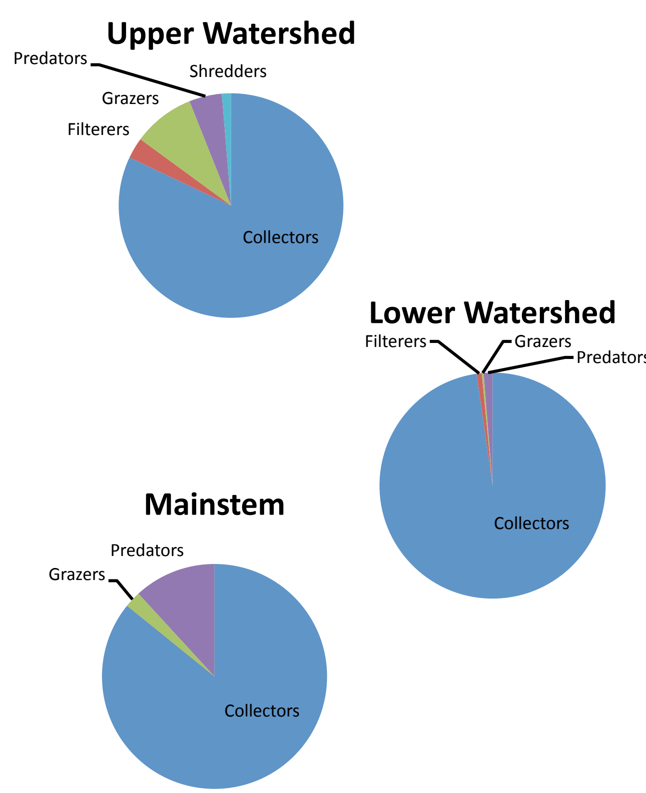

Figure 6. Relative proportions of macroinvertebrate functional feeding groups in each watershed subregion for all random sites combined, 2005 to 2009.

Table 2. Summary statistics for samples collected from three subregions of the San Gabriel River watershed and compared against regulatory water quality objectives where applicable.

	Upper Watershed		Mainstem		Lower Watershed		Min	Max	No. of Exc.
	Mean ± Std. Dev.	Median	Mean ± Std. Dev.	Median	Mean ± Std. Dev.	Median			
General Chemistry									
DO (mg/L)	8.3 ± 1.9	8.8	12 ± 4.2	12	11 ± 4.9	9.9	1.9	24	4
pH (-log[H$^+$])	8.1 ± 0.28	8.2	8.1 ± 0.6	8.3	8.4 ± 0.79	8.1	7.2	10	16
Salinity (mS/cm)	0.2 ± 0.07	0.2	0.59 ± 0.08	0.60	5.8 ± 18	0.6	0.08	79	no obj.
Temperature (°C)	16 ± 2.4	16	28 ± 1.7	28	23 ± 5.7	23	12	36	
Alkalinity (mg/L)	196 ± 56	183	154 ± 19	154	203 ± 109	199	64	448	
Hardness (mg/L)	185 ± 88	162	197 ± 47	200	398 ± 315	266	74	1480	
TSS (mg/L)	2.86 ± 1.89	2.5	7.6 ± 5.5	5.0	38 ± 89	38	0.50	408	
TOC (mg/L)	1.7 ± 0.79	1.8	6.4 ± 0.87	6.9	12 ± 13	6.6	0.47	46	
Dissolved Metals (µg/L)									
As	11 ± 1.7	0.50	1.0 ± 0.37	0.90	2.1 ± 2.4	1.9	0.20	12	0
Cr	0.22 ± 0.07	0.25	0.74 ± 0.65	0.60	0.82 ± 0.88	0.6	0.08	4.1	0
Cu	0.6 ± 0.32	0.50	3.5 ± 1.9	2.7	6.1 ± 5.7	3.6	0.13	22	2[b], 1[c]
Fe	36 ± 36	25	72 ± 27	70	82 ± 108	48.3	1.25	465	no obj.
Pb	0.19 ± 0.48	0.05	0.27 ± 0.17	0.22	0.45 ± 0.74	0.2	0.01	2.5	0
Ni	0.28 ± 0.11	0.25	3.8 ± 0.89	4.0	3.4 ± 4.7	2.5	0.01	23	0
Se	0.33 ± 0.13	0.28	0.61 ± 0.34	0.50	2.4 ± 2.0	2.2	0.00	6.7	3c
Sr	348 ± 121	333	528 ± 130	548	817 ± 508	732	174	2176	no obj.
Zn	1.5 ± 1.9	0.50	36 ± 2.9	37	11 ± 9.3	8.5	0.02	39	0
Nutrients (mg/L)									
NH_4^+ (total)	0.04 ± 0.02	0.05	0.23 ± 0.26	0.19	0.11 ± 0.14	0.1	0.01	0.90	
NO_3^- N (dissolved)	0.27 ± 0.29	0.10	4.4 ± 4.4	3.4	1.2 ± 1.6	0.0	0.01	21.5	1
NO_2^- N (dissolved)	0.02 ± 0.01	0.02	0.18 ± 0.1	0.19	0.03 ± 0.03	0.0	0.01	0.38	0
TN-Kjeldahl	0.44 ± 0.47	0.25	1.78 ± 0.6	1.7	1.9 ± 1.9	1.1	0.05	7.4	
PO_4 (total)	0.12 ± 0.12	0.09	0.3 ± 0.26	0.25	0.16 ± 0.25	0.1	0.01	1.2	
TP	0.03 ± 0.01	0.03	0.26 ± 0.13	0.21	0.31 ± 0.41	0.1	0.02	1.6	
Physical Habitat Assessments									
CRAM Score	82 ± 11	82	34 ± 3.4	34	42 ± 15	37	27	96	
IBI Score	52 ± 17	52	17 ± 9.9	16	16 ± 13	10	0	89	40

Note: As, arsenic; Cr, chromium; Cu, copper; DO, dissolved oxygen; Exc., exceedances; Fe, iron; NH_4^+, ammonium; Ni, nickle; NO_2^-, nitrite; NO_3^-, nitrate.; no obj., no objective; Pb, lead; PO_4, orthophosphorus; Se, selenium; Sr, strontium; TN, total nitrogen; TOC, total organic carbon; TP, total phosphorus; TSS, total suspended solids; Zn, zinc.
DO water quality objective: 5 or ≥ 7.
pH water quality objective: 6.5 – 8.5.
NO_3^- N water quality objective: 10 mg/L.
NO_2^- N water quality objective: 1 mg/L.
[a] Hardness-adjusted dissolved metals compared to the CTR
[b] CTR acute threshold value
[c] CTR chronic threshold value

SGRRMP tested a total of 61 water samples for acute and chronic toxicity using water fleas. Out of 122 survival and reproduction endpoints measured, 13 (11%) indicated toxicity in at least one sample. Toxic endpoints indicative of toxicity were most frequent in 2005, when 6 of the 23 samples (26%) showed reproductive toxicity. All of the toxic endpoints measured during the five years were in the lower or upper watershed; no toxicity was measured on the San Gabriel River mainstem (Table 3).

Q2: Are conditions at areas of unique interest getting better or worse?

Assessing the baseline condition of sites and following them over time can inform managers regarding the success or necessity of restorative or protective measures. SGRRMP chose major stream confluences to act as water quality sentinel sites for the main subwatersheds. The four wetland sites chosen by stakeholders represent some of the last relatively natural ecosystems in the highly urbanized lower watershed; by understanding their status, managers may be able to make better decisions regarding their protection and/or restoration.

The results from the target sampling sites support the spatial variability in IBI scores shown by random sites. Biological communities were consistently similar to reference conditions at upper watershed confluence sites and impaired at sites in the lower watershed (Figure 7). Interestingly, Site 505 is located immediately below Morris dam, the last reservoir in the upper watershed before the river enters the highly urbanized lower watershed. The riparian zone at this site is in relatively good condition; however, the intermittent discharges from the dam are evidenced in the impaired IBI score.

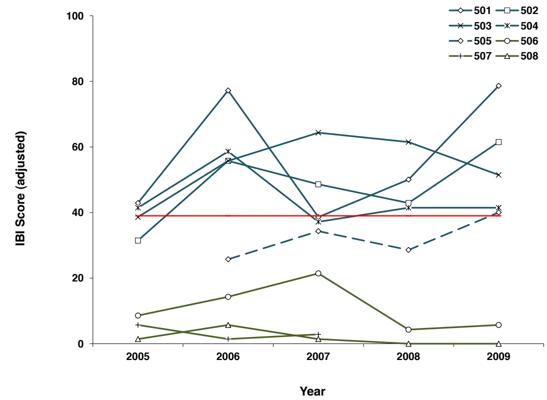

Figure 7. Southern California IBI scores at confluence sites. Sites with an IBI ≥ 39 (red horizontal line) have biological communities similar to those of reference sites for the region; sites with an IBI < 39 have biological communities that are degraded relative to reference conditions.

We found no clear temporal trends in aquatic chemistry parameters, particularly for those constituents with inherently high daily variability, such as pH and water temperature. Similarly, we observed very little annual variability for

Table 3. Water flea (*Ceriodaphnia dubia*) survival and reproduction—significant response endpoints.

Year	Endpoint	Significant Endpoints		Significant Response by Subregion		
		n	Signif. Tox.	Mainstem	Lower	Upper
2005	Survival	23	1	0	0	1
	Reproduction	23	5	0	2	3
2006	Survival	10	0	0	0	0
	Reproduction	10	0	0	0	0
2007	Survival	9	0	0	0	0
	Reproduction	9	2	0	1	1
2008	Survival	9	2	0	1	1
	Reproduction	9	2	0	1	1
2009	Survival	10	0	0	0	0
	Reproduction	10	1	0	1	0
	Totals	**122**	**13**	**0**	**6**	**7**
	%		**11**	**0**	**5**	**6**

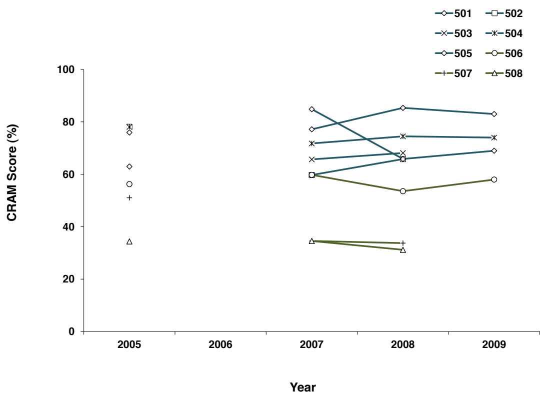

Figure 8. California Rapid Assessment Method scores at confluence sites.

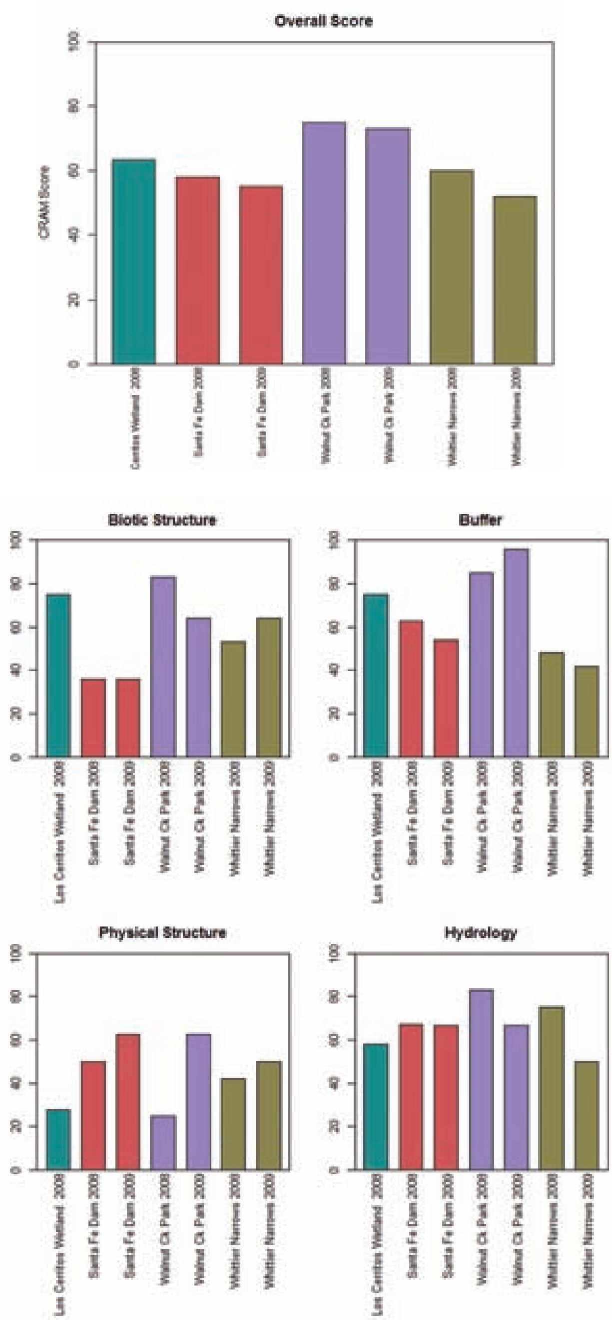

Figure 9. California Rapid Assessment Method attribute and overall scores for four unique habitats.

physical habitat conditions, as measured by CRAM, over the period (Figure 8).

CRAM assessment at each of the three riverine wetland habitats was relatively stable over the two-year period (Figure 9). The highest scores were calculated for Walnut Creek Park, which is characterized by a relatively natural streambed, wide and pervious buffer zones, good vegetative cover and layering, and few nonendemic species. Whittier Narrows and Santa Fe dam had lower CRAM scores, mostly because of the relatively poor buffer zone and poor biotic structure, respectively. CRAM scores at Los Cerritos wetland in 2008 were moderate. One of the last functioning estuarine wetlands in the greater Los Angeles area, Los Cerritos wetland is encroached upon from all sides by break walls, shoring, and heavy urbanization. A large effort is underway to protect and restore this habitat.

Discussion

Prior to 2005, managers knew little about the ambient water quality condition of streams in the San Gabriel River watershed, other than at fixed points located around discharges monitored under the National Pollutant Discharge Elimination Systems mainly in the cement-lined mainstem channel. As a result, the conditions in the lower watershed tributaries and upper watershed were unknown. The results from the first five years of monitoring clearly demonstrate the value of combining randomized watershed-scale sampling with targeted sampling at confluences and sites of unique interest. The multiple lines of evidence collected by SGRRMP (bioassessment, aquatic chemistry, aquatic toxicity, and physical habitat) have (1) provided a basis for investigating the factors contributing to the degradation of stream condition and (2) enabled stakeholders to begin to draw conclusions about the condition of the entire watershed.

Most importantly, the state of the biological communities was strongly associated with the physical habitat conditions of the streambeds and riparian zones. This suggests that protective measures should include efforts to reduce impacts to physical habitat in the upper watershed while simultaneously restoring riparian and stream habitat in the lower watershed where possible. SGRRMP did not find evidence of widespread impairment of water quality based on levels of individual chemicals or measures of toxicity.

When we observed toxicity, it was confined to sites in the upper watershed. For individual chemical constituents, such as copper, selenium, and zinc, the exceedances of regulatory objectives are localized; managers can use this information to implement best management practices to reduce the sources and/or concentrations of the contaminants.

A clear benefit to managers who choose to use the probabilistic sampling design will be the ability to compare the San Gabriel River watershed with other watersheds in the state and throughout the western United States. For example, in 2009, SMC's Regional Monitoring Program identified aquatic toxicity at numerous upper watershed sites throughout Southern California (Mazor et al. 2009). Prior to this, SGRRMP stakeholders assumed that the toxicity measured in the upper San Gabriel River watershed was an anomaly in the region. Potential sources of this toxicity, which are currently under investigation, include contaminants that are not being measured, underlying geologic features of the region, or naturally occurring cyanotoxins (products of blue-green algal metabolism).

Moreover, although biological communities in the upper watershed were generally similar to reference conditions, IBI scores were below the impairment threshold (39) at several sites during the five-year period. These sites had good physical habitat conditions and did not exceed regulatory thresholds for measured chemical constituents. Data from regional and statewide monitoring programs support these results; this has led to a much larger discussion regarding which stream reaches in California are truly perennial. This is important because the IBI scores developed for each of the state's ecoregions are based on biological condition data collected from perennial streams. It is not known how intermittent drying of a streambed might affect the biological communities.

The areas of concern identified after five years of monitoring by SGRRMP are consistent with the findings from other regional and state monitoring programs. As a result, collaborative efforts to design, fund, and conduct special follow-up studies are preferred over watershed-specific studies with more limited applicability. The follow-up toxicity study will potentially revisit sites that previously showed evidence of toxicity throughout the Southern California region to conduct toxicity identification evaluations, a process designed to identify the contaminant(s) causing toxicity. SMC is designing a stream perenniality study; this study will require site revisits throughout the Southern California region at the end of the dry season (September) to determine whether streams are still flowing. These studies have resulted from the probabilistic sampling design employed at the local, regional, and state levels.

Results from five years of monitoring at confluence sites and sites of unique interest demonstrated that trends are not discernible at this monitoring frequency. Other longer-term monitoring programs, such as the US Geological Survey's National Stream Quality Accounting Network, suggest that many more years of monitoring at target sites will be required to clearly discern trends.

The design of SGRRMP is based on clear statements of rationale and criteria for decision making about design options. SGRRMP also reflects a high degree of consensus among a broadly representative group of stakeholders in the watershed. It represents a significant advance toward the regional integration of monitoring efforts and the assessment of watershed condition. However, it is important to recognize that, although the program will enhance the ability to assess the status of some beneficial uses, it will not provide the means, across the entire watershed, to (1) fully determine compliance with water quality objectives, (2) define impairment, or (3) determine whether the requirements of the listing/delisting process under Section 303(d) of the Clean Water Act are being met. Such purposes require more spatially and temporally intensive sampling efforts, the requirements of which are met by only some of the components of SGRRMP.

Conclusion

SGRRMP has successfully shown that an integrated watershed monitoring program can provide context to essential management questions, improve monitoring efficiencies, and provide a collaborative platform for the comparison of monitoring results at the local, regional, and state levels. In the future, SGRRMP will continue to address specific issues, such as changes in the condition of critical habitat areas and public health risks associated with swimming or consuming fish. During 2011–2012, the program will (1) fund pilot studies to gain a better understanding of the speciation of mercury in fish tissues, (2) collaborate with the State Water Board's SWAMP to monitor polybrominated diphenyl ethers (flame retardants) in sediments within the watershed, and (3) continue sampling at sites burned by the 2009 Morris fires to monitor their recovery. SGRRMP is focused on assisting watershed managers in identifying areas of concern to prioritize management actions.

Acknowledgments

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A Method for Disaggregating Existing Model Pollutant Loads for Subwatersheds

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Abstract

Sediment is the primary pollutant that results in nonattainment of Virginia's aquatic life use (general) water quality standard. Because the US Environmental Protection Agency's Total Maximum Daily Load (TMDL) Program requires pollutant load reductions that are protective of aquatic life use, and because Virginia has no sediment water quality standard, modeling procedures were needed to quantify existing and endpoint sediment loads and the corresponding required pollutant reductions. Previous sediment TMDLs in Virginia used a paired reference watershed approach (Yagow 2004). However, the recent model-based quantification of the Chesapeake Bay TMDL offers a simpler and potentially more consistent method for calculating target sediment loads for impaired watersheds within the Chesapeake Bay watershed. This paper illustrates the application of an alternative procedure, the *disaggregate* method, for developing target pollutant loads; this method should be applicable to many watersheds nationwide. The disaggregate method uses land use inputs to, and pollutant load outputs from, an existing model together with a locally derived land use inventory. Using this method, one can determine the pollutant load reductions needed to achieve target pollutant loads for upstream, low-order subwatersheds whose areas are smaller than the smallest modeling segments generally used in basin-scale modeling.

Introduction

Water quality modeling is often performed at the basin scale for planning purposes. However, modeling at this scale often yields insufficient detail for establishing specific loads or for determining specific, needed management changes at the subwatershed scale. This paper describes the *disaggregate* method, which determines target pollutant loads from land-based pollutant sources at the subwatershed scale, allowing for the development of more fine-tuned pollutant control measures. The method uses land use–specific unit-area loads (UALs)—calculated from the output of existing models of land-based pollutant sources (as opposed to point or population-based sources) coupled with fine-scale local land use data—to determine target pollutant loads. This method further increases the utility of existing model output by providing information for management decisions at a finer geographic level. Furthermore, the disaggregate method should promote greater consistency between larger-scale (basin-level) and smaller-scale (subwatershed-level) planning efforts.

Modeling studies typically include a scenario that represents existing conditions and one or more management scenarios that explore different ways to achieve some targeted load reduction. One widespread application of modeling is for load quantification in the US Environmental Protection Agency's (USEPA) Total Maximum Daily Load (TMDL) Program. The TMDL Program is based on Section 303(d) of the 1985 federal Clean Water Act and USEPA's current water quality planning and management regulations, 40 CFR Part 130 (2012), which require states to identify causative pollutants and develop TMDLs for "impaired" water bodies that violate state water quality standards (USEPA 1999). A TMDL study determines (1) the amount of each identified causative pollutant a water body can receive and still meet water quality standards and (2) the level of load reductions required from each source category. Essentially, a TMDL provides an outline of actions needed to restore water quality.

USEPA's Chesapeake Bay Program developed the Chesapeake Bay Watershed Model (CBWM) to simulate the fate and transport of nutrients and sediment in the 64,000-square-mile (mi^2; 165,760-km^2)[1] watershed that drains to the Chesapeake Bay. This model has evolved over time in complexity and accuracy. The first version was developed in 1983; the latest version (phase 5.3.2) was released in June 2011. Significant efforts have gone into developing the CBWM, and its characterization of nutrient and sediment sources contributing to the Bay, and designing the pollution control measures to reduce the adverse impact of

[1] English units have been used throughout this paper based on the CBWM model.

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those sources. USEPA (2010a) has overseen the calibration of the CBWM over a 21-year period at 287 flow gauging stations and at 164 water quality stations with varying periods of sediment data. Although simulated as 1,194 river segments, most of the CBWM inputs are based on county-averaged data and distributed on an area-weighted basis to portions of river segments that intersect each county.

The scale of CBWM output limits the development of targeted management actions at a finer spatial scale. As an example, the 31-mi^2 (80-km^2) Moore's Creek was listed as "impaired" in the 2008 *Virginia Water Quality Assessment 305(b)/303(d) Integrated Report* because of water quality violations of the general aquatic life use water quality standard (Virginia Department of Environmental Quality [VADEQ] 2008). This listing required the state to oversee the development of a TMDL for Moore's Creek. The impaired segment of Moore's Creek is located within the Rivanna River basin in Virginia, with 91% of the contributing watershed area in Albemarle County and the remainder in the City of Charlottesville (Figure 1). The Rivanna River drains into the James River, which empties into the Chesapeake Bay.

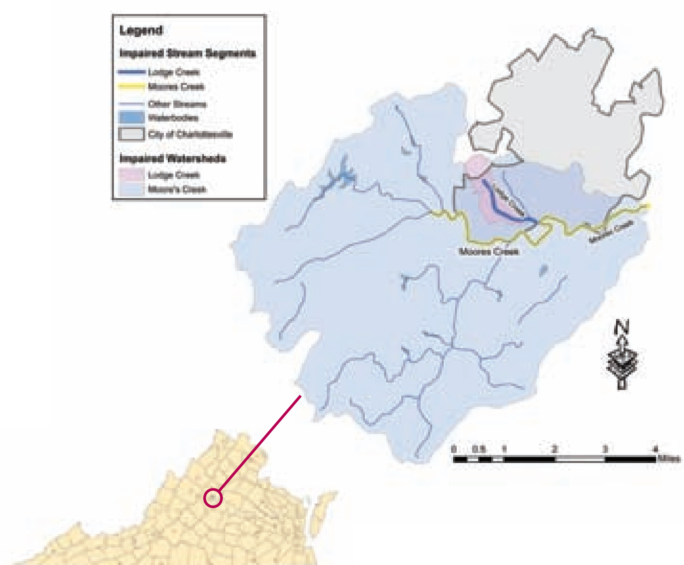

Figure 1. Location of the Moore's Creek subwatersheds.

A violation of the aquatic life use standard in Virginia is based on measurements of the instream benthic macroinvertebrate community compared against an accepted value of Virginia's multimetric stream condition index (VADEQ 2008). A follow-up stressor analysis on the violation in Moore's Creek found that sediment was the most probable stressor, based on repeated poor habitat metric scores and observations of insufficient riparian buffer, erosion, and bank instability at many locations in the watershed.

The development of a TMDL requires the calculation of pollutant loads for an existing, or baseline, condition and for a target condition. The target condition reflects load reductions that are expected to allow the benthic macroinvertebrate community to recover and, in time, to meet the aquatic life use water quality standard. Whereas the identification of impairments is based on monitoring data that are periodic, short-term, and related to ambient conditions, modeling allows the TMDL developer to calculate both existing and target pollutant loads under long-term, variable hydrologic conditions. Because target TMDL loads are typically based on an instream pollutant concentration standard, and because Virginia has no numeric water quality standard for sediment (State Water Control Board 2011), TMDL developers under contract to the state needed a different method for establishing a sediment reference endpoint (the TMDL target load) representing the restoration condition.

In many watersheds with an aquatic life use impairment where sediment has been identified as the primary pollutant, TMDL developers have used a *reference watershed* approach to quantify the TMDL target load for the impaired watershed. This approach pairs two watersheds—one whose streams are supportive of their designated uses (the reference watershed) and one whose streams are impaired. TMDL developers select a reference watershed based on its similarity with the impaired watershed in terms of land use and topographical, ecological, and soils characteristics. They then simulate sediment loads for both watersheds and use the area-adjusted load from the reference watershed as the reference load that quantifies the TMDL target load for the impaired watershed (Yagow 2004).

Prior to the development of the Chesapeake Bay TMDL (USEPA 2010a), the state coordinated development of many local TMDLs for sediment throughout the Chesapeake Bay watershed in Virginia; but most of these TMDLs were developed independently of each other and focused on headwater stream segments. The process for development of these local TMDLs did not include considerations of downstream water quality consequences—for instance to the Chesapeake Bay. As of December 30, 2010, however, all of the waters in the Chesapeake Bay watershed, including the Moore's Creek watershed, also became subject to the provisions of the Chesapeake Bay TMDL, which includes a sediment load component. As a result, all TMDL target loads for the same pollutant in the same river basin must sum up to the TMDL load for each of the 92 impaired downstream Chesapeake Bay tidal segments. The disaggregate method arose from the need to maintain a degree of consistency between the development of local upstream TMDLs and the downstream Chesapeake Bay TMDLs.

The Moore's Creek watershed includes portions of two CBWM land–river segments, the smallest geographic units in the model. For load calculations, we applied the disaggregate method to each portion separately—the Albemarle County portion and the City of Charlottesville portion—and summed together the loads from each portion. This paper illustrates the application of the disaggregate method to quantify a long-term average annual TMDL target sediment load to address the aquatic life use impairment for the Albemarle County portion of Moore's Creek, referred to as "Moore's Creek (Alb)." This illustration uses CBWM-simulated, land-based pollutant load output from the Albemarle County land–river segment and applies it to the local land use inventory for the Moore's Creek (Alb) portion.

The disaggregate method arose from the need to maintain a degree of consistency…

The Disaggregate Method

The disaggregate method uses simulation inputs and outputs from an existing model, including pollutant loads by land use and land use areas, to calculate UALs in units of tons per acre per year for each land use within the smallest available geographical modeling segment. One then applies the UALs from the existing model to a spatially derived local land use inventory that is presumably more representative of the geographically smaller, impaired subwatershed to calculate pollutant loads. The disaggregate method allows one to determine loads for both existing (baseline) and future (target) conditions. The future conditions include a representation of management measures to achieve the required pollutant reduction. Below, we describe the disaggregate method in general and then illustrate each step using the CBWM data for the Moore's Creek (Alb) application example.

Step 1. Download Existing Model Land Use Data and Create Land Use Groups

In this step, one obtains the land use category and area distribution from existing model inputs for the smallest model segment that includes the subwatershed of interest. If all of the land use categories are not spatially explicit (derived from a hard copy or digital map source), some type of grouping of the land use categories may be necessary to provide a basis for matching with the local land use inventory and categories (see step 2).

In the case of Moore's Creek, we obtained land use category and area data (inputs to the CBWM) and simulated sediment loads (output from various CBWM simulation scenarios) using the online Virginia Assessment Scenario Tool developed for the Commonwealth of Virginia by the Interstate Commission on the Potomac River Basin (2011). We obtained output for two modeling scenarios: we used the 2009 Progress–VA scenario for existing (baseline) load calculations and the WIP 1–VA scenario (a November 7, 2011 modification of the Virginia Watershed Implementation Plan for the Chesapeake Bay TMDL) as the reference (target) scenario to quantify the TMDL endpoint.

The CBWM incorporates 31 land use categories (USEPA 2010b). Since the disaggregate method applies only to land-based pollutant sources, this paper does not discuss the four point source categories that are also included in the CBWM (details available in Yagow et al. 2011). USEPA's Chesapeake Bay Program created the CBWM's 31 land-based land use categories using a combination of digital spatial data, such as National Land Cover Data imagery; statistical data, such as the US Department of Agriculture's Census of Agriculture statistics data, by county; and state-specific databases describing the type and extent of implemented best management practices (BMPs). To relate the more detailed CBWM land use categories to fewer, less specific, locally developed land use categories, we combined many of the CBWM's 31 land use categories into broader agricultural and urban/residential land use groups (Table 1). Table 1 shows the distribution of specific land use categories within each land use group; the color coding used to distinguish land use groups in Table 1 is repeated in subsequent tables.

Table 1. Existing model (CBWM Albemarle segment) land use categories, aggregated land use groups, and land use category distributions within each land use group.

CBWM Land Use Code	CBWM Land Use Category	Area (acres)	Land Use Group	Distribution within Each Group (%)
hom	High-till without manure	282.7	Conventional tillage, no manure	95.9
nho	High-till without manure NM	12.1		4.1
hwm	High-till with manure	49.9	Other row crops	46.6
nhi	High-till with manure NM	2.1		2.0
lwm	Low-till with manure	52.7		49.3
nlo	Low-till with manure NM	2.3		2.1
hyw	Hay with nutrients	4,262.4	Hay	72.3
nhy	Hay with nutrients NM	182.2		3.1
alf	Alfalfa	123.5		2.1
nal	Alfalfa NM	5.3		0.1
hyo	Hay without nutrients	1,325.8		22.5
pas	Pasture	8,400.3	Pasture	93.3
npa	Pasture NM	359.1		4.0
trp	Pasture corridor	245.9		2.7
afo	Animal feeding operation	39.0		0.0
cfo	Confined animal feeding operation	0.0		0.0
for	Forest	68,032.1	Forest	99.0
hvf	Harvested forest	685.8		1.0
cid	CSS impervious developed	0.0	Impervious developed	0.0
rid	Regulated impervious developed	766.0		29.2
nid	Nonregulated impervious developed	1,858.0		70.8
cpd	CSS pervious developed	0.0	Pervious developed	0.0
rpd	Regulated pervious developed	3,762.0		43.5
npd	Nonregulated pervious developed	4,712.0		54.5
ccn	CSS construction	0.0		0.0
rcn	Regulated construction	166.2		1.9
cex	CSS extractive	0.0	Extractive	0.0
rex	Regulated extractive	0.0		0.0
nex	Nonregulated extractive	219.4		100.0
urs	Nursery	15.8	Nursery	100.0
atdep	Atmospheric deposition	870.7	Water	100.0

Notes: 1 acre ≈ 0.4046 ha; CBWM, Chesapeake Bay Watershed Model; CSS, combined sewer system; NM, nutrient management.

Step 2. Obtain Local Land Use Data for Baseline Scenario and Assign Land Use Groups

One can often obtain local land use data from a variety of sources, including National Land Cover Data (USEPA 2006), the cropland data layer from the National Agricultural Statistics Service (e.g., NASS 2009), and local sources such as county-level land use data derived from satellite and/or aerial imagery. When land use categories obtained from local sources differ from those used by the larger-scale model, grouping the land use categories into common, broadly defined land use groups allows for matching between the data sources.

We compiled local land use data for the Moore's Creek (Alb) watershed from the Rivanna River Basin Commission's (RRBC) Rivanna Watershed and Vicinity Land Use/Land Cover Map geodatabase (RRBC 2009) and the NASS cropland data layer (NASS 2009). In general, we used the RRBC land use data as the primary source for nonagricultural land uses and the NASS data to quantify agricultural sources. Additional details about the land use data are available in the draft Moore's Creek TMDL report (Yagow et al. 2011). Table 2 summarizes the Moore's Creek (Alb) land use categories and their corresponding assigned land use group.

Step 3. Distribute Locally Derived Land Use Data to Existing Model Land Use Categories

In this step, one sums the areas for each of the land use groups from the locally derived land use data (Table 2) and then redistributes the total area to the existing model's land use categories, using the land use category distribution within each land use group (Table 1).

For the Moore's Creek (Alb) example, we summed the relevant areas from Table 2 for each land use group and redistributed the total area according to the land use category distribution within each land use group from the CBWM land–river segment (Table 1). We calculated the area assigned to animal feeding operations ("afo" in Table 1) based on actual numbers of livestock farms of each animal type, also described in the draft TMDL report (Yagow et al. 2011). We subtracted the afo acreage calculated by this method from the total "Pasture" group acreage. Table 3 shows the summed group areas and the distributed areas. Based on input from local stakeholders, we determined that some of the land use categories in the CBWM Albemarle segment were not present in the Moore's Creek (Alb) watershed.

Table 2. Moore's Creek (Alb) watershed: Local land use categories and corresponding land use groups.

Local Land Use Category	Land Use Data Source	Area (acres)	Land Use Group
Orchard/vineyard	RRBC	60.6	Conventional till., no manure
Corn	NASS	7.2	Other row crops
Soybeans	NASS	3.0	
Hay	NASS	781.5	Hay
Pasture	NASS	207.5	Pasture
Deciduous tree	RRBC	11,097.7	Forest
Evergreen tree	RRBC	1,763.4	
Pine plantation	RRBC	199.9	
Forest harvest	RRBC	20.7	
Urban impervious	RRBC	1,44.6	Impervious developed
Golf course	RRBC	155.4	Pervious developed
Urban pervious	RRBC	4,346.2	
Bare earth	RRBC	47.9	
Water	RRBC	227.7	Water
Total areas (acres)		19,963.4	

Note: T1 acre ≈ 0.4046 ha.

Table 3. Moore's Creek (Alb) watershed: Local land use group areas distributed to CBWM land use categories.

Land Use Group	Group Area (acres)	CBWM Land Use Code	CBWM Land Use Category Name	Distribution within Each Group (%)	Distributed Area (acres)
Conventional tillage, no manure	60.6	hom	High-till without manure	95.9	58.1
		nho	High-till without manure NM	4.1	2.5
Other row crops	10.3	hwm	High-till with manure	46.6	4.8
		nhi	High-till with manure NM	2.0	0.2
		lwm	Low-till with manure	49.3	5.1
		nlo	Low-till with manure NM	2.1	0.2
Hay	781.5	hyw	Hay with nutrients	72.3	564.7
		nhy	Hay with nutrients NM	3.1	24.1
		alf	Alfalfa	2.1	16.4
		nal	Alfalfa NM	0.1	0.7
		hyo	Hay without nutrients	22.5	175.6
Pasture	207.5	pas	Pasture	93.3	189.5
		npa	Pasture NM	4.0	8.1
		trp	Pasture corridor	2.7	5.5
		afo	Animal feeding operation	0.0	4.4
Forest	13,081.7	for	Forest	0.0	12,951.2
		hvf	Harvested forest	99.0	130.6
Impervious developed	1,044.6	cid	CSS impervious developed	0.0	0.0
		rid	Regulated impervious developed	29.2	304.9
		nid	Nonregulated impervious developed	70.8	739.7
Pervious developed	4,549.5	cpd	CSS pervious developed	0.0	0.0
		rpd	Regulated pervious developed	43.5	1,980.9
		npd	Nonregulated pervious developed	54.5	2,481.1
		ccn	CSS construction	0.0	0.0
		rcn	Regulated construction	1.9	87.5
Water	227.7	atdep	Atmospheric deposition	100.0	227.7
Total	19,963.4				19,963.4

Notes: 1 acre ≈ 0.4046 ha; CBWM, Chesapeake Bay Watershed Model; CSS, combined sewer system; NM, nutrient management.

Step 4. Calculate Local Land Use Distribution for a Target Pollutant Reduction Scenario

In step 1, one obtains existing model data for a baseline scenario. In this step, one obtains similar data for a target scenario. In some cases, the baseline and target land use categories and the areal distributions may be the same. However, in many cases, one may need to use additional land use categories, or shift land use areas from one category to another, to represent the management changes that result in the pollutant load reductions associated with the target scenario.

The baseline and target scenarios each simulated different combinations and extents of implemented BMPs.

In the Moore's Creek (Alb) watershed, the CBWM runs we used in creating the targeted TMDL scenario were based on land use categories that incorporated BMPs. Some of these BMPs were represented as a change in area from one land use to another, while other BMPs were represented as reductions in load—either applied to the land surface, or delivered to the edge-of-stream. BMPs simulated as load reductions resulted in changes in the UALs for the applicable land use. The baseline and target scenarios each simulated different combinations and extents of implemented BMPs. The disaggregate method represents the shift in acreage between the baseline and target scenarios, both as changes in the percentage of land use group acreages (Table 4) and as changes in the percentage distributions of land use categories within each land use group (Table 5). The "nursery" and "extractive" land use categories in Table 4 were not present in the Moore's Creek (Alb) watershed, so they do not appear in subsequent tables.

Table 4. CBWM Albemarle segment: Percentage change in land use group acreage between baseline and target scenarios.

Land Use Group	Baseline Scenario (acres)	Target Scenario (acres)	Change as % of Total Area
Conventional tillage, no manure	294.8	259.2	–0.037
Other row crops	107.0	100.1	0.007
Pasture	9,044.3	7,611.3	–1.486
Hay	5,899.2	6,231.5	0.345
Forest	68,717.9	70,069.2	1.401
Impervious developed	2,624.0	2,427.2	–0.204
Pervious developed	8,640.2	8,837.0	0.204
Extractive	219.4	11.2	0.216
Nursery	15.8	15.8	0.000
Water	870.7	870.7	0.000
Total area	96,433.2	96,433.2	

Notes: 1 acre ≈ 0.4046 ha; CBWM, Chesapeake Bay Watershed Model.

Table 5. CBWM Albemarle segment: Percentage change in land use category acreage within each group between baseline and target scenarios.

Land Use Group	Land Use Categories in Each Group	Baseline Scenario (% of Group)	Target Scenario (% of Group)	Change as % of Baseline
Conventional tillage, no manure	hom	95.9	0.0	–100.0
	nho	4.1	100.0	2,339.0
Other row crops	hwm	46.6	0.0	–100.0
	nhi	2.0	10.0	402.0
	lwm	49.3	0.0	–100.0
	nlo	2.1	90.0	4,169.8
Pasture	pas	93.3	87.5	–6.3
	npa	4.0	12.2	208.2
	trp	2.7	0.3	–88.8
	afo	0.0	0.0	—
Hay	hyw	72.3	0.0	–100.0
	nhy	3.1	62.7	1,929.7
	alf	2.1	0.0	–100.0
	nal	0.1	1.8	1,929.7
	hyo	22.5	35.5	57.9
Forest	for	99.0	99.0	0.0
	hvf	1.0	1.0	–1.9
Impervious developed	cid	0.0	0.0	—
	rid	29.2	29.2	0.0
	nid	70.8	70.8	0.0
Pervious developed	cpd	0.0	0.0	—
	rpd	43.5	43.2	–0.7
	npd	54.5	54.9	—
	ccn	0.0	0.0	—
	rcn	1.9	1.9	–2.2

Note: CBWM, Chesapeake Bay Watershed Model.

The large percentage increases for several land uses in Table 5 result from the application of nutrient management (NM) control measures to agricultural land uses. The use of such measures leads to large shifts of area from a land use without NM, such as "high-till without manure" (hom) to its counterpart with NM, "high-till without manure NM" (nho). The change percentages are especially large where the initial baseline group percentages were very small. Table 6 shows the resulting land use distributions for both the baseline and target scenarios.

Table 6. Moore's Creek (Alb) watershed: Summary of CBWM land use distributions between baseline and target scenarios.

CBWM Land Use Code	CBWM Land Use Category	Baseline Scenario (acres)	Target Scenario (acres)
hom	High-till without manure	58.1	0.0
nho	High-till without manure NM	2.5	60.1
hwm	High-till with manure	4.8	0.0
nhi	High-till with manure NM	0.2	1.0
lwm	Low-till with manure	5.1	0.0
nlo	Low-till with manure NM	0.2	9.2
hyw	Hay with nutrients	564.7	0.0
nhy	Hay with nutrients NM	24.1	487.4
alf	Alfalfa	16.4	0.0
nal	Alfalfa NM	0.7	14.1
hyo	Hay without nutrients	175.6	275.9
pas	Pasture	185.4	169.4
npa	Pasture NM	7.9	23.7
trp	Pasture corridor	5.4	0.6
afo	Animal feeding operation	8.8	8.8
for	Forest	12,951.2	13,004.5
hvf	Harvested forest	130.6	128.5
cid	CSS impervious developed	0.0	0.0
rid	Regulated impervious developed	304.9	301.7
nid	Nonregulated impervious developed	739.7	731.8
cpd	CSS pervious developed	0.0	0.0
rpd	Regulated pervious developed	1,980.9	1,953.1
npd	Nonregulated pervious developed	2,481.1	2,480.8
ccn	CSS construction	0.0	0.0
rcn	Regulated construction	87.5	85.0
atdep	Atmospheric deposition	227.7	227.7
Total		19,963.4	19,963.4

Notes: 1 acre ≈ 0.4046 ha; CBWM, Chesapeake Bay Watershed Model; CSS, combined sewer system.

Step 5. Obtain Model Load Data and Calculate Unit-Area Loads

In this step, one obtains annual loads (in tons per year) corresponding to each land use category for the appropriate model segment and calculates UALs by dividing the loads by the corresponding acreage for each land use category.

For application in the Moore's Creek watershed, we obtained UALs by dividing CBWM-simulated average annual load data, corresponding to the model segments that included Moore's Creek, by their respective areas for each applicable land use category. Table 7 shows an example of the data used for the baseline scenario in the CBWM Albemarle segment. We used similar data and calculations from simulated output for the target scenario.

Table 7. CBWM Albemarle segment: Baseline scenario areas, loads, and unit-area loads.

CBWM Land Use Code	CBWM Land Use Category	Area (acres)	TSS (tons/year)	TSS UAL (tons/acre/year)
hom	High-till without manure	282.7	38.3	0.14
nho	High-till without manure NM	12.1	1.7	0.14
hwm	High-till with manure	49.9	5.5	0.11
nhi	High-till with manure NM	2.1	0.2	0.11
lwm	Low-till with manure	52.7	3.6	0.07
nlo	Low-till with manure NM	2.3	0.2	0.07
hyw	Hay with nutrients	4,262.4	165.4	0.04
nhy	Hay with nutrients NM	182.2	7.1	0.04
alf	Alfalfa	123.5	4.8	0.04
nal	Alfalfa NM	5.3	0.2	0.04
hyo	Hay without nutrients	1,325.8	50.4	0.04
pas	Pasture	8,400.3	7,991.5	0.95
npa	Pasture NM	359.1	346.0	0.96
trp	Pasture corridor	245.9	2,917.2	11.86
afo	Animal feeding operation	39.0	120.2	3.08
cfo	Confined animal feeding operation	0.0	0.0	—
for	Forest	68,032.1	2,203.9	0.03
hvf	Harvested forest	685.8	136.8	0.20
cid	CSS impervious developed	0.0	0.0	—
rid	Regulated impervious developed	766.0	618.5	0.81
nid	Nonregulated impervious developed	1,858.0	1,500.3	0.81
cpd	CSS pervious developed	0.0	0.0	
rpd	Regulated pervious developed	3,762.0	482.7	0.13
npd	Nonregulated pervious developed	4,712.0	604.6	0.13
ccn	CSS construction	0.0	0.0	—
rcn	Regulated construction	166.2	389.8	2.35
cex	CSS extractive	0.0	0.0	—
rex	Regulated extractive	0.0	0.0	—
nex	Nonregulated extractive	219.4	716.5	3.27
urs	Nursery	15.8	67.9	4.30
atdep	Atmospheric deposition	870.7	0.0	0.00

Notes: 1 acre ≈ 0.4046 ha; 1 ton ≈ 0.9072 metric tons; 1 ton/acre/year ≈ 2.2422 metric tons/ha/year; CBWM, Chesapeake Bay Watershed Model; CSS, combined sewer system; TSS, total suspended sediment. Land uses without UAL values were not represented in the Albemarle segment.

Step 6. Calculate Local Subwatershed Pollutant Loads

In this step, one calculates local subwatershed pollutant loads by multiplying the redistributed land use category areas for each scenario by their corresponding UALs. Table 8 illustrates the UAL calculations for the Moore's Creek (Alb) baseline scenario.

Table 8. Moore's Creek (Alb) watershed: Local sediment loads calculated from CBWM unit-area loads and redistributed areas.

CBWM Land Use Code	CBWM Land Use Category	Redistributed Area (acres)	CBWM UAL (tons/acre/year)	Total Suspended Sediment (tons/year)
hom	High-till without manure	58.1	0.14	7.9
nho	High-till without manure NM	2.5	0.14	0.3
hwm	High-till with manure	4.8	0.11	0.5
nhi	High-till with manure NM	0.2	0.11	0.0
lwm	Low-till with manure	5.1	0.07	0.3
nlo	Low-till with manure NM	0.2	0.07	0.0
hyw	Hay with nutrients	564.7	0.04	21.9
nhy	Hay with nutrients NM	24.1	0.04	0.9
alf	Alfalfa	16.4	0.04	0.6
nal	Alfalfa NM	0.7	0.04	0.0
hyo	Hay without nutrients	175.6	0.04	6.7
pas	Pasture	189.5	0.95	180.3
npa	Pasture NM	8.1	0.96	7.8
trp	Pasture corridor	5.5	11.86	65.5
afo	Animal feeding operation	4.4	3.08	13.6
for	Forest	12,951.2	0.03	419.6
hvf	Harvested forest	130.6	0.20	26.0
rid	Regulated impervious developed	304.9	0.81	246.2
nid	Nonregulated impervious developed	739.7	0.81	597.3
rpd	Regulated pervious developed	1,980.9	0.13	254.2
npd	Nonregulated pervious developed	2,481.1	0.13	318.3
rcn	Regulated construction	87.5	2.35	205.2
		19,963.4		2,373.3

Notes: 1 acre ≈ 0.4046 ha; 1 ton ≈ 0.9072 metric tons; 1 ton/acre/year ≈ 2.2422 metric tons/ha/year; Chesapeake Bay Watershed Model; CSS, combined sewer system.

Step 7. Compare Baseline and Target Scenario Pollutant Loads

In the example presented here, we developed the Moore's Creek TMDL for sediment. Sediment fate and transport are simulated similarly for many of the 31 land-based land use categories used in the CBWM. We aggregated the land use categories reported in Table 9 across those land use categories for which the sediment simulation was the same (e.g., we aggregated the various hay land use categories into the "hay" land use category and aggregated the pasture and pasture NM categories into the "pasture" land use category). Additionally, we consolidated urban land use categories into the "pervious developed," "impervious developed," and "construction" categories. Table 9 illustrates the simulated sediment loads (tons per year) for the Moore's Creek (Alb) baseline and target scenarios.

Table 9. Moore's Creek (Alb) watershed: Comparison of baseline and target scenario sediment loads.

CBWM Land Use Category	Baseline Scenario		Target Scenario	
	Area (acres)	Total Suspended Sediment (tons/year)	Area (acres)	Total Suspended Sediment (tons/year)
Conventional tillage, no manure	60.6	8.2	60.1	6.3
High-till cropland	5.0	0.5	1.0	0.1
Low-till cropland	5.3	0.4	9.2	0.6
Hay	781.5	30.2	777.4	26.7
Pasture, other	193.3	183.9	193.2	126.0
Pasture corridor	5.4	64.1	0.6	7.0
Animal feeding operation	8.8	27.2	8.8	16.8
Forest	12,951.2	419.6	13,164.3	421.3
Harvested forest	130.6	26.0	130.1	23.0
Impervious developed	1,044.6	843.5	1,630.9	692.0
Pervious developed	4,462.0	572.5	5,561.9	472.0
Construction	87.5	205.2	90.6	199.3
Average annual sediment load		2,381.3		1,991.1

Notes: 1 acre ≈ 0.4046 ha; 1 ton ≈ 0.9072 metric tons; CBWM, Chesapeake Bay Watershed Model.

Based on the target scenario load, the long-term target average annual sediment load for Moore's Creek (Alb) watershed is about 2,000 tons/year (1,814,360 kg/year). The comparison between the baseline and target scenarios indicates that a sediment load reduction of 16.4% is needed to achieve restoration conditions. The load reductions are effected through the simulation of management practices that take the form of both land use changes (reflected in land use category area changes) and load reductions (reflected in UAL changes). In the actual Moore's Creek sediment TMDL, Yagow et al. (2011) calculated land-based loads for both the Albemarle County and City of Charlottesville portions of Moore's Creek watershed and also included point source loads.

The disaggregate method promotes consistency between TMDLs developed for localized impairments and those required ...by basin-scale modeling.

Summary

We developed the disaggregate method to leverage output from an existing, publicly available, basin-scale model to assist in developing spatially consistent TMDL loads for upstream subwatersheds in the Chesapeake Bay watershed. In general, however, one could apply this method in any area for which publicly available, basin-scale modeling has been performed and more detail is desired in a particular subwatershed. TMDL development in smaller, upstream subwatersheds is one general application in which one can use the disaggregate method. In the Moore's Creek example, this method allowed for refinements to the land use distributions in the CBWM by incorporating locally available land use data. Although we used sediment in the Moore's Creek example, a similar procedure could be used for any land-based pollutant simulated by an existing basin-scale model. The disaggregate method promotes consistency between TMDLs developed for localized impairments and those required to meet downstream target pollutant loads established by basin-scale modeling. In addition, it provides an alternative to the reference watershed approach for quantifying target loads for pollutants without numeric water quality standard criteria.

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Local Monitoring Data Used To Support Watershed-Based Hydrologic Modeling of Downscaled Climate Model Output

Maintaining a safe and abundant water supply is a primary concern of water system managers. The traditional approach for maintaining an adequate water supply has been to design the system based on the worst-case scenario from the area's recorded history. This approach has worked well in the modern era of development. However, given our current uncertainty about the effects of climate change and of the continued land use changes brought about by an ever-expanding population, it is not clear how much longer we can continue to rely on the historical hydrologic record to inform future water supply.

…it is not clear how much longer we can continue to rely on the historical record to inform future water supply.

Researchers at Virginia Tech's Department of Civil and Environmental Engineering are studying the effects of climate and land use change on water resources within the Occoquan Reservoir watershed. The Occoquan watershed, which encompasses 1,500 km^2 in northern Virginia, lies in the Middle Potomac River basin (Figure 1).

The Occoquan Watershed Monitoring Laboratory (OWML) at Virginia Tech has been collecting data within the watershed for almost 40 years. This study used local weather station data from OWML and from the National Climatic Data Center to calibrate the hydrologic model and to downscale climate model output. The downscaling methodology employed in this study uses statistical relationships to incorporate the climate signal generated by global-scale climate models while maintaining the observed local-scale weather phenomena. Drawing from three downscaled climate model projections, we used an ensemble average output of modeled local streamflow for the past century (1901–2000) and this century (2001–2100) to make a comparative (historical to future) analysis of water supply impacts from both climate and land use change. The three climate models chosen to make up the ensemble were (1) the Parallel Climate Model (NCPCM), developed in the United States by the National Center for Atmospheric Research, the Department of Energy, the National Aeronautics and Space Administration, and the National Oceanic and Atmospheric Administration; (2) the Model for Interdisciplinary Research on Climate (MIMR), developed in Japan by the Center for Climate System Research, University of Tokyo, the National Institute for Environmental Studies, and the Frontier Research Center for Global Change, Japan Agency for Marine-Earth Science and Technology; and (3) the ECHAM5 climate model, developed in Germany by the Max Planck Institute for Meteorology. In this study area, the NCPCM showed the greatest increase in precipitation, the MIMR showed the greatest decrease in precipitation, and the ECHAM5 modeled streamflow with the closest agreement to streamflow in the observed years 1981–2000.

This study used land use data assembled by the OWML from the Northern Virginia Regional Commission (from the years 1977, 1979, 1984, 1989, 1995, 2000, and 2006). Land use patterns in the Occoquan watershed from 1977 through 2006 show a steady increase in urban area with equivalent decreases in agricultural and forested land. The western half of the watershed is less urbanized, with large areas of agricultural lands, and the eastern half is predominantly urbanized, with small areas of agricultural lands. This study assumes that the trend of increased suburban and urban development will continue during this century, forming the basis of future land use projections.

This study evaluated the effects of climate and land use change on local streamflow using the Hydrological

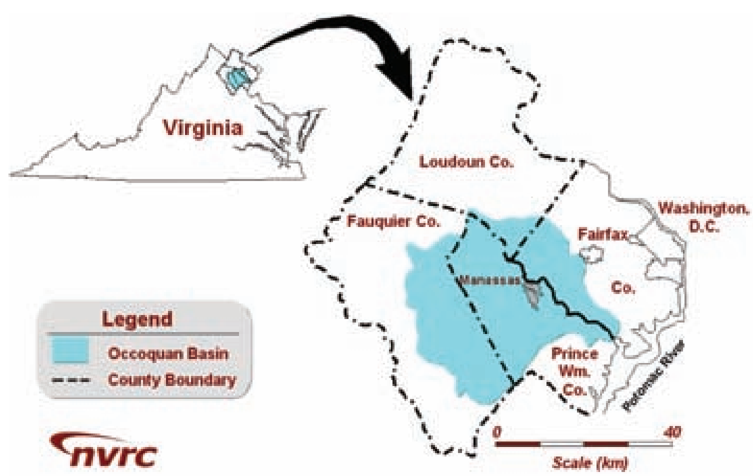

Figure 1. Location map of the Occoquan watershed. Source: Northern Virginia Regional Commission, "Occoquan," http://www.novaregion.org/index.aspx?NID=410 (used with permission).

Simulation Program–FORTRAN (HSPF) watershed model. Researchers modeled three scenarios of future change to discern the effects of climate change (S1), land use/demand change (S2), and joint climate and land use/demand change (S3). We used these scenarios, along with multiple analysis metrics, to focus the assessment on low flows (drought conditions) within the watershed, which are of primary concern to water supply managers. In the initial results, the influence of reclaimed water return flows upstream of the Occoquan Reservoir on the drought sensitivity of the system stood out as an important variable to analyze. By modeling the system without accounting for expansion in the reclaimed water inflows, a clear picture of the growing reliance on this source became apparent within this century (2001–2100). This vignette is a summary of a more detailed analysis presented by Philip Maldonado (see List of Sources).

The ensemble analysis of three HSPF-modeled climate outputs, projected through the end of this century (2001–2100), produced the following outcomes:

- Scenario S1 (climate change alone) showed an increase in the winter and spring low flows through the end of this century. The summer and fall low flows decreased slightly compared to the historical ensemble model.
- Scenario S2 (land use/demand change alone) modeled the future hydrology based on the past century's climate (1901–2000) while using three projections of future land use, to the years 2040, 2070, and 2100. Each of the three future models showed increases in low flows for all seasons compared to both Scenario S1 and the historical ensemble model.
- Scenario S3 (joint climate and land use effects) modeled the future hydrology based on the downscaled projection of future climate (2001–2100) along with the three projections of future land use. As in Scenario S2, each of the three future models showed increases in low flows compared to Scenario S1 and the historical ensemble model. However, when compared to Scenario S2, the winter and spring low flows showed a larger increase, whereas the low-flow statistics for summer and fall showed a slight decrease.

To assess the importance of the reclaimed water supply stream, we repeated Scenarios S2 and S3 without accounting for expansion in reclaimed water return flows. The analysis employed a metric that incorporated the storage response curves used by operators to manage the reservoir. The response curves indicated a failure of the reservoir given a repeat of the drought of record, as influenced by climate change, and given the land use/demand projections for the latter part of this century (land use/demand years 2070 and 2100).

This study reinforces the established scientific contention that anthropogenic climate and land use change are likely to affect the timing and magnitude of runoff, changing the behavior of reservoirs managed for water supply. These changes are dampened by the expansion of reclaimed water inflows in the Occoquan watershed. The total volume of water increased for all scenarios, but when analyzed on a seasonal basis, the majority of increases occurred in the winter and spring, whereas the summer and fall showed little to no increase. With reclaimed water expansion, these increases maintained a supply sufficient to meet expanding demands; but without reclaimed expansion, demands outpaced supplies in the latter part of this century (land use/demand years 2070 and 2100). The increases in the Scenario S2 model projections beyond those of Scenario S1 show that increases in runoff due to increased imperviousness (land use change) are large relative to those due to climate change when modeled through the end of this century. Based on this study, increased flows and drought resistance appear very likely to occur in the Occoquan Reservoir system.

Because of the uncertainty in these models and the statistical methods used, one should interpret these results only as an indication of the sensitivity of the current system to predicted future climate, not as a prediction of the specific effects of climate change. Regardless, place-specific studies such as this one are crucial for future water supply plans as our climate and landscape continue to evolve. As this study shows, the current management practices of the Occoquan watershed are well-positioned to absorb the impact from both climate and land use change. This type of study is possible only through a reliable and well-maintained watershed monitoring system, which forms the basis of a well-developed management plan. Such studies are useful tools through which planners and managers can gain insight into unforeseen impacts from future development and quantify system vulnerabilities to previously unassessed risks.

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Locally Derived Water Balance Method To Evaluate Realistic Outcomes for Runoff Reduction in St. Louis, Missouri

Introduction

The Metropolitan St. Louis Sewer District (MSD) is the coordinating authority of a 61-permittee Phase II municipal separate storm sewer system (MS4) permit. MSD is carefully following the development of new national postconstruction stormwater regulations, which focus on maintaining or restoring the runoff component of the undeveloped (i.e., natural) water balance. If the Energy Independence and Security Act (EISA) Section 438 technical guidance is the "writing on the wall" for a national rule, then development projects would be required to implement postconstruction controls that capture and retain on-site (i.e., no discharge) the 95th percentile daily rainfall depth (3.8 cm in St. Louis).

Figure 1. Example of naturally vegetated Missouri prairie and sinkhole pond.

Stormwater professionals may question whether a rule like this would be appropriate nationwide. MSD developed a water balance model to evaluate the potential runoff reduction that may be achieved in local watersheds in response to the targeted EISA rule. The predevelopment water balance in the St. Louis region has not previously been studied for this purpose. This vignette presents a "simple" approach to developing an annual estimate of runoff, and one that may be a useful tool for other stormwater managers whose watersheds' predevelopment hydrology has not been assessed.

Methods

The water balance is the balance between the input of water from precipitation and the output of water by runoff, evapotranspiration, storage, and infiltration. Numerically, the runoff component of the water balance is expressed as $R = P - ET - N - \Delta S$, where R is runoff, P is precipitation, ET is evapotranspiration, N is infiltration or recharge, and ΔS is the change in storage (in soil).

The one-dimensional Thornthwaite method is used to estimate components of the water balance on a daily time-step. MSD used a modified version of this method, as described below.

Climate, Evapotranspiration, and Vegetation

MSD obtained 21 years of daily weather data from the National Weather Service[1] for Lambert St. Louis Airport for the period January 1989 to December 2009. We calculated daily potential evapotranspiration rates according to the American Society of Civil Engineers (ASCE) standardized reference evapotranspiration equation, thus replacing the Thornthwaite evapotranspiration rates with the ASCE rates. We then multiplied daily reference evapotranspiration rates by the landscape coefficient for a grass prairie (0.5), a reasonable approximation of an undeveloped, naturally vegetated condition in Metropolitan St. Louis and much of Missouri (see Figure 1). This prairie landscape coefficient is consistent with the US Geological Survey (USGS) rain garden report, *Evaluation of Turf-Grass and Prairie-Vegetated Rain Gardens in Clay and Sand Soil, Madison, Wisconsin, Water Years 2004–2008*, which estimates the landscape coefficient for a prairie-planted rain garden area to range from 0.2 to 0.7.

Infiltration (Recharge)

The near-surface geology of much of St. Louis City and County consists of urbanized (e.g., cut, filled, and reworked) clayey silt soil over limestone bedrock. The thickness of urbanized fill over bedrock varies greatly. MSD used results for Southwest Missouri from the USGS report, *Groundwater-Flow Model and Effects of Projected Groundwater Use in the Ozark Plateaus Aquifer System in the Vicinity of Greene County, Missouri—1907–2030*, to estimate groundwater recharge as only limited research and modeling of groundwater has been conducted for Metropolitan St. Louis. The surficial geologic conditions (clay or silt soil over limestone bedrock) in Southwest Missouri and St. Louis are similar in many ways.

[1] National Oceanic and Atmospheric Administration's National Weather Service, "NHDS Access of Historical Data," http://amazon.nws.noaa.gov/hdsb/data/archived/index.html.

The USGS groundwater report estimated recharge to be an average of 2.5% of annual precipitation. Thus, only a limited amount of precipitation can result in deep infiltration.

Soil Storage

The maximum available water storage is the product of the soil's porosity (saturation) and the thickness of the root zone. When the maximum available water storage is exceeded, runoff occurs (if the precipitation is not frozen). The minimum available water storage is the product of the wilting point and the thickness of the root zone. The values MSD used in calculations were representative of silt loam. The root zone thickness used for the prairie condition was 1.5 m; this is consistent with observations reported in the USGS rain garden report.

Model Limitations

This modified Thornthwaite water model has a number of limitations. First, the model does not account for rainfall intensity; thus, where the intensity of the storm exceeds the infiltration rate of the soil, runoff is underestimated. Second, the model assumes that runoff occurs on the same day as precipitation. This assumption is supported by recent work by Debusk and colleagues, who showed that, in an undeveloped watershed with clayey soils, nearly all precipitation (even interflow) is discharged within 18 hours after runoff begins. Third, this model assumes that all snowmelt runoff occurs on the first day on which the air temperature is above freezing. This assumption makes little difference for annual or seasonal water balance comparisons because snow melts during a time of year when soil is typically saturated and evapotranspiration rates are low. Finally, because the model is one-dimensional, calculations do not differentiate between runoff as interflow or overland flow.

Results and Discussion

Tables 1 and 2 summarize the results. The total average annual precipitation was ~100 cm; of this, 42% resulted in runoff, primarily between January and July.

Table 1. Summary of water balance conditions.

Component	Annual Quantity (cm)	Percentage of Annual Precipitation
Evapotranspiration	55	55
Deep Infiltration	2.5	2.5
Runoff	42	42

Table 2. Summary of runoff (discharge) conditions.

Time Period	Annual Avg. Runoff (cm)	Runoff as % of Annual Precipitation	Runoff as % of Quarterly Precipitation
Total	42	42	
January–March	12	12	60
April–June	16	16	50
July–September	5	5	19
October–December	9	9	40

Forthcoming nationwide stormwater regulations may mandate that runoff from a developed site should amount to only 5% of annual rainfall. However, this study shows that runoff accounts for a much greater percentage of annual rainfall (42%) and is a natural process in undeveloped, naturally vegetated conditions in St. Louis, Missouri.

By illustrating that runoff (discharge) is a major component of the water balance in undeveloped, natural conditions, this analysis suggests a shortcoming to a nationwide retention rule applied to local watersheds. During summer, rainfall is absorbed into the soil and then removed through evaporation and transpiration. Because evapotranspiration rates are highest during summer months, much of the soil's water-holding capacity is available to absorb precipitation through early fall. However, after rainfall occurs in late fall, soil becomes saturated. Snow that accumulates over already saturated soil results in mid-winter snowmelt runoff. Rainfall in late winter and early spring, even small events, results in runoff. In this model, about 67% of the annual runoff occurred from precipitation events with rainfall depths less than the 95th percentile daily rainfall. Requiring retention of all storms less than the 95th percentile daily rainfall is not a surrogate for water balance restoration.

Conclusions

Attempts to mimic the runoff conditions of an undeveloped, naturally vegetated site can be affected by many factors, especially the available water storage capacity of the site's soil. Available water capacity is affected by weather, geology, soil type, vegetation, and evapotranspiration.

A clear definition of postconstruction best management practice performance goals is needed. However, requiring retention of all storms up to the 95th percentile daily rainfall is difficult to justify in St. Louis—and in much of Missouri—and is potentially counterproductive to the improvement of water quality. Instead, a balanced performance goal composed of some infiltration and some attenuated discharge would better approximate a natural condition.

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Contributor

This vignette was prepared by Jay Hoskins, PE, Metropolitan St. Louis Sewer District.

HAVE A QUESTION YOU'D LIKE US TO ASK OUR EXPERTS? *The upcoming Fall 2012 issue will focus on watershed planning and address how well the watershed-based approach is working, how many plans have been developed and recommendations implemented, and recent innovations in watershed planning and implementation. AWSPs members and Bulletin subscribers may email their questions to bulletin@awsps.org. The Bulletin features interviews with experts in the watershed and stormwater professions to discuss the topic of each issue. In this issue, four professionals weigh in with their perspectives on the use of models and monitoring to inform the decision-making process for the protection or improvement of watershed health. Here is what our experts had to say….*

Nicholas A. DiPasquale

Director, Chesapeake Bay Program Office, US Environmental Protection Agency

Nicholas DiPasquale is the director of the Chesapeake Bay Program Office (CBPO), a unique regional partnership program that has coordinated the restoration of the Chesapeake Bay watershed since 1983. As the lead federal agency, the US Environmental Protection Agency (USEPA) coordinates activities and implements strategies for meeting the restoration goals of the Chesapeake Bay watershed.

Nick has nearly 30 years of public policy and environmental management experience in the public and private sectors. He previously served as deputy secretary in the Pennsylvania Department of Environmental Protection; director of the Environmental Management Center for the Brandywine Conservancy in Chadds Ford, Pennsylvania; and secretary of the Delaware Department of Natural Resources and Environmental Control. Most recently, he served as a senior consultant on environmental and ecological restoration issues with Duffield Associates, an environmental engineering consulting firm located in Wilmington, Delaware.

Q: How have you been involved in watershed-based monitoring or modeling?

A: I work in the Chesapeake Bay and the surrounding six-state, 64,000-square-mile watershed that provides a home to more than 17 million Mid-Atlantic residents—from Oswego, New York, south to Norfolk, Virginia, and stretching from Moorefield, West Virginia, to Laurel, Delaware. The Bay has 11,684 miles of shoreline, supports 3,600 species of plants, fish, and wildlife, and produces 500 million pounds of seafood per year. Every watershed resident lives just a few minutes from one of the more than 100,000 streams and rivers that drain into the Bay.

For the past 27 years, CBPO has provided significant USEPA funds supporting the continued operation of the tidal and watershed water quality monitoring networks by the Chesapeake Bay Program partnership's states (NY, PA, WV, MD, DE, VA, and DC), universities, river basin commissions, and federal agencies (the US Geological Survey [USGS], US Army Corps of Engineers, and National Oceanic and Atmospheric Administration). In FY2011, the partnership invested over $4 million to fund these monitoring networks. Over the same time frame, we funded an entire team of federal agency and university staff to provide network coordination, quality assurance oversight, shared data management, and data analysis and interpretation support to the partnership's monitoring networks.

Q: In this issue, we focus on the use of models and monitoring to assess watersheds and how these tools are used to inform the decision-making process. How do you see monitoring and modeling applied in watershed management?

A: Since the Chesapeake Bay Program's formation in 1983, we have actively applied results from our shared monitoring networks and suite of models to make Bay and watershed restoration decisions. An example of this integrated use of monitoring and modeling is the precedent-setting 2010 USEPA Chesapeake Bay total maximum daily load (TMDL) that establishes source sector allocations for 92 separate impaired tidal water segments. These segment-specific allocations would not have been possible without the extensive record of monitoring data, used directly to help derive Bay-specific water quality criteria and standards and to calibrate the suite of models over a 21-year period.

We also use monitoring data to provide a direct understanding of current status and long-term trends in stream water quality. Our suite of models provide managers with the opportunity to ask "what if" questions concerning nutrient reduction practices and technologies and likely instream water quality responses.

Based on an independent review of the partnership's monitoring networks, we are investing $2 million in 2011 and 2012 to build the watershed monitoring network up to 120 stations across the six states. We are turning our coordination support, quality assurance expertise, and funding resources toward the expansion of the watershed monitoring networks that bring more nontraditional partners—watershed organizations, riverkeepers, counties, permitted dischargers—into the water monitoring network.

Q: Stream restoration is an option considered for sediment and nutrient load reduction. What modeling and monitoring is needed for stream restoration projects? How can these results be applied to watershed and stormwater management to credit this as a practice through which to reduce pollutant loads?

A: The Chesapeake Bay Program partnership is currently working with the Bay region's scientific community to develop a more robust set of nitrogen, phosphorus, and sediment load reduction efficiencies for various stream restoration practices and techniques. These efficiencies are integral to the use of models in support of planning for implementation of the most effective practices and of tracking and crediting the effectiveness of stream restoration. We are also working with the growing network of funders of stream restoration projects to invest in monitoring of pre- and postrestoration actions.

Q: Most TMDLs are developed using a combination of (simple to complex) monitoring and modeling, and all are associated with some level of uncertainty. What recommendations do you have for reducing this uncertainty and improving the accuracy of TMDL allocations using monitoring or modeling? How can local jurisdictions better address the urban versus agricultural source load allocations using monitoring or modeling?

A: One of the best ways to help reduce uncertainty and improve decision making on source allocations is to improve the verification, tracking, and reporting of the full suite of implemented nutrient and sediment reduction practices and technologies. A more comprehensive accounting of practices implemented on land will help improve watershed model calibration, thereby reducing uncertainty. In parallel, the better the accounting of the practices in place that are directed toward reducing specific pollutant source sectors, the better local jurisdictions will be positioned to assess what further reduction actions are possible and necessary for each source sector. In addition, uncertainty is minimized when an organization is willing to continually learn from its actions, such that a process is set up to manage adaptively by taking actions, testing assumptions, monitoring success, and adjusting actions where necessary.

Q: Can you share a "success story" where monitoring or modeling efforts have helped improve understanding or further watershed protection efforts? If so, who was involved (e.g., organizations, volunteers, or researchers)?

A: The seven jurisdictions' development of their Phase I Watershed Implementation Plans, which USEPA used in establishing the Bay TMDL allocations, were based on an application of the partnership's full suite of monitoring networks and modeling tools. These plans are now being refined to define the load reduction and implementation responsibilities of the multitude of local jurisdictions across the six-state watershed. These collective actions are resulting in local partners and communities taking ownership of their part of achieving the larger Bay pollution diet.

Q: Based on your experience with monitoring or modeling, what research or other work (e.g., coordination or programs) is still needed for the effective watershed management application of monitoring and modeling?

A: In the case of monitoring, the steps still needed include the adoption of quantifiable and measurable environmental restoration goals, the designation of a strong coordination body with decision-making responsibilities on shared network funding, and the adoption of a process for independent reviews of the monitoring network to ensure that data collection efforts are relevant and meet the needs of managers. In the case of modeling, the collection of data critical for model calibration and model operation at the scale of local decision making is the most important need.

Useful Resources

- For access to information on the Chesapeake Bay Program partnership's monitoring networks and the suite of models, please access the partnership's website at www.chesapeakebay.net.
- Detailed information on the Chesapeake Bay TMDL is available at www.epa.gov/chesapeakebaytmdl.

Lawrence E. Band, PhD

Voit Gilmore Distinguished Professor of Geography and Director, Institute for the Environment, University of North Carolina

Lawrence (Larry) Band is the Voit Gilmore Distinguished Professor of Geography and the director of the Institute for the Environment at the University of North Carolina (UNC), Chapel Hill. At UNC since 1998, he teaches and conducts research in watershed hydrology, geomorphology, geographic information systems, and environmental modeling. His current research focuses on two Long Term Ecological Research (LTER) sites: the Coweeta LTER in western North Carolina and the Baltimore Ecosystem Study (BES). He has recently developed collaborative research projects with colleagues at North Carolina State University and Duke University as part of the National Science Foundation (NSF)–funded Triangle ULTRA-EX (Urban Long Term Research Area–Exploratory) project in the North Carolina Triangle. In 2010, he was chair of the Board of Directors for the Consortium of Universities for the Advancement of Hydrologic Science, a consortium of about 130 US and foreign universities, nonprofit institutes, and domestic and foreign water science and management agencies (www.cuahsi.org).

Q: How have you been involved in watershed-based monitoring or modeling?

A: My research group is involved in monitoring and modeling of water, carbon, and nutrient cycling and export in watersheds, including urban, agricultural, and forested sites. We have worked in watersheds in different parts of the country and globally, but our current focus is on collaborative projects as part of NSF's LTER network, working in well-instrumented catchments in Baltimore City and Baltimore County, Maryland, as well as western and central North Carolina. The Baltimore and central North Carolina sites include heavily urbanized-to-rural gradients, while the western North Carolina site is a set of experimental forested catchments and catchments in largely rural areas.

Q: In this issue, we focus on the use of models and monitoring to assess watersheds and how these tools are used to inform the decision-making process. How do you see monitoring and modeling applied in watershed management?

A: Watershed monitoring and modeling should proceed hand-in-hand to provide information on the where, how, and when of coupled water, carbon, and nutrient cycling and export from catchments to downstream water bodies. We need to consider where nutrients are introduced, transformed, and transported in the watershed to help better identify places for treatment or mitigation. With this more integrated view of the watershed, we would be able to design more efficient strategies for pollution reduction.

The data generated from monitoring and modeling output can better serve the decision-making process if we adopt a true adaptive management approach. Unanticipated changes can occur, and incomplete knowledge of watershed systems means that planned restorations should be stated as hypotheses and tested. TMDLs are a good example. That is, once the initial modeling and monitoring are completed, implementation of the TMDL should include ongoing programs. I have participated in reviews for a number of TMDLs and planned restorations, including the Chesapeake Bay, Cape Cod, and the Florida Everglades. In all cases, multiyear or decadal efforts can be adaptive; such an approach will allow decisions to evolve with current events and new data.

Q: Indicators for watershed health focus predominantly on surface water chemistry. To what extent does or should monitoring and modeling focus on other indicators?

A: Chemistry is one piece of our information pipeline, but not the only stressor. We should be looking at watershed flow regimes, channel and riparian characteristics, the connectivity of the channel with upland sources, and groundwater circulation as indicators of potential source or retention sites for nutrients. Biological indicators are good overall indicators of ecosystem health, but multiple stressors can result in the same biological impairment. In general, one should not rely on only a single indicator when it is necessary to determine causes of impairment; so a suite of flow, chemistry, geomorphic, and biological measures should be planned.

Q: Stream restoration is an option considered for sediment and nutrient load reduction. What modeling and monitoring is needed for stream restoration projects? How can these results be applied to watershed and stormwater management to credit this as a practice through which to reduce pollutant loads?

A: Stream restoration is often a localized manipulation of the channel and riparian zone with insufficient thought as to the stream's connection to the watershed. Restoration efforts need to consider the full watershed and design channel conditions compatible with upstream inputs; otherwise prerestoration conditions have a tendency to reestablish.

To provide crediting within a reasonable level of certainty, prerestoration sampling should characterize prior conditions, and follow-up sampling should establish the restoration impact. This also requires a control site without restoration to enable one to distinguish between temporal changes and the actual stream treatment. This is often overlooked due to timing or cost, but the cost of monitoring is a tiny fraction of the actual restoration. There is a need to collect information and learn from the activity by synthesizing, managing, and maintaining a library of documented restorations.

Q: What programs or assistance are available to advance the application of monitoring and modeling for watershed protection? Which assistance needs are being met and which types of assistance are still unavailable? Is the information about available assistance getting to the practitioners?

A: This is a current shortcoming. Watersheds are often managed independently within jurisdictions, rather than as integrated hydrologic and ecological systems. Collaboration across jurisdictions is beginning to emerge, but needs more formal guidance and governance to promote more effective and coordinated full watershed management. Currently, data sharing and management are variable or nonexistent. One potential source of help, in addition to state extension and USEPA programs, are data services from the Consortium of Universities for the Advancement of Hydrologic Science, Inc. This community organization developed advanced water data management and is providing data management and data publication services as well as the ability to search USGS, USEPA, and additional data archives based on location and keyword search. Currently, a set of water management groups are beginning to use these services, but more needs to be done to reach practitioners and improve the service.

Q: Can you share a "success story" where monitoring or modeling efforts have helped improve understanding or further watershed protection efforts? If so, who was involved (e.g., organizations, volunteers, or researchers)?

A: It is important to show how well a best management practice (BMP) works. To do this, pre- and post-BMP monitoring data are needed. Long-term data sets to show trends are very valuable, but rare. For example, through the BES LTER, stream sampling pre- and post-sanitary sewer retrofitting by the City of Baltimore showed clear improvements in water quality. However, the barriers of cost and sampling inconvenience need to decline significantly to expand long-term monitoring sites.

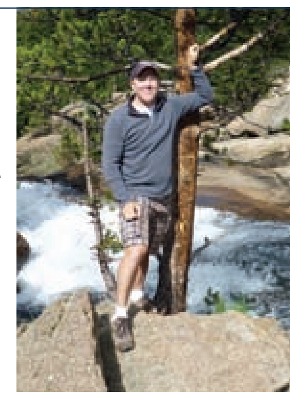
Additional BES work found that suburban catchments can retain surprisingly high levels of nitrogen, but are sensitive to small-scale conditions, such as the decoupling of riparian areas and channels. Where channels are incised, the ability of riparian areas to act as sinks for nitrogen is significantly limited. suggesting that particular types of restoration could be strategically designed to reduce nitrogen export.

Useful Resources

- LTER sites: the BES (www.beslter.org) and Coweeta in western North Carolina (coweeta.uga.edu).
- Consortium of Universities for the Advancement of Hydrologic Science, Inc: www.cuahsi.org and his.cuahsi.org.
- Major stormwater recommendations in the 2008 report *Urban Stormwater Management in the United States* from the National Research Council: www.epa.gov/npdes/stormwater.

Kevin J. Kirsch

Water Resource Engineer, Wisconsin Department of Natural Resources



Kevin Kirsch has a BS and an MS in biological systems (agricultural) engineering from the University of Wisconsin–Madison. Kevin worked as a consultant prior to joining the Wisconsin Department of Natural Resources (DNR). For DNR, Kevin provides statewide modeling and engineering support and develops policies and performance standards to address urban and agricultural runoff. His recent work has focused on the development of a water quality trading framework for Wisconsin and the development of sediment and nutrient TMDLs.

Q: How have you been involved in watershed-based monitoring or modeling?

A: The Wisconsin DNR oversees and carries out monitoring programs and modeling initiatives throughout the state and its watersheds. For monitoring, Wisconsin uses a three-tiered approach: tier 1, monitoring for baseline information to meet needs at a broad spatial scale and to evaluate trends; tier 2, targeted monitoring in support of Clean Water Act Section 303(d) listing and delisting, development of TMDLs, and assessment of individual water bodies; and tier 3, monitoring that evaluates the responses of core indicators from tiers 1 and 2 for management actions.

Modeling is conducted statewide at various scales for the various purposes. Modeling is conducted at the watershed scale for TMDLs and additional efforts to prioritize areas for restoration and to help set reduction goals that aid in meeting water quality goals. In addition, modeling is conducted at smaller scales because the municipal and field scale can support regulatory requirements and implementation strategies. For example, in Wisconsin, all the permitted municipalities modeled their permitted urban areas to evaluate pollutant loads and management strategies. The majority of our municipalities use the Windows version of the Source Loading and Management Model (WinSLAMM). On the other hand, agricultural producers evaluate soil and nutrient management strategies using SNAP-Plus.

Q: In this issue, we focus on the use of models and monitoring to assess watersheds and how these tools are used to inform the decision-making process. How do you see monitoring and modeling applied in watershed management?

A: Monitoring and modeling support and inform each other. For example, traditionally, modeling is used to develop TMDLs, and monitoring supports the calibration and validation of the models. In addition, monitoring is used to list or delist impaired streams, and modeling is used to evaluate implementation plans and the likely impacts of various management practices. The challenge with monitoring is always how to evaluate monitoring data given the spatial scale and climate variability inherent in a monitoring strategy. This variability can be partially addressed by using modeling to augment the monitoring. For example, if the goal is to delist a stream segment or to evaluate the impact of management practices, then models can help determine the management practice density and location needed to document water quality changes.

Currently, an exemplary watershed project is being conducted by multiple partners in the Pecatonica Basin, located in south-central Wisconsin. The project illustrates how ideal modeling and monitoring dovetail with efforts to identify and target water quality improvement source areas. Also, the many partners in this example come together to illustrate the importance of collaboration. I hope that what we learn in this study and in similar studies will be replicated in Wisconsin's rural watersheds.

Q: What type of monitoring or modeling is working best to predict or plan nutrient load reduction to streams in the watersheds where you work?

A: We need to remember that there is a clear distinction between the field-scale analysis conducted with SNAP-Plus and the watershed-scale modeling often done in a TMDL. The numbers coming out of the two efforts are generally not directly related, and both efforts are essential. The watershed-scale analysis provides the big picture, potentially prioritizes subwatersheds to target implementation based on overall contribution, and helps set reduction goals that link pollutant loads with water quality response. The smaller field-scale analysis is needed to target the critical areas in the subwatershed and provide implementation analysis for individual nonpoint sources. For example, the SNAP-Plus model successfully helps farmers plan their nutrient management options (e.g., the manure application includes both the application amount and the timing and has a goal to reduce phosphorus entering surface waters). The problem is that, generally, a linkage between the two is not provided. For example, a watershed-scale model in a TMDL may say that the stream can assimilate 100 pounds of phosphorus, but this represents a delivered load, often over a specific time period (e.g., daily, monthly, or seasonal). The field-scale tools that are available do not provide that same delivered number. For example, the existing agricultural tools like SNAP-Plus and the Revised Universal Soil Loss Equation (RUSLE2) allow only average annual evaluations. This results in obvious challenges that can be successfully overcome.

Q: What programs or assistance are available to advance the application of monitoring and modeling for watershed protection? Which assistance needs are being met and which types of assistance are still unavailable? Is the information about available assistance getting to the practitioners?

A: The assistance available depends on the scale at which we work. We have a combination of grant programs composed of state and federal funds to support planning efforts, but the need is far greater than the available funds. This is especially true when we look at the financial resources available for implementation phases. In my experience, we generally know what needs to be done to put the practices in place, but need the resources, as well as the political or societal will, to do it. A conservation obstacle we face in Wisconsin is that farmers can receive a 75% cost-share requirement for installing required management practices. This cost-share requirement is often viewed in a negative light. I feel it is important to note that numerous farmers voluntarily install management practices without cost-share assistance. Regardless of the philosophical discussions regarding cost sharing, the biggest obstacle is the funding to put our modeling and monitoring efforts into action.

Q: Most TMDLs are developed using a combination of (simple to complex) monitoring and modeling, and all are associated with some level of uncertainty. What recommendations do you have for reducing this uncertainty and improving the accuracy of TMDL allocations using monitoring or modeling? How can local jurisdictions better address the urban versus agricultural source load allocations using monitoring or modeling?

A: First, to reduce the uncertainty, we need to improve the statistical methods used to allocate pollution load reductions. To do this, we need to address uncertainty in TMDLs by using long-term data sets, not just one year of rainfall data, and by qualifying the data by either stating the limitations or relating the data to the site conditions (e.g., rainfall) during the collection period(s). In addition, compliance with water quality standards in a TMDL should be expressed on a probability basis.

Second, one issue of TMDL allocation expressions is that generally no link is made between the watershed-scale model sent by the regulator and the field-scale implementation that is used by the regulated entity. We need to express the pollution load reductions in transferable and useable units for field-scale implementation. For example, a farmer can understand that he or she needs to reduce edge-of-field pollution by 3 pounds per acre and can use modeling tools such as SNAP-Plus to run management scenarios. However, a farmer will struggle to interpret a load reduction of 100 pounds of phosphorus for the entire watershed.

Q: Can you share a "success story" where monitoring or modeling efforts have helped improve understanding or further watershed protection efforts? If so, who was involved (e.g., organizations, volunteers, or researchers)?

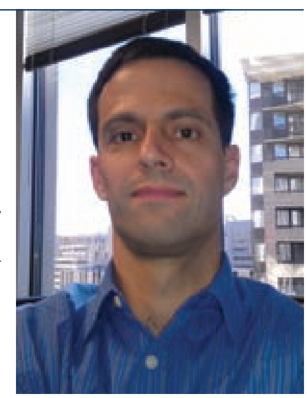
A: A success story here in Wisconsin highlights the importance of scale that I discussed earlier. In Pleasant Valley, we combined modeling and monitoring to focus on target areas at the field scale where implementation will reduce the majority of the pollution load. To get at that finer scale, researchers interviewed 95% of the farmers, input their management practices with soils monitoring data into SNAP-Plus, and honed in on the "hotspot" areas where implementation could make the biggest improvement. Several studies found that 20% to 30% of the farm fields were the source of about 50% of the phosphorus and sediment load. Stakeholders are now working with these farmers to reduce the loads on their land in ways that coincide with their farming practices, beliefs, and abilities. Interestingly, we found that some very high pollution loads can be reduced with only slight changes to a farmer's management and can often save production costs over longer time frames.

Useful Resources

- WinSLAMM: winslamm.com/.
- Snap-Plus: www.snapplus.net/.
- RUSLE2: www.ars.usda.gov/research/docs.htm?docid=6010.
- Pecatonica River: Wisconsin Buffer Initiative Pilot Project: wi.water.usgs.gov/surface-water/9ko46/documents/USGS_FieldTrip_Handout_092010.pdf.

Jason Papacosma

Watershed Programs Manager, Arlington County Department of Environmental Services, Office of Sustainability and Environmental Management



Jason Papacosma is currently the watershed programs manager for the Arlington County Department of Environmental Services. He has worked for Arlington since 1999. His work unit develops and implements watershed management policies, programs, and projects; manages Arlington County's municipal separate storm sewer system (MS4) permit; and develops and oversees development-related stormwater regulations. Jason's watershed programs team also performs stream assessments, water quality monitoring, and stormwater facility inspections; reviews development plans for stormwater management compliance; and works with citizens on a variety of watershed issues. Jason holds an MS in environmental science from the University of Maryland and a BS in environmental studies and biology from Bowdoin College, Brunswick, Maine.

Q: How have you been involved in watershed-based monitoring or modeling?

A: I manage Arlington County, Virginia's, local watershed management program. Specifically, I help meet regulatory measures and protect or restore impacted stream systems. My expertise is in applying academic and professional training to implement Arlington County watershed improvement projects. Arlington County is urban, covering 26 square miles, with 42% impervious cover; it is home to 210,000 people, has 28.5 stream miles, and 330 miles of storm sewer. We monitor for permit requirements, such as the MS4 permit, for which we monitored four storm sewer outfalls for five years. We also do instream chemical monitoring, along with geomorphological, biological, and bacteria monitoring. We use several models, including USEPA's Stormwater Management Model (SWMM), the US Army Corps of Engineers' Hydrologic Engineering Centers River Analysis System (HEC-RAS), the Center for Watershed Protection's Watershed Treatment Model (WTM) and Runoff Reduction Method (RRM), and others to help us plan and design specific projects as well as anticipate future levels of effort, watershed benefits, and milestones in the county.

Q: In this issue, we focus on the use of models and monitoring to assess watersheds and how these tools are used to inform the decision-making process. How do you see monitoring and modeling applied in watershed management?

A: We use monitoring to provide a snapshot of watershed health, track trends, and meet permit requirements. We use modeling to provide watershed management planning tools, such as the level of effort needed, pollutant load scenarios, and cost–benefit analyses for our management options. We use what the monitoring experts tell us about the performance of various stormwater practice technologies and designs for local implementation. Often, monitoring for permit requirements to assess program effectiveness is inefficient because we have too few data points or because the data are too variable or have large errors. In Arlington County, local-level implementation is our business and where we need to and have to focus our efforts!

Q: What type of monitoring or modeling is working best to predict or plan nutrient load reduction to streams in the watersheds where you work?

A: Chemical stormwater monitoring is labor- and cost-intensive, and the results are not particularly useful given what we already know about stormwater quality from years of monitoring around the country. We prefer the use of biological monitoring to assess stream ecology and trends because it integrates physical and chemical stream conditions. And, we use geomorphological monitoring to support stream restoration planning efforts and postrestoration performance. Also, we use simple land use–based models, such as the WTM or RRM to evaluate our current programs and assess possible future scenarios. For example, we used the WTM for the MS4 permit report to see what pollution load reduction we have achieved, we used the RRM to assess Watershed Implementation Plan strategies, and we used SWMM to assess the storm sewer capacity as well as the projected retrofit impact in a watershed. Various models allow for cross-checking of scenarios, which can result in more effective and efficient watershed planning efforts.

Q: Indicators for watershed health focus predominantly on surface water chemistry. To what extent does or should monitoring and modeling focus on other indicators?

A: Surface water chemistry is labor-intensive and highly variable; therefore, it is the least useful overall for watershed assessment or planning. For us, the more useful watershed health indicators are those that represent key aquatic ecosystem function or quality measures, like those used in the Chesapeake Bay (submerged aquatic vegetation, water clarity, and chlorophyll *a*).

Q: Stream restoration is an option considered for sediment and nutrient load reduction. What modeling and monitoring is needed for stream restoration projects? How can these results be applied to watershed and stormwater management to credit this as a practice through which to reduce pollutant loads?

A: High-quality monitoring is needed for pre- and post–stream restoration. For the prerestoration assessment, data such as the stream type, channel evolutionary stage, Rosgen bank erosion hazard index, and additional typical parameters should be gathered by highly qualified research professionals. This level of monitoring is likely not practical or cost-effective for every local government to conduct for every project. We should focus on more cost-effective and time-efficient consolidated monitoring efforts that cross jurisdictional lines and allow local governments to focus their limited resources on the implementation of watershed BMPs.

I'd like to see practitioners look into an approach where local, state, and federal governments pool their resources to fund regional monitoring programs that focus on specific applications and stream types. Then these high-quality data can be used to evaluate local benefits as well as for stream restoration credits in the Chesapeake Bay model.

Q: How can monitoring and/or modeling be used to inform the policy or regulatory decision-making process? Is this being done at an appropriate scale (e.g., with local land use–based decisions and stormwater programs that are required to meet federal regulations)? If not, what is missing?

A: Using the Chesapeake Bay TMDL as an example, it is now commonly understood that the resolution is not sufficient to assign local load allocations, but can be used at the subwatershed scale to frame the level of effort needed to meet the nutrient and sediment allocations for the Chesapeake Bay. Then this information can be used to establish targets for the MS4s and other permittees to scope their implementation levels of effort. Because of severe land use constraints in urban watersheds, we do not know how much "restoration" we will ultimately be able to achieve and what ecological function will exist after our stream restoration and stormwater retrofit efforts. We also know that it will take a long time and a lot of work to see improvements. The regulatory agencies need to work with local governments on an implementation approach that accounts for these uncertainties, limits, and long implementation time frames, rather than insist on strict compliance with TMDL load allocations. This is consistent with the "maximum extent practicable" standard established for urban stormwater in the Clean Water Act. We will do as much work as we can, using the best available information we have, to work to restore the Bay, then reevaluate and move forward again.

Q: Can you share a "success story" where monitoring or modeling efforts have helped improve understanding or further watershed protection efforts? If so, who was involved (e.g., organizations, volunteers, or researchers)?

A: We use our expertise to "train the trainer" in a successful volunteer biological monitoring program that has provided an excellent source of integrated data over the last eight years. This volunteer monitoring network increased Arlington County's monitoring capacity at minimal cost. The benefits are astounding. For example, because of this program, we have pre–stream restoration data in Donaldson Run and are now looking at post–stream restoration data to be able to measure the success of this practice, compare this site to other sites, and inform future stream restoration management choices. To date, the monitoring results at Donaldson Run tell us that the instream habitat is improving.

Useful Resources

- Little Pimmit Run stream corridor study: http://www.arlingtonva.us/departments/EnvironmentalServices/cpe/page60407.aspx. See, in particular, the final Little Pimmit Run hydrologic and hydraulic modeling analyses to learn more about how SWMM and the HEC-RAS model were used to assess and predict future watershed change, including the addition of watershed retrofits (http://www.arlingtonva.us/departments/EnvironmentalServices/cpe/documents/file77387.pdf).

- Volunteer stream monitoring program: http://www.arlingtonva.us/departments/EnvironmentalServices/epo/page82828.aspx. See the interactive station locations map with data.

NOMINATE A

Watershed Superstar

Do you know a watershed or stormwater professional who has demonstrated leadership and dedication to watershed protection? Consider recognizing that person's achievements by nominating him or her as a Watershed Superstar. The winner will be featured in the next issue of the *Watershed Science Bulletin*. To submit a nomination, please send an email to bulletin@awsps.org with "Watershed Superstar" as the subject line and include the following information:

- nominee's full name
- professional title
- affiliation
- short description of why the nominee is a Watershed Superstar
- name and contact information for the person submitting the nomination
- name and contact information for two people we may contact as additional references

Please keep entries to no more than 500 words; entries longer than 500 words will be disqualified. Nominees will be judged based on their accomplishments in the field of watershed and stormwater management as well as the unique qualities that make up a Watershed Superstar, including ambition, innovation, collaboration, and dedication.

Nominations must be received by Friday, May 18, 2012.

AWSPs Photolog Contest Winner

The Association of Watershed and Stormwater Professionals sponsored a photolog contest as a way to feature the watersheds in which we live, work, and play. Entries were accepted between October and November 2011, and the winner was selected by the Center for Watershed Protection, Inc.

And the winner is...

Teresa Brooks for her photo, taken near Poulsbo, Washington, of a rain garden that protects Dogfish Creek from horse manure and other pollutants. The rain garden, which is approximately 9 m long and 2 m wide, receives runoff from the barn roof and the nearby heavy-use paddock area. Native plants, such as slough sedge, slender rush, and feather reed grass, were planted to clean up pollutants. Approximately 90% of the south fork Dogfish Creek sub-basin lies within the city of Poulsbo, which is located on the shore of Puget Sound. This rain garden in the upper basin will benefit the downstream area by reducing pollutants, slowing peak flow, and recharging groundwater. This type of stormwater management is expected to result in increased steelhead, Coho, and cutthroat runs.

Latest News from AWSPs

Membership Information

Enjoy reading the *Watershed Science Bulletin*? Consider joining the Association of Watershed and Stormwater Professionals (AWSPs).

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Sponsors of the *Watershed Science Bulletin* benefit from the Center for Watershed Protection's status among top decision makers in the watershed and stormwater business. For additional information about sponsorship within the journal, please visit www.awsps.org/media-kit.

Future Bulletin Issues

Fall 2012	Spring 2013
Watershed Planning	*Green Infrastructure*

The deadline for article submission for the Spring 2013 issue is **Friday, October 5, 2012**. For submission requirements, visit www.awsps.org/publications/watershed-science-bulletin.html.

Upcoming Events

- April 18, 2012, 12–2 p.m., Webcast: Build This—Stormwater Retrofit Construction Issues (www.cwp.org/our-work/training/webcasts)
- June 20, 2012, 12–2 p.m., Webcast: Stream Restoration: Implementation You Can Take to the BANK (www.cwp.org/our-work/training/webcasts)
- August 15, 2012, 12–2 p.m., Webcast: Get the Dirt on Stormwater (www.cwp.org/our-work/training/webcasts)
- October 8-10, 2012, Watershed & Stormwater Conference (www.CWP2012EVENT.com)

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