Fundamentals of Urban Runoff Management

TECHNICAL AND INSTITUTIONAL ISSUES

2nd Edition | 2007

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NALMS

Fundamentals of Urban Runoff Management: Technical and Institutional Issues

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The North American Lake Management Society (NALMS) support for this effort is most appreciated. NALMS' mission is to forge partnerships among citizens, scientists, and professionals to foster the management and protection of lakes and reservoirs for today and tomorrow. Lakes and reservoirs are used more now than ever and water pollution is a key problem in maintenance of lake health. NALMS supports the development, communication, and use of excellent science and cutting-edge management to further lake protection goals.

We hope this manual will provide information that can be used to further NALMS' goals.

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Preface

In 1994, the Watershed Management Institute, through the Terrene Institute and in conjunction with the U.S. Environmental Protection Agency (EPA), published *Fundamentals of Urban Runoff Management: Technical and Institutional Issues*. That manual combined technical and institutional information to provide a handy resource for practitioners and regulators for both erosion and sediment control and stormwater management. The manual was well received.

As luck would have it, in 2001 several of the original authors met up at a conference, and began to discuss the amount of new information available and our desire to update the previous work. The idea was planted and communication between the original authors began. Most of the authors wanted to contribute so we went about looking for a vehicle for distribution. As two of the previously involved organizations were not available, discussions began with the North American Lake Management Society (NALMS) which felt the manual would provide a resource for its members and anyone working with stormwater impacts on aquatic habitats. Discussion began with the EPA for funding assistance, which was subsequently approved.

If the new information represented only an evolution or increase in the data available, this book would probably not have been pursued. Rather, there has been a significant shift in program direction that represents a movement from the historic mitigation-based approach for stormwater treatment to a more source-based approach. The main reason for this shift in thinking is based on an increased recognition that streams are a valued aquatic resource that should be protected.

This change in thinking necessitated a philosophical shift from larger stormwater practices on streams to the use of practices on individual subdivisions and even individual lots. Linking stormwater goals to aquatic resource protection mainly necessitated this change in approach. Much more information continues to become available to demonstrate the significant shift necessary to protect and enhance aquatic resources. We are pleased that this new edition of *Fundamentals of Urban Runoff Management* can play a role in that shift.

CHAPTER 1

Introduction

Introduction

Discussion

To begin, this chapter sets out some of the reasons for updating *Fundamentals of Urban Runoff Management*. These include:

- The shifting emphasis and impacts of stormwater management programs and regulations;
- The Storm Water Phase II Rule published on December 8, 1999, which greatly expanded the scope and coverage of the Phase I program;
- The increased emphasis on the Total Maximum Daily Load (TMDL) approach to stormwater management;
- Changing hydrologic approaches that increasingly consider long-term continuous simulation of rainfall to more accurately size BMPs;
- The increase in the quantity and quality of water data;
- The increased prominence of biomonitoring and biocriteria; and
- New and improved stormwater management practices.

Shifting Program Emphasis

When the original *Fundamentals* manual was written in 1994, programs were focused on limiting peak discharges and providing water quality treatment. The performance of practices for quality treatment was more assumed than realized. In addition, there was little widespread documentation of practice performance and the relationships between hydrology and water quality. Aquatic ecosystems were also not recognized to any great degree. There was an assumption that removal of contaminants would be good for the environment, but at the time few studies had been done to verify the accuracy of that assumption. There has since been significant work done in this regard, with one of the studies (Horner et al., 2001) assessing the effectiveness of structural practices to protect stream aquatic resources from a watershed-wide perspective. They make a number of interesting statements, although some need to be further documented. Key findings were:

- Until watershed total impervious area exceeds 40 per cent, biological decline was more strongly associated with hydrologic fluctuation than with chemical water and sediment quality decreases. Accompanying hydrologic alteration was loss of habitat features, such as large woody debris and pool cover, and deposition of fine sediments.
- Structural BMPs at current densities of implementation demonstrated less potential than the non-structural methods (riparian buffers, vegetation preservation) to forestall resource decline as urbanization starts and progresses. There was a suggestion in the data, however, that more thorough coverage would offer substantive benefits in this situation. Moreover, structural BMPs were seen to help prevent further resource deterioration in moderately and highly developed watersheds. Analysis showed that none of the options is without limitations, and widespread landscape preservation must be incorporated to retain the most biologically productive aquatic resources.

- Structural BMPs can make a substantive contribution to keeping stream ecosystem health from falling to the lowest levels at moderately high urbanization and, with extensive coverage, to maintaining relatively high biotic integrity at light urbanization.

Clearly, we are finding that you cannot separate water quantity and water quality issues if aquatic resource protection is a program goal. Some people, mainly from an anecdotal perspective, have recognized this, but now increasing amounts of literature support that fact. What has clearly come out of recent research is the relationship of land use to aquatic system health and well-being: it isn't just pollutants that are an issue.

Phase II Storm Water Rule

The Storm Water Phase II Rule published on December 8, 1999 greatly expanded the extent of the Phase I program. This was done by requiring operators of municipal separate storm sewer systems (MS4s) and operators of small construction sites (greater than one acre) to obtain National Pollutant Discharge Elimination System (NPDES) permits that implement programs and practices to control polluted stormwater runoff.

The expansion of Phase II is directed toward municipalities with populations under 100,000, which were not covered in Phase I. There are a number of variations to the general requirement, best set out in fact sheets developed by the EPA (Storm Water Phase II Final Rule Fact Sheet Series).

The bottom line is that most municipalities and federal facilities in the U.S. are now covered by the Storm Water Program and must implement programs and practices that control stormwater runoff. The minimum control measures required by the EPA as essential to an effective stormwater management program are:

- Public education and outreach on stormwater impacts;
- Public involvement/participation;
- Construction site stormwater runoff control;
- Post-construction stormwater management in new development and redevelopment;
- Pollution prevention/good housekeeping for municipal operations; and
- Illicit discharge detection and elimination.

Increased Emphasis on the TMDL Approach

The Total Maximum Daily Load (TMDL) approach to stormwater management has existed for a number of years (originally identified in the *Clean Water Act*, 1972). For various reasons TMDLs have now assumed much more priority on a national and state basis than was the case historically. A number of TMDLs done around the country are now serving as templates to be followed. The approach is evolving fairly rapidly with new guidance information available almost on a routine basis.

Biannually, states, territories, and authorized tribes must list those impaired waters that do not meet applicable water quality standards. Lists submitted to the EPA must identify the pollutants that cause the impairment and the water bodies targeted for TMDL development. TMDLs must then be established at levels necessary to achieve applicable water quality standards, along with a margin of safety that takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality.

A TMDL specifies the amount of a particular pollutant that may be introduced into a water body and allocates the total allowable pollutant load among sources. The TMDL provides a roadmap for efforts to attain and maintain state water quality standards. TMDLs consider both point and nonpoint source pollutant loadings in determining the overall state of a receiving system and allow prioritization of efforts to achieve compliance with water quality standards.

The core of a TMDL is a computer model or simulation that predicts outcomes for various pollutants on a watershed basis. Most models in use today have been around for quite some time and are generally understood in terms of data entry and model process. Where improved data is especially important in the TMDL process is for pollutant loadings from various land uses and performance data for BMP treatment expectations. More data is absolutely essential if the TMDL process is to provide for a reasonable consideration of alternatives in a given watershed and selection of a preferred approach. There are huge issues related to funding, both public and private, and the anticipated outcome must be defined as much as possible.

Changing Hydrologic Approaches

While stormwater management has historically relied on event-based approaches to BMP design, more practitioners are now considering long-term continuous simulation of rainfall to accurately size BMPs. By considering actual long-term rainfall records in a given area, a better gauge of performance may be obtained. Analysis of continuous rainfall data over a given time, possibly supplemented by simulation of much longer terms, may give a different performance expectation than would be expected using an event-driven sizing approach. This will have a major influence on models used for analysis and on existing design standards and sizing methodologies.

Better Water Quality Data

Water quality data is becoming much more available than was the case historically. In the past, early monitoring was based to a very large extent on the results of the National Urban Runoff Program (NURP) done in the late 1970s and early 1980s. The NURP study provided a national perspective on water quality issues, but there are now many other studies done in the U.S. and around the world, notably Australia, Canada, England, New Zealand, and a number of European countries.

Another excellent source of water quality data for practice performance is the International Stormwater Management Best Management Practices Database, which provides access to BMP performance data for about 200 studies conducted over the past 15 years. This data was compiled by the Urban Water Resources Research Council of the American Society of Civil Engineers (ASCE) to provide consistent and scientifically defensible data for BMP designs and related performance. That information is available at www.bmpdatabase.org.

In addition, considerable research is being done on the performance of wetland systems, filter systems, and newer practices such as rain gardens. Many proprietary practices are also becoming more commonly used. The development of proprietary systems will continue; this should be encouraged, subject to collection of good monitoring data that would justify their use. Some of that data is already available.

Increased Prominence of Biomonitoring and Biocriteria

While there remains an important role for chemical monitoring, biological indicators are increasingly recognized as a necessary component of stormwater monitoring and assessment.

Chemical monitoring provides a picture over the period monitoring is done, while sediment sampling provides a rate of accumulation. Biological monitoring adds to the picture by providing an overall health rating of the receiving system, including a compilation of the effects of stressors on aquatic organisms, a perspective that is not available through chemical or sediment monitoring. As such, it can be considered the third leg of the monitoring stool: without all three legs the picture is not complete.

Improved and New Practices

Stormwater Management

This is an exciting time to be considering stormwater management and means of reducing impacts related to society's use of land. Initial stormwater management efforts focussed on control of water quantity related to flooding impacts. Flood control programs were generally initiated in response to a local flooding event and involved channel modifications, detention dams, or floodplain regulation. As the issue of water quality became more recognized, the existing infrastructure of flood control programs was generally modified to incorporate water quality concerns.

The approach at the time was to modify existing water quantity practices to also provide water quality improvement; however, the overall design philosophy was still directed toward large, on-line stormwater treatment systems that first and foremost provided control of downstream flooding and through design approaches (wet ponds) provided water quality treatment. There was little consideration of the stream or receiving system as an important resource. That lack of importance changed in the early 1990s. It was also recognized that one practice could not provide treatment for a wide range of pollutants: filter systems, wetlands, and biofiltration practices were all investigated for pollutant reduction.

As a result, stormwater management has become a very different entity than it was in the 1980s. There is much more emphasis on practices at the headwaters of perennial streams, and practices are being much more targeted to the pollutants generated through specific land use activities than was done historically.

Finally, there are new practices being developed as variations of their historic counterparts. Filter systems are being used with filter media other than sand, infiltration practices are being considered on a wider basis, and newer practices (at least from a U.S. perspective) such as rain gardens, green roofs, and water re-use are being advocated. These really are exciting times.

Erosion and Sediment Control

Erosion and sediment control practices have not evolved to the same extent as those for stormwater. The suite of practices in use has remained pretty much the same since the early 1970s. That can't be good. Other components that have more recently begun to emerge consider temporary and permanent revegetation, phasing work to limit open areas, and chemical flocculation of sediment ponds to provide for enhanced sediment discharge reduction, especially of clay soils. More attention to erosion and sediment control practices is needed for improved treatment to be achieved.

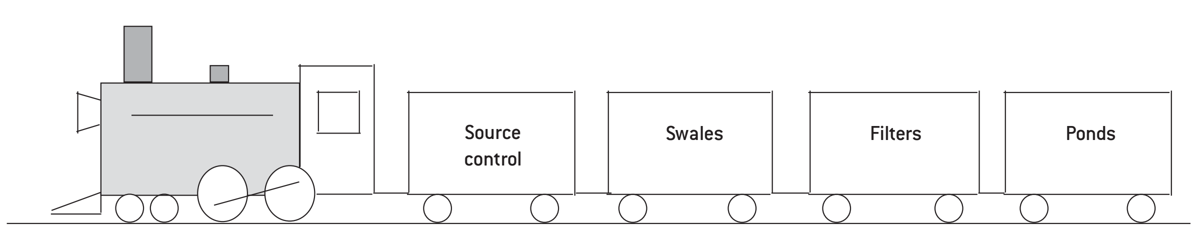
Where From Here

As both our understanding and practices evolve, more emphasis will be placed on the "treatment train" concept, where several types of stormwater practices are used together and integrated into a comprehensive stormwater management system. Although this is obvious when multiple issues are considered (such as stormwater quantity, quality and aquatic ecosystem protection), it is also sometimes needed when considering a single issue. For example, stormwater quality may include a variety of contaminants to be managed, but processes that facilitate one type of pollutant in one practice may not facilitate removal of a pollutant in another phase (liquid versus particulate). The treatment train approach to stormwater management will become increasingly important to reduce overall stormwater impacts on the urban environment.

For erosion and sediment control programs, technology must improve and approaches further refined for aquatic resource protection to be realized. An aggressive stormwater management program will not realize its goals if the receiving systems are severely impacted during the construction phase of a project. In addition to significant sediment loads, the amount of stormwater exiting a construction site can be significantly increased, causing downstream channel instability concerns. Erosion and sediment control must be given greater attention by regulators and designers. It is a positive step, therefore, that the Phase II program is also emphasizing erosion and sediment control on smaller sites than did the Phase I program.

Most importantly, if we are to reverse the existing trend of aquatic and terrestrial destruction that so defines traditional development, we must alter our existing approach to land use. There may be areas of significant habitat, groundwater recharge, or steep slopes where intensive land development is simply not appropriate. Those areas should protected, regardless of their location, and urban planners should instead insist on higher densities in other areas. Stream corridors should be protected, riparian cover established (or re-established), water re-use emphasized to reduce the use of potable water in addition to reducing stormwater runoff, and the use of green roofs should be expanded,

Figure 1-1: Stormwater Treatment Train



especially for redevelopment opportunities. Finally, stormwater management implementation should be done as an integral component of site development and as an urban retrofit.

We are approaching a point where we now have the tools to eliminate further declines in receiving systems, and in a number of situations actually improve on existing conditions.

Concluding Thoughts

Stormwater management has historically been an afterthought – when thought of at all – to the site development process. Development tends to first lay out streets, lots, and public areas, and then consider how to deal with any required stormwater management concerns. As long as stormwater management remains an afterthought, even the best resource protection intentions are doomed to fail.

In the same regard, we too often design for minimum standards in environmental areas, with no factor of safety. If a code says to stay out of wetlands, we stay out of them, barely. In the same regard, if we have a design standard of 80 per cent reduction in TSS, that is what designers will design for – very seldom does someone intentionally design for a higher standard. We really ought to consider a factor of safety in land development to allow for better assurance of a desired outcome.

We must also recognize that we have not yet fulfilled our potential understanding of how best to protect the environment. We are learning, and we hope to apply our increasing knowledge to better outcomes, but stormwater management is an inexact science and there are huge pressures on land use, along with infrastructure provision, to be considered. We aren't alone in our efforts, although it may seem like it at times. People all around the world are dealing with the same problems and developing innovative solutions we have not yet thought of.

All of us have never-ending jobs in teaching other staff members, politicians, members of the design and construction community, and the public. At the same time, we must never cease to be students, always willing to learn and apply new information and insight for the betterment of the environment.

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CHAPTER 2

Water Quantity Impacts of Urban Land Use

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> *Urban runoff is a by-product of the land's interaction with rainfall. Since, by definition, urban runoff remains on and moves along the land's surface, it is the most visible of the many forms into which rainfall is converted. This chapter provides the technical fundamentals of the rainfall-runoff … process. It also describes ways that land development alters this process and quantifies some of the adverse impacts.*

So began Chapter 2 of the 1994 edition of *Fundamentals of Urban Runoff Management*. And while it still can serve as the opening paragraph of this new Chapter 2, our technical knowledge of both urban runoff hydrology and the effects of land use change has grown considerably in the intervening years. As a result, the technical content of this new chapter goes beyond the original version, including new and updated topics. However, in presenting this technical information, the chapter's goal remains the same: to present the information not as an end in itself, but so as to assist in the development of urban runoff management programs. The arrival of the EPA's Stormwater Phase II Final Rule in 1999, which requires municipalities and other entities to develop such programs by 2003, highlights the value of such assistance.

The volume of stormwater runoff produced by a rain event, the rates, velocities, and depths at which it flows, and the pollutants that it carries depend on several factors. In addition to the quantity, intensity, and duration of the rain itself, the resultant runoff will be determined by the characteristics, condition, and relative areas of the various surfaces on which it falls. As explained in detail in the following sections, these characteristics include the type of surface cover, the surface slope, and the texture, density, and permeability of the surface and subsurface soils. Conditions that affect stormwater runoff also include the thickness and quality of the surface cover and the amount of water already stored both on the surface and within the soil profile.

Conversely, stormwater runoff also affects the surfaces upon which it is created and/or that it flows across. These effects include both the deposition of pollutants captured from the atmosphere by the falling rain and the mobilization and removal of pollutants previously stored on the surfaces. The most readily visible effects are erosion and sedimentation, where forces created by the moving runoff become large enough to dislodge, suspend, and transport soil particles and associated pollutants downstream. This process continues until slower velocity areas are encountered, whereupon the particles drop out of the runoff and back onto the surface. Depending on the type and character of the surface cover, this process of dislodging soil particles and mobilizing pollutants can be aided by the impact of the falling raindrops themselves. Further erosion, sedimentation, and pollutant loading can occur downstream in swales, channels, streams, and rivers, depending on the rate, depth, velocity, and duration of the runoff flowing in them.

From the above, three key conclusions can already be reached:

- Since the volume, rate, and velocity of runoff from a particular rain event will depend upon the characteristics of the surfaces on which the rain falls, changes to these surfaces can significantly change the resultant runoff volume, rate, and velocity. Changes normally associated with land development and urbanization that increase impervious cover and decrease soil permeability can significantly increase runoff.

- Since pollutant mobilization and soil erosion are the direct result of excessive runoff rates and durations, changes in land surfaces can also significantly increase both surface and channel erosion rates and runoff pollutant loadings.
- In developing urban runoff management programs, the greater the knowledge of the rainfall-runoff process, the more effective the resultant program will be.

While the details of the rainfall-runoff process are highly complex and much remains to be learned about them, the fundamentals are readily understandable, particularly when presented in a direct, concise manner. That is the goal of this chapter. Equipped with the information presented here, those involved in developing urban runoff management programs at all levels, as well as those responsible for complying with them, can base their efforts on a sound understanding of the basic hydrologic processes at the core of their program.

This chapter provides readers with basic information on the rainfall-runoff process. It also highlights some of the important unknowns and uncertainties of the process and recommends ways to acknowledge and account for them in computation methods and program requirements. Using this information, the chapter also provides information on the adverse impacts land use change and urbanization can have on runoff quantity and the damaging consequences of excessive increases in runoff rates, volumes, and velocities.

Next, the chapter utilizes this rainfall-runoff information to illustrate how various practices can either avoid or control such impacts. This broad approach not only helps ensure that decisions made during the development of an urban runoff management program are based on an informed understanding of runoff fundamentals, but also helps readers to better understand the more technically complex topics presented in subsequent chapters.

The chapter concludes with a list of recommended textbooks, research papers, and other references. These works were selected from a constantly growing body of technical information on urban runoff and the impacts of land use change based upon their seminal or definitive role in the field of urban runoff management. In light of the chapter's broad scope and emphasis on learning the fundamentals first, these references can be used to expand readers' knowledge beyond the pages of this book.

It is important to note that, as our understanding of urban runoff processes and impacts continues to grow, so does the scope and requirements of the programs we've developed to manage them. Following along and, at times, inspiring this growth has been an increasing emphasis on and understanding of runoff fundamentals. It is this greater understanding that has allowed us to progress from relatively simple runoff quantity controls in the 1970s to the integrated quantity and quality programs of today. It has also allowed us to expand the scope and applicability of both our mathematical models and the various measures and practices we can now use to implement their findings. For example, the growing use of nonstructural measures and low-impact development practices essentially began with a detailed re-examination of the fundamental principles of the hydrologic cycle which, in turn, became the basis for their design and implementation. Therefore, it is hoped that the runoff fundamentals presented in this chapter will continue to inspire and direct the development of urban runoff programs with ever greater scopes, goals, and accomplishments.

Reality vs. Theory

In most complex technical matters, differences exist between reality and theory. That is because theories developed to explain or simulate reality can only go so far. Typically, there are aspects of reality that are not entirely understood and, therefore, are either ignored or simplified in the theory. Recognizing these differences is important when developing and implementing a technology-based regulatory program such as one that manages urban runoff. The "real" runoff processes that occur during an actual storm event can be extremely complex and can be influenced by an equally complex, highly variable set of factors and circumstances. Due to this complexity, the theories on which we base our runoff computations and models cannot include all aspects and factors.

For example, the mechanics of infiltration that govern the amount and rate at which rain will enter a soil (and therefore the amount and rate that will become runoff) are difficult to precisely discern. They can include the forces that govern the movement of water entering and moving through the void spaces within the soil as well as the intensity of the rainfall, the sizes, shapes, and chemical characteristics of the soil particles, the number and size of the void spaces between the soil particles, the amount of moisture already stored within the soil void spaces at the onset of rainfall, the slope and relative smoothness of the soil surface, and the type and character of the cover on the surface. Further complications include the fact that many of these forces and factors typically change over time, not only from storm to storm, but during a single storm event. This inherent complexity of the process, coupled with the complexity and variability of the factors that influence it, makes it difficult to develop a comprehensive theory that can precisely predict the resultant runoff from a specific rainfall event.

At first glance, this difficulty in precisely predicting runoff volumes, rates, and velocities from rainfall events does not bode well for the development of a regulatory program intended to effectively manage that runoff and its impacts. However, an awareness of these difficulties and the complexities, uncertainties, and variability that cause them can help us develop assumptions, simplifications, and representative values that enable us to overcome these difficulties and produce accurate, reliable, and safe runoff estimates. This ability further underscores how important it is for runoff management program developers to possess an understanding of runoff fundamentals.

Generally, there are three analytic techniques typically employed to overcome the complexities and uncertainties of estimating runoff and produce safe, usable results. The first involves analyzing the various processes that help convert rainfall to runoff and determining the relative influence each of their many factors may have on the process's outcome. Those parameters that are found to exert very small influence on the outcome or answer are typically dropped from further consideration in the computations or, if their presence is needed for mathematical rigor, they are assigned a nominal value. At times, factors that have minimal influence individually but, when combined, can have a meaningful and estimable effect on the outcome are grouped together and assigned a value that reflects that combined influence. Such factors are often referred to as lumped parameters in recognition of their combined contribution to the outcome. Mathematical models that utilize such parameters to estimate runoff from rainfall are known as lumped parameter models.

The second analytic technique that is used at times to address the complexities and uncertainties normally associated with runoff computations is an outgrowth of the first technique. Following the identification and analysis of the factors or parameters that influence the various rainfall-runoff processes, those factors that are found to exert a meaningful influence are further analyzed for the ways and amounts in which they do so. Sometimes called sensitivity analysis, this procedure fixes the value or influence of all other significant factors and then allows the parameter in question to vary over a range of possible or probable values. Each time the parameter value changes by a certain percentage of its total value range, both the qualitative and quantitative effects of such a change on the outcome or answer are noted. Once the entire range of parameter values is evaluated, the parameter's influence can be assessed. This assessment can indicate to the runoff modeler how much the outcome or answer will vary due to certain changes in parameter value. The assessment also indicates which direction (i.e., higher or lower) the answer will move. For example, does an increase in parameter value cause the answer to similarly increase or, in fact, to decrease? While direction influences can be readily determined for certain parameters in simple, generally steady-state rainfall-runoff models merely by analyzing their basic equations and algorithms, more complex, dynamic models may require more extensive sensitivity analysis.

Once the sensitivity and direction of a model parameter is understood, the second analytic technique then assigns it a value that the runoff modeler considers to be both a) reasonably representative of its typical value for the circumstances under consideration, and b) safe for the application or action that the model results will be used for. "Typical" values in many models are usually determined from representative numbers of actual parameter measurements taken either in the field or the laboratory. "Safe" values are based upon the parameter's directional influence and the acceptable risk inherent in the application of its results.

For example, in designing a stormwater facility to reduce peak runoff rates and pollutant loads from a land development site, a key design parameter would be the ability of the site's soils under developed conditions to infiltrate rainfall. While there may be extensive data available to the designer upon which to select a typical infiltration value, the designer may also allow the desire for a safe value (and, consequently, a safe design) to influence the final selection. As a result, the designer may select an infiltration rate for the developed site that is somewhat lower than the typical value, knowing that its use value will result in greater runoff volume and peak rate to the facility which, in turn, would require a somewhat larger facility size than if the typical value was selected. Once again, the selection of a safe parameter value may be a matter of experience and professional judgment when using simple, generally steady-state rainfall-runoff models or may require extensive statistical analysis when using more complex ones.

Selection of safe design parameters may also be complicated by the design itself. For example, in the design described above, the selected infiltration rate for the site soils under developed conditions was lower than the actual or typical rate in order to achieve a conservative facility design. However, let's assume that the required peak outflow rate from the facility could not exceed the peak rate from the site in existing or predeveloped conditions. In computing this predeveloped peak rate, use of a lower than actual soil infiltration value would not be considered safe, since it would result in a peak rate from the predeveloped site (and, therefore, the stormwater facility under developed site conditions) that was greater than the actual predeveloped site rate. In order to select a safe value, the designer would instead need to select a soil infiltration rate for the predeveloped site that was actually higher than the actual value.

As illustrated by these examples, a stormwater facility designer must understand the basics of the rainfall-runoff process in order to consistently select safe parameter values. We cannot be sure that our assumptions, computations, and, ultimately, our runoff management programs are inherently safe unless we understand the fundamental aspects of urban runoff well enough to identify all pertinent factors and parameters and understand their effects. This conclusion once again revisits the "learn the fundamentals first" theme of this chapter.

It should be noted that the use of a "safe" parameter value cannot typically be relied on to address process complexity and uncertainty when attempting to estimate runoff from actual rain events. Such events are often described as "historic" events to distinguish them from synthetic design storms, which are typically based upon a hypothetical arrangement of rainfall depths, intensities, and durations that are often used to design stormwater facilities. Estimating runoff from actual rainfall events is often necessary to demonstrate the accuracy of a particular rainfall-runoff model or to provide feedback that can be used to improve its accuracy. Such procedures are known as model calibration and verification, where a model's algorithms and/or parameter values are adjusted so that its predicted outcomes match the recorded outcomes from actual or historic storm events. Once so adjusted (or calibrated), the model is then used to predict the outcomes for one or more additional historic storms. The predicted results from the calibrated model are then compared with the additional storms' recorded outcomes to verify or validate that the model remains accurate for storms other than the one by which it was calibrated. When estimating outcomes for actual rain events, the selection of model parameter values must usually be based only on the parameter's actual value (or values) during the actual event, a process that requires considerably more understanding of the rainfall-runoff process and usually event-specific records of parameter data.

The third analytic technique addresses the complexities and uncertainties normally associated with runoff computations by including such uncertainties in the runoff computations. To do so requires a rainfall-runoff model that will simulate a large number of storm events. While doing so, the model will allow the value of the uncertain parameter to vary from event to event or even within a particular event based upon the way the parameter may be expected to vary in reality. Such variations may follow a particular pattern (e.g., exponentially or logarithmically) so that, while the actual parameter value for a particular rain event may not be known, the overall range of values and the pattern by which the parameter value varies within that range is known or can be reasonably estimated. Equipped with such information and utilizing a technique known as Monte Carlo simulation (Pitt and Voorhees, 1993), the model will allow the parameter value to vary within the known range and pattern either randomly or in accordance with prescribed probabilities. The results produced by the model can then be statistically analyzed to determine an appropriate answer. Depending upon the parameter, such variations in parameter value can represent a more accurate way to address parameter value uncertainty than selecting typical and/or safe values. However, use of Monte Carlo simulations requires the use of generally more sophisticated rainfall-runoff models and long-term rainfall input data. Further discussion of such models is presented in later sections of this chapter.

In summary, the above section presented the following ideas and information:

- Inherent complexities in the rainfall-runoff process lead to differences between the theories, equations, and models we use to estimate runoff rates and volumes and the actual amounts that may occur;

- To safely address these differences, we utilize both our understanding of rainfall-runoff fundamentals and techniques such as sensitivity analysis to select equation or model input parameters that will produce answers that are accurate and safe; and
- In certain instances where appropriate data and models are available, we may actually allow an input parameter to vary during the computations rather than using a single value for it. Known as Monte Carlo simulation, it produces a range of possible answers that can then be statistically analyzed to produce an accurate and safe answer.

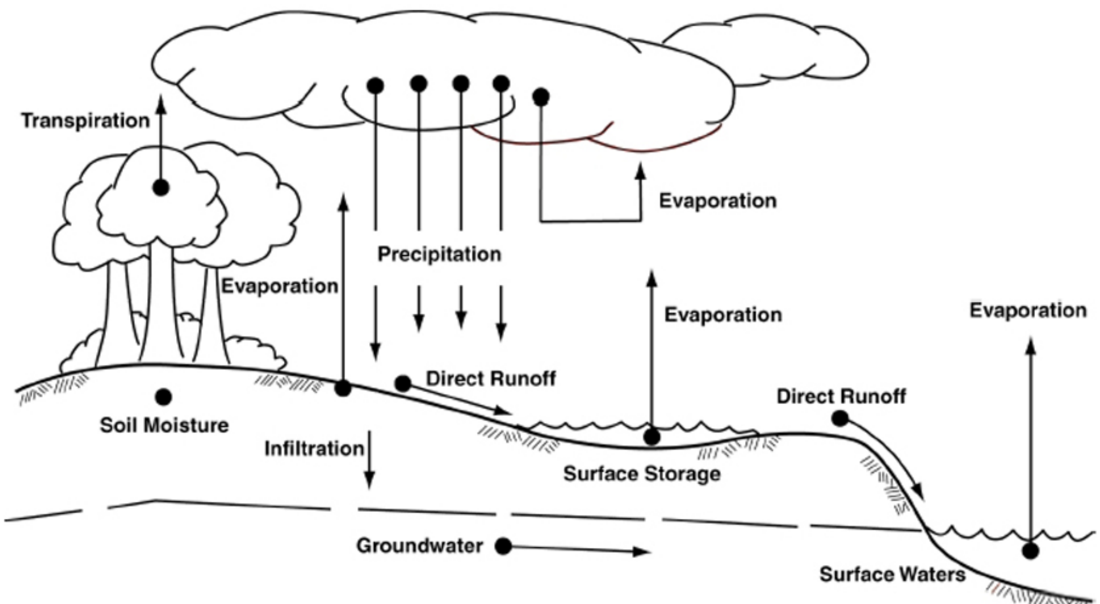
Finally, the role of urban runoff management program developers should not be overlooked in the above. That's because the theories, equations, models, and input parameter values they choose to incorporate into their programs will influence and even require designers to follow certain procedures, include certain parameters, and/or select certain data values. As such, it is just as important for the program developer to understand the fundamentals of the rainfall-runoff process.

The Rainfall-Runoff Process

As described in the chapter's opening paragraph, runoff represents a by-product of the land's interaction with rainfall. As such, changes in the character or cover of the land can cause changes in runoff volumes, rates, and velocities. However, to better understand the rainfall-runoff process, it is important to realize that it is only a portion of a larger, cyclical process that is constantly taking place. This process, known as the hydrologic cycle, involves all of the forms water can take as it continually moves on, above, and within the earth.

The hydrologic cycle is illustrated in Figure 2-1. Due to its cyclical nature, there are no starting or ending points in the hydrologic cycle, just points along the way as water moves between the earth's surface and atmosphere, changing its form as necessary. Selecting the atmosphere as a starting point, Figure 2-1 demonstrates how water vapor is converted into rainfall and other forms of precipitation and is pulled by gravity toward the earth's surface. On the way, some of the precipitation may be converted back to water vapor and remain suspended in the atmosphere, while the remainder continues to fall. Upon reaching the earth's surface, precipitation can follow one of several routes. It can be stored in surface depressions or infiltrate into

Figure 2-1: The Hydrologic Cycle



Source: Fundamentals of Urban Runoff Management the soil. Once there, it can be taken up by plant roots and, through the transpiration process, returned to the atmosphere as water vapor or remain in the soil as soil moisture.

Other infiltrated precipitation may continue to move down, again by gravity, until it reaches the groundwater table, which can then re-emerge on the surface as flow in waterways. Precipitation stored on the surface can be evaporated into the atmosphere, along with that intercepted by vegetation. Finally, a certain amount of the original precipitation can become runoff, moving across the earth's surface to waterways and bodies, including the oceans. Once there, evaporation can then return the water to the atmosphere, where precipitation can resume.

It is important to recognize two basic aspects of the hydrologic cycle. First, the movement of water from the atmosphere to the earth is exactly balanced by its movement in the opposite direction. We know this is true because, as noted in the 1994 *Fundamentals of Urban Runoff Management*, the skies would get very cloudy or inland property owners would eventually have ocean or lakeside views if it weren't. From the standpoint of urban runoff management, we can use this mass balance to help estimate how much water may exist in each of the hydrologic cycle's available forms, including runoff.

Second, due to the interaction between all of the various water forms within it, the hydrologic cycle is not easily separated into discrete components. Depending on actual conditions, the precipitation that became runoff from a parking lot may join flow in an adjacent stream, or moisture in the soil surrounding the lot, or groundwater moving below the lot. In fact, the water that was originally parking lot runoff and then groundwater may eventually become flow in the stream or evaporate back into the atmosphere where the precipitation originated.

Despite its complexity and interrelationships, experience and research has demonstrated that, to be successful, an urban runoff management program must not only be based upon an understanding of the hydrologic cycle, but must also utilize as many water forms and processes within the cycle as possible. As such, it is no longer sufficient to target and regulate only the runoff process. Instead, the program must also utilize the infiltration, transpiration, and even the evaporation processes to optimal levels in order to manage urban runoff and prevent the adverse runoff impacts of the land use changes caused by urbanization. Coordinated use of all available hydrologic cycle components and processes allows a program to move beyond simple runoff control to true runoff management, limiting the amount of rainfall that becomes runoff to begin with as well as managing the runoff that is ultimately created. In doing so, the program can also provide protection of groundwater resources, waterway and wetland baseflows, and soil moisture levels necessary for healthy vegetated covers.

In summary, the above section presented the following ideas and information:

- The hydrologic cycle represents the complex, interrelated movement of water in various forms on, above, and under the earth's surface.
- Despite its complexity, there are fundamental concepts and processes in the hydrologic cycle that can be readily grasped and utilized.
- To be successful, an urban runoff management program must be based upon the hydrologic cycle and utilize as many of its concepts and processes as possible.

Runoff Estimation: Typical Parameters

As noted above, the actual process by which rainfall is converted to runoff is complex with variable and, at times, unknown factors. Fortunately, from years of research, experimentation, and experience, the essential factors or parameters that most strongly govern or influence the process have been identified. These fundamental or typical parameters are described below.

Rainfall

Since runoff is considered its by-product, rainfall can readily be considered the most significant factor in estimating runoff. Actual rainfall amounts and patterns measured at gages are used to estimate the runoff from real or historic rain events. Hypothetical or synthetic design rainstorms are frequently used for design and regulatory purposes. Actual rainfalls can also be used to check the results produced by a design storm method or can even serve as the design storm itself if it has the appropriate magnitude, duration, and probability. This is particularly true for long-term rainfall records, which can provide superior results to design storms in certain instances (James and Robinson, 1982). As a result, the use of such rainfall records can be expected to grow in the future, particularly in the analysis and management of runoff quality, as more data becomes available and computer programs are developed to utilize it. Long term records may also serve as a valuable indicator of climate change impacts on rainfall, in which care must be taken in their use.

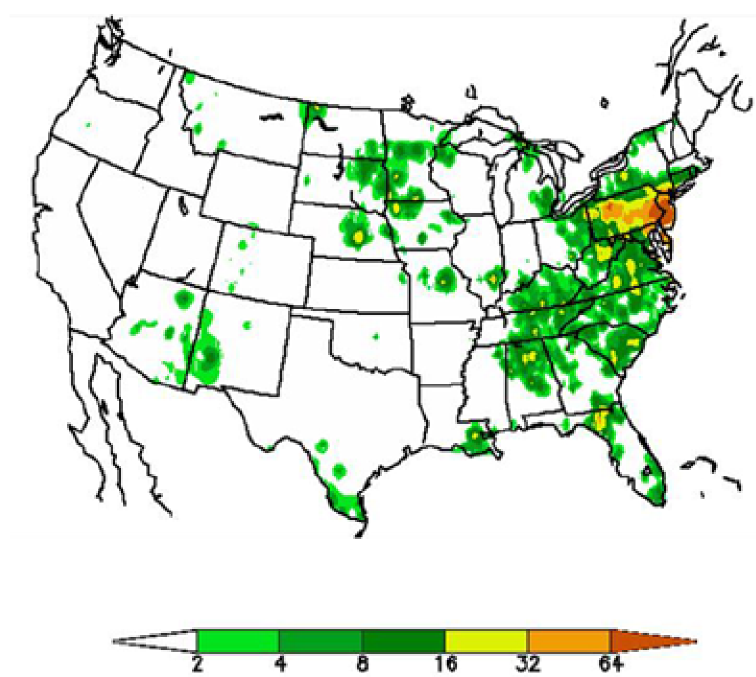
In general, our interest in rainfall not only focuses on real and hypothetical events, but also on both small and large rainfall amounts. From statistical analyses and experience, we know that small rainfalls occur much more frequently than large ones. As such, relatively small rainfalls are typically associated with runoff pollution and erosion problems and their associated environmental consequences, while larger rainfalls are typically associated with flooding and its associated threat to lives and property. The following examples highlight these various interests and the use of data from real rainfall events.

Figure 2-2 depicts radar-based total rainfall estimates in the United States during a 24-hour period ending at 8 a.m. on July 13, 2004. From the scale at the bottom of the figure, it can be seen that the greatest rainfall occurred in the northeastern United States, particularly in New Jersey and Delaware. Figure 2-3 presents a more detailed view of the rain event in this area. As can be seen in the figure, 24-hour rainfall totals of more than 11 inches fell in Kent County, Delaware, and more that 13 inches fell in Burlington County, New Jersey. As documented by the National Weather Service (NWS), U.S. Geological Survey (USGS), and the N.J. Department of Environmental Protection, this rain event resulted in record or near-record flooding on several southern New Jersey waterways, including Rancocas Creek and the Cooper River. The rain also led to the failure of 21 dams in Burlington County. An analysis of the rain event in the county by the NWS indicated that the event had an estimated average recurrence interval or frequency of approximately 1,000 years. As described later in this chapter, such an event would statistically have only a 0.1 percent chance of occurring in any given year.

Rainfall data from such an extreme rain event is not only useful in analyzing the runoff, flooding, and damage caused by the event itself. The data may also be used to evaluate the design of dams, spillways, and other hydraulic structures produced through the use of hypothetical design rainfall events or, where appropriate, may even serve as the design storm itself. Such use would depend upon the total depth, duration, and probability of the actual rain event compared with the required design frequency of the structure.

At the opposite end of the rainfall depth and frequency spectrum, data from much smaller and more

Figure 2-2: 24-Hour Rainfall in Millimeters Ending 8 a.m., July 13, 2004

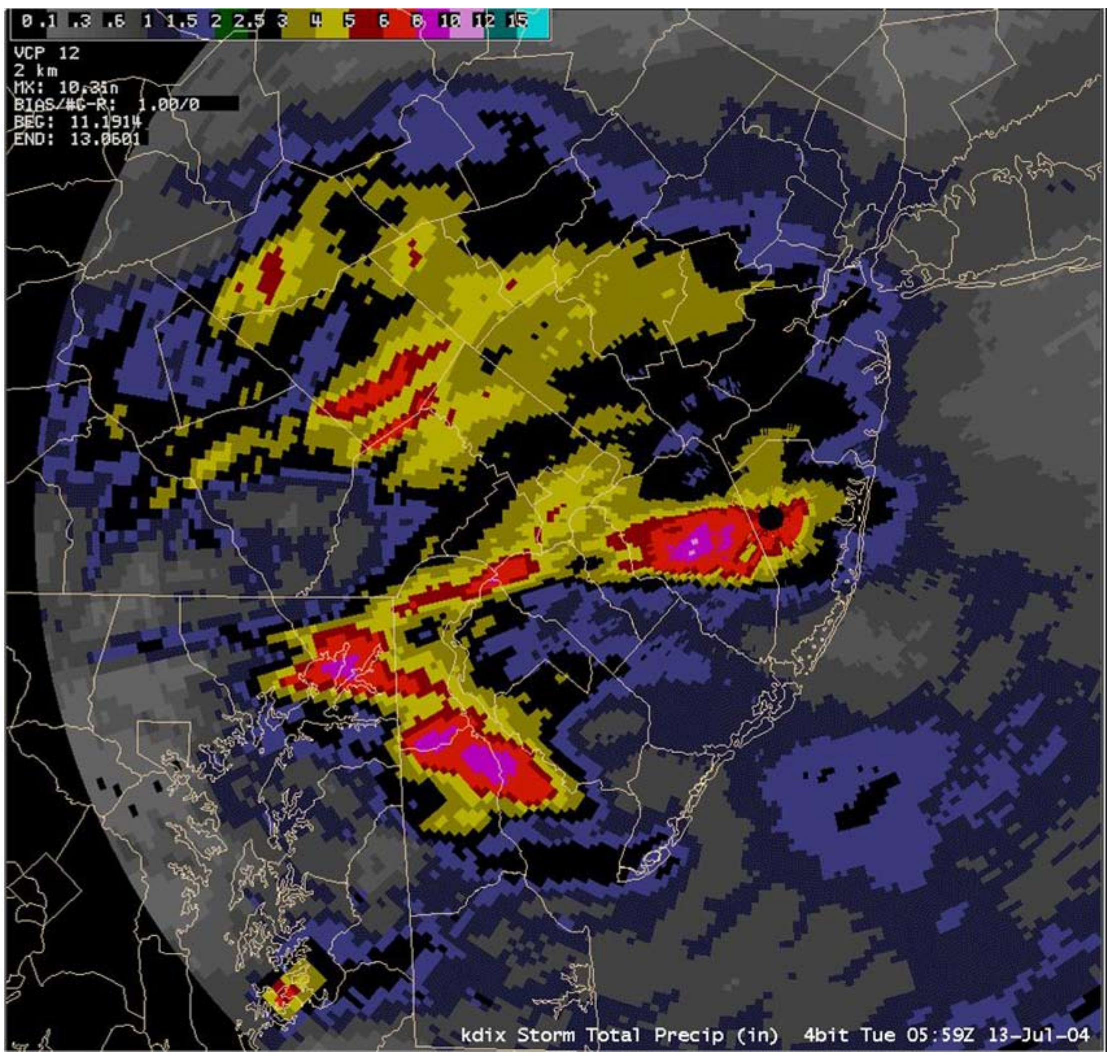


common rain events can also be used in the analysis and design of certain hydraulic structures. As described above, such rainfalls are not typically associated with structures intended to withstand the effects of a very large, rare rainfall event, such as a dam's spillway. Instead, they would be intended to reduce pollutant loadings in runoff and waterway flows or prevent surface or waterway erosion. Such rainfall data can also be used to evaluate the impacts that land development practices and policies have on producing pollution and erosion problems in the first place.

Figure 2-4 depicts the rainfall depth from approximately 750 storm events recorded at Newark Liberty International Airport in Newark, New Jersey between 1982 and 1992. It was taken from the long-term precipitation records contained in the computer program WinSLAMM – Source Loading and Management Model (Pitt and Voorhees, 1993). Such data can be used in programs like WinSLAMM and the EPA's Stormwater Management Model (SWMM) to estimate runoff amounts over the long periods of time which problems such as runoff pollution and erosion typically take to manifest. Assuming that the length and accuracy of the rainfall data is sufficient, structure designs and practice evaluations based upon such data can be considerably more robust than those based upon hypothetical or synthetic design storms (James, 1995).

This increased robustness is due to the uncertainties associated with the rainfall-runoff process noted above and the ways in which they are addressed differently

Figure 2-3: 24-Hour Rainfall in New Jersey-Delaware, July 12-13, 2004



Source: National Weather Service, Mt. Holly, New Jersey Forecast Office through the use of long-term rainfall records versus single-event design storms. When using a hypothetical design storm approach, decisions must be made as to the total amount of rain, how long it will fall, how it will vary in intensity (if at all) over this duration, how long it has been since the previous rain fell and, if significant, in what time of year the event will occur. Such decisions must be made by the designer or modeler, either actively through the development of an appropriate design storm or by default through the selection of a previously developed, standardized design storm often specified by an urban runoff management program. Selecting fixed values for each of these factors can and often will affect the resultant runoff estimate.

However, when using a suitably long and accurate record of actual rainfall, these decisions do not have to be made. Instead, the long-term rainfall record contains all of these factors, and its use allows them to vary over a naturally-occurring range of values. The result is a similarly varied series of runoff estimates that reflect this natural range of conditions. Analyzing this resultant runoff series with relatively simple statistical techniques can then produce results for a storm with a particular depth, frequency, duration, etc.

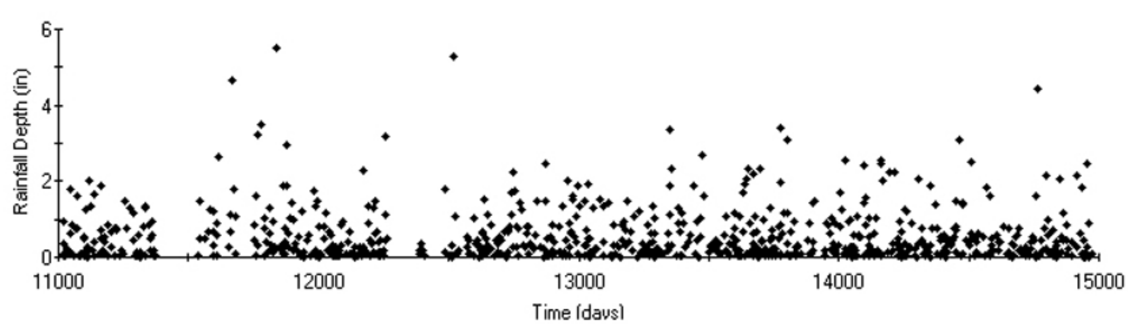
Despite this enhanced robustness or accuracy and its applicability to a range of analytic and design problems, the use of actual rainfall data, either from single, extreme events or over long time periods, is not without its problems. First and foremost is the availability of such data. While the number of recording rain gages in the United States is constantly increasing along with their reliability and data accessibility, there still remain many areas with inadequate gage coverage.

Second, the data record available must be sufficiently long for the intended use. Even the design of practices or facilities that must control the runoff from relatively high-frequency, low-depth rain events can require up to five to ten years of continuous rainfall data. The design of facilities such as dams and flood control works to control much lower frequency, higher recurrence interval events would typically require several decades of data at a minimum, unless one or more events in the available record can be accurately designated as statistically extreme. In these cases, such as the one illustrated in Figure 2-3, such extreme events may be used, with suitable caution, as design storms or, more typically, to supplement or evaluate the results produced by a hypothetical design storm.

Third, the data must have been recorded in time increments suitable for the event analysis or facility design in question. As explained more fully in following sections, rainfall data that has been recorded in time increments that approach or even exceed the length of time it will take for an area of land to respond to rainfall may be suitable for estimating total runoff volumes from rainfall events, but are generally not appropriate for predicting peak runoff rates or runoff hydrograph shapes. Use of such data can cause rounding and other errors that can lead to underestimated peak runoff rates, hydrographs, and, in certain models, runoff volumes (James and Robinson, 1982; Pitt and Voorhees, 2003).

An additional problem typically cited in the past with using actual rainfall data, particularly long-term records, was difficulty inputting, storing, and processing large amounts of rainfall data. It should be noted that this problem has been largely eliminated through the vastly larger data storage capacities and higher data processing

Figure 2-4: Rainfall Data for Newark, New Jersey, 1982 to 1992



Source: WinSLAMM Version 8.7

speeds of modern computers. If any computer-related problems remain in this area, it may be in the relatively limited number of computer programs that can accept long-term rainfall data.

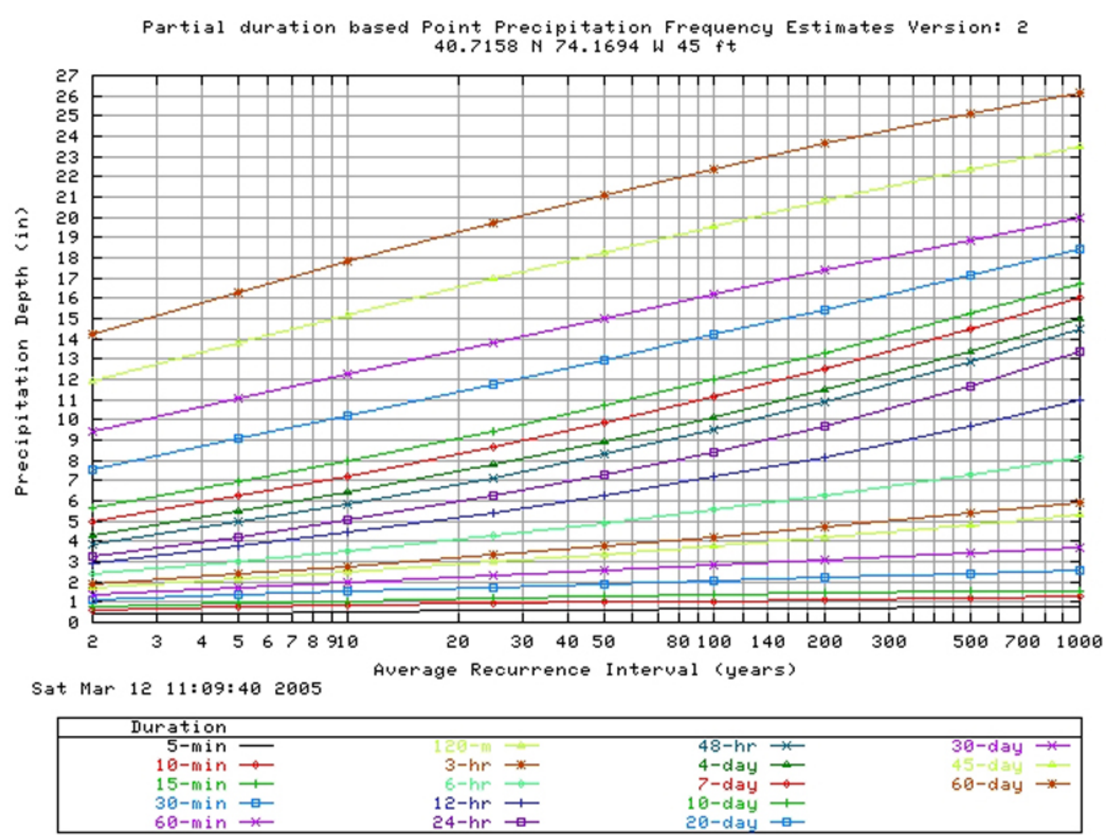
As a result, the use of hypothetical or synthetic design storms in urban runoff management programs remains relatively high. The data used to develop such storms is obtained from statistical compilations and extrapolations of real rainfall data collected over a statistically significant period of time. Figure 2-5 presents such a compilation. It depicts rainfall depth-duration-frequency curves for Newark, New Jersey based on hourly rainfall collected at Newark Liberty International Airport between 1948 and 2000. The curves predict the expected rainfall depth for a given period of rainfall and storm frequency, with the storm frequency expressed as an average exceedance probability in years. For example, the expected 100-year, 1-hour rainfall depth at the airport would be approximately 2.8 inches, while similar frequency storms for 2-, 6-, and 24-hour periods would have depth of approximately 3.8, 5.5, and 8.4 inches, respectively.

Similar curves can be developed for average rainfall intensity, which is obtained by dividing the rainfall depth by the rainfall period.

The curves in Figure 2-5 were developed by the Hydrometeorological Design Studies Center (HDSC) of the National Weather Service and were published in 2004 in the National Oceanic and Atmospheric Administration's (NOAA) *Atlas 14 – Precipitation Frequency Estimates*. Rainfall data for this and other U.S. locations is available at the HDSC Precipitation Data Frequency Center (PFDS) at http://hdsc.nws.noaa.gov/hdsc/pfds/. Additional rainfall data is also available through various publications and agencies throughout the country.

Rainfall data such as that shown in Figure 2-5 can be used in a variety of ways. If the total rainfall depth for a specific storm frequency and rainfall period is needed (for example, to estimate total runoff volume to a stormwater facility), the depth can be taken directly from charts or associated tables like the one in Figure 2-5. As described above, the depth can also be converted

Figure 2-5: Rainfall Depth-Duration-Frequency Data for Newark, New Jersey



Source: NOAA Atlas 14 – Precipitation Frequency Estimates to an average rainfall intensity in instances where a peak runoff rate is required (for example, to select the appropriate size of a storm sewer).

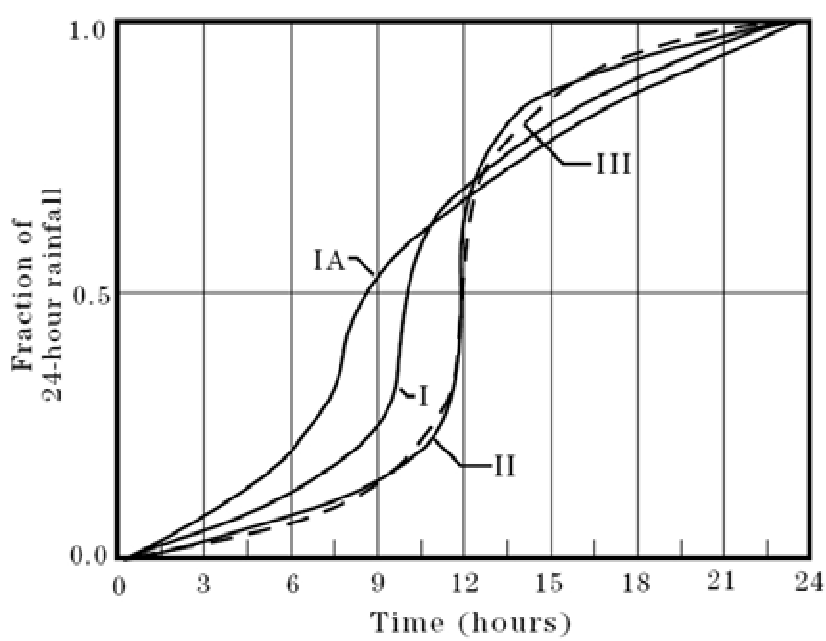
In addition, rainfall data like that shown in Figure 2-5 can be used to construct an entire hypothetical design storm. Such storms are typically needed when some or all of the runoff hydrograph (a depiction of how the runoff rate varies with time) is needed, not just the total runoff volume or peak runoff rate. Hydrographs are typically necessary for the analysis or design of any drainage area or stormwater facility where the variation of runoff rate over time is critical. Such areas include two or more subareas of a larger watershed that are added together to determine a combined peak rate or hydrograph. Time-sensitive stormwater facilities include wet ponds and detention basins.

The rainfall data in Figure 2-5 could be used, for example, to construct a 24-hour, 100-year hypothetical design storm for Newark by allowing the rain intensity to vary in such a way that the various 100-year rainfalls for durations less than 24 hours occur over the storm's total 24-hour duration. For example, such a storm would have maximum 1-, 2-, 6- and 12-hour rainfalls of 2.8, 3.8, and 5.5 inches respectively falling within its total 24-hour rainfall of 8.4 inches. It should be noted that, as shown in Figure 2-5, each of these rainfall-duration combinations have a 100-year frequency.

Figure 2-6 depicts the temporal distribution of four hypothetical design storms that are regularly used for drainage area runoff analysis and stormwater facility design. All four storms have varying rainfall intensities over their 24-hour length. They were developed by the Natural Resources Conservation Service (NRCS) of the U.S. Department of Agriculture and are used in NRCS rainfall-runoff methods and models. They have also been adopted for use by many urban runoff management programs throughout the country. Coordinates of the various NRCS design storm events can be obtained from the NRCS State Conservation Engineer in each state.

As shown in Figure 2-6, the rainfalls associated with each of the four NRCS hypothetical design storms is expressed as a percent of the total 24-hour rainfall. As such, an entire design storm for a given frequency can be computed simply by selecting a 24-hour rainfall depth with that frequency and applying it over the 24-hour period to the various rain depths in the appropriate design storm. An example of such a design storm with a 100-year frequency for Newark, New Jersey is shown in Figure 2-7. It was developed by multiplying the

Figure 2-6: NRCS Design Storm Distributions



Source: NRCS Technical Release 55

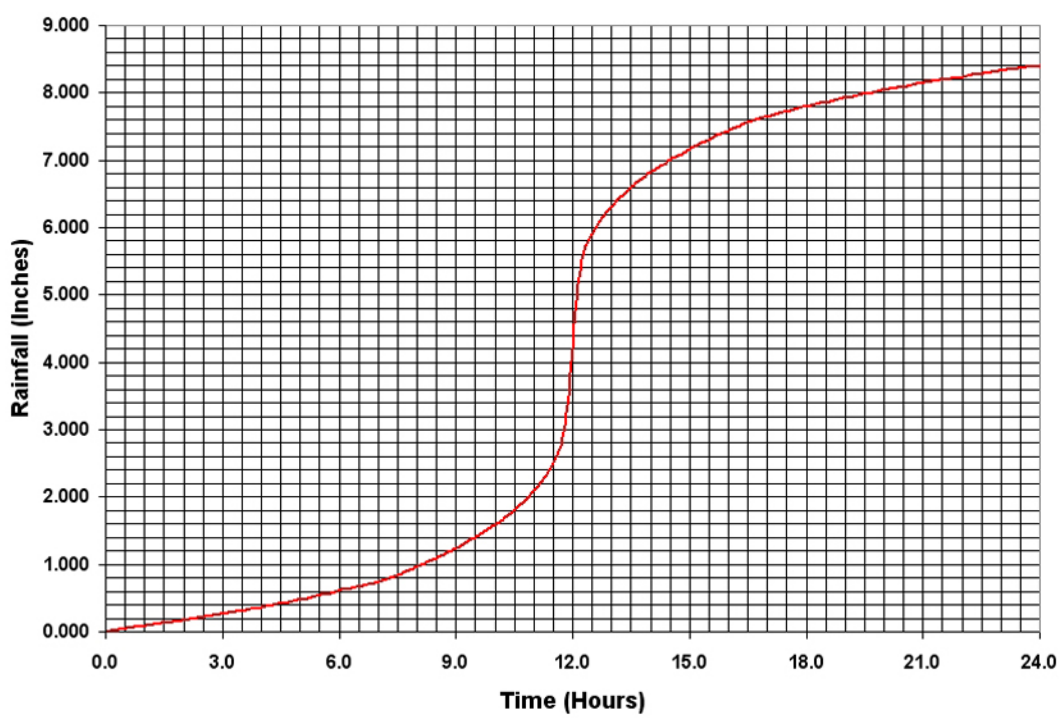
100-year frequency, 24-hour rainfall for Newark by the various rainfall depths shown in Figure 2-6 for the Type III design storm which the NRCS has designated as the most appropriate of the four design storms shown in Figure 2-6 for the city.

There are some interesting and helpful observations that can be made about the four different NRCS design storm distributions shown in Figure 2-6, all of which would have the same total rainfall at the end of the 24-hour event. First, it can be seen that the Types II and III storms are distributed more or less symmetrically about the storm's 12-hour midpoint, while Types I and IA are not. Second, in the Type II and III storms, the rain falls at lower intensities at the beginning and end of each storm (evidenced by the relatively flat slope of the curves between hours 0 and 9 and between hours 15 and 24) than the Type I and IA storms. As a result of these lower starting and ending rainfall intensities, the Type II and III storms have greater intensities during their middle periods and these high intensity periods last longer than the Type I and IA. In fact, as can be seen in Figure 2-7, fully 50 percent of the total rain depth of 8.4 inches falls in the middle two hours (between hours 11 and 13) of the Type III storm for Newark, New Jersey. Finally, the high-intensity rainfall periods in the Type II and III storms occur later than the Types I and IA. As a result of these differences, the Type II and III design storms can be expected to produce higher peak runoff rates than the Type I and IA storms for the same total 24-hour rainfall. This illustrates the complexities and influences that must be considered when developing or selecting a hypothetical design storm.

In addition to the four NRCS design storms, several other hypothetical design storm distributions have been developed and adopted by various jurisdictions and agencies with urban runoff management programs. These include the City of Austin, Texas; the State of New Jersey; the South Florida Water Management District, and the U.S. Army Corps of Engineers. And as additional rainfall data is collected and statistically analyzed, modifications to existing hypothetical distributions or the development of entirely new ones may be necessary in the future.

Finally, our discussion of rainfall would not be complete without mentioning rain that may have fallen during prior storms. While most of the runoff from a storm may have long since drained away, some is likely to still be present as soil moisture or stored in surface depressions in the drainage area. The exact amount of such water, referred to as the antecedent rainfall or

Figure 2-7: NRCS 100-Year Type III Design Storm for Newark, New Jersey



moisture condition, can influence the amount of runoff from a subsequent design storm by affecting how much of that storm's rain can infiltrate into the soil or be stored in the depressions. As such, its effect must be quantified in all rainfall-runoff computations.

Antecedent moisture conditions are particularly critical when recreating real storm events or analyzing both real and design storms with relatively low rainfall depths. For real storms, the antecedent moisture conditions can be estimated from the rainfall data for the antecedent period. When using a design storm, however, many runoff estimating methods assume for simplicity that average antecedent conditions exist in a drainage area prior to the start of the design storm. As a result, the frequency of the runoff event will equal that of the rainfall that produced it, an occurrence that is not always true. Such assumptions highlight the advantage of using long-term rainfall data, where the actual antecedent rainfall condition for each storm can be directly estimated from the prior event's data. More sophisticated methods allow the analyst to vary the antecedent condition to judge its sensitivity to the answer or to increase the conservatism or "safety" (as discussed above) of the answer.

In summary, the above subsection presented the following ideas and information:

- In estimating runoff, rainfall from both actual and hypothetical storm events may be used;
- Various hypothetical design storms have been developed and are used in many runoff estimation methods and runoff management programs;
- Hypothetical design storms can produce reliable results, particularly for large, relatively infrequent storms where the depth of the rainfall dominates the rainfall-runoff process;
- Conversely, design storms may be less reliable for smaller, more frequent storms where antecedent rainfall, climate, soil type, slope, and cover have greater influence on the resultant runoff;
- Design storms may need periodic updating or replacement as additional rainfall data is collected and analyzed;
- Data from actual rain events may be used to supplement or check design storm results;
- Suitable, actual rain data may also be used for design purposes, provided it represents a sufficiently long period of time or severity of storm;
- The use of long-term rain data to estimate runoff from smaller, more frequent storms is increasing as more suitable data and computer models become available; and
- Long term rain data may also serve as an indicator of climate change on rainfall. If verified, such effects must be taken into consideration when using such data.

Time

Time plays a critical role in the actual rainfall-runoff process and, as such, plays a similar role in the various theoretical methods used to simulate it. This is not surprising, since the gravitational, thermodynamic, and other natural forces involved in the creation of runoff from rainfall are constantly changing with, and therefore influenced by, time. These influences can be exceptionally complex. The following discussion presents a simplified description of how time affects runoff estimates.

Two fundamental measures or lengths of time are important when performing runoff estimates from rainfall. The first is the runoff response time of the drainage area to a rainfall input. This response time indicates how quickly the runoff created by a given amount of rain drains to the outlet of the drainage area and how quickly the rate of that runoff will change as the rainfall rate changes. In more sophisticated estimating methods, this response time may also affect the volume of runoff produced by the rain.

Several terms and definitions can be used to describe this response time; most are applicable to a particular runoff estimating technique. The most common term is Time of Concentration (TC), which the USDA Natural Resources Conservation Service and others define as the time it takes runoff (once it has begun) to flow from the most distant point in the drainage area to the drainage area's outlet. Numerous procedures, equations, and nomographs are available for estimating TC, including those presented in Chapter 3 of the *NRCS Technical Release 55 – Urban Hydrology for Small Watersheds* (TR-55), which is used as the hydrologic basis of many urban runoff management programs.

Regardless of the method used to estimate TC, it is important to recognize its direct effect on the resultant rate of runoff, including the peak rate. As noted above, TC is a measure of how quickly the runoff from a given amount of rain throughout a drainage area can flow to the area's outlet. Stated differently, it represents how much time it takes the runoff produced throughout the drainage area to concentrate at the outlet. The more quickly a fixed volume of runoff can concentrate at the outlet, the more runoff will exist at any point in time at that outlet. As such, the TC will directly affect the overall shape of the runoff hydrograph, including the peak runoff rate. The shorter the TC, the higher the runoff rate, including the peak. In light of these effects, it can be seen that whether we seek to estimate a peak runoff rate or an entire runoff hydrograph for a given rainfall, we must compute a reasonably accurate estimate of TC.

In computing runoff peaks and hydrographs, TC can also assist us in another way. Since most rainfall data, whether for a real event or hypothetical design storm, is rarely provided in a continuous form over time but rather in discrete time increments, we must assume an average rate of rainfall will occur during each of these time increments. Since TC is a measure of how quickly the rate of runoff will vary due to changes in rainfall rate, we can use it to determine how small of a time increment we must divide our rain event into to produce an accurate runoff peak or hydrograph.

For example, a drainage area that takes six hours to respond at its outlet to rain falling within it will show little change in runoff rate from a change in rainfall intensity lasting only a few minutes. Therefore, using a time increment of 30 to 60 minutes (during which rain is assumed to fall at an average rate) would be appropriate. However, using a 30-minute time increment for a drainage area that responds in 15 minutes would not be appropriate, since the assumption of a uniform rainfall rate during each 30-minute storm increment would mask any shorter-term variations in rainfall rate that would have a significant effect on the resultant runoff rate. Such time increment-induced errors are examples of the "rounding errors" described above that may occur in the use of actual rainfall data. This also illustrates the problem that can be encountered when attempting to find actual rainfall data in sufficiently short time increments.

The second fundamental period of time in rainfall-runoff computations is the effective event time. When computing only a peak runoff rate from a drainage area, this time is typically based upon the time the area can respond to rainfall and, as a result, can be set equal to the drainage area's TC. When performing such computations, therefore, we are interested only in a period of rainfall within a longer storm event; namely, the period with the greatest rainfall rate or intensity. For example, if we wish to estimate the peak 10-year rate of runoff from a drainage area in Newark, New Jersey with a 30-minute TC, we would use a 10-year recurrence interval, 30-minute rainfall of 1.5 inches from Figure 2-5.

However, if we wish to estimate the total runoff volume for a 10-year storm event, the effective event time will have to include the entire storm duration in order to obtain the total rain depth. While such times are readily available when using data from actual rain events, they must be carefully selected when using a hypothetical design storm. For example, while Figure 2-5 indicates that a 10-year, 1.5-inch period of rainfall would last for 30 minutes (see previous paragraph), it gives no indication of the total duration or depth of the storm in which that 1.5-inch, 30-minute rainfall would occur, other than the fact that it would last for at least 30 minutes. However, it could also be part of a longer, much larger storm event.

In addition, when designing certain runoff treatment or control practices such as infiltration basins, the effective event time may also include some additional period of time following the end of the rainfall event. This additional time, known as the inter-event dry period (Wanielista and Yousef, 1993), reflects the time by which the practice artificially prolongs or extends a drainage area's response time (through its slow release of stored runoff) and, therefore, the effective event time. As a result, when developing or selecting an appropriate hypothetical design storm to estimate total runoff depth, judgment must be used to ensure that the total event time is appropriate for the design or analysis at hand.

In summary, the above subsection presented the following ideas and information:

- Time plays a critical role in the rainfall-runoff process and the various methods and models used to simulate it.
- This role includes influencing the various rates of runoff that may occur during a rain event, including the peak runoff rate, and, in certain methods, the total volume of runoff.
- There are two fundamental lengths of time that are important when performing rainfall-runoff computations.
    - The first one is the time a drainage area takes to respond to the rain falling within it. This time, typically expressed as the

area's Time of Concentration, can be used to both estimate peak runoff rates and determine the maximum time interval that rainfall data should be divided into to produce reliable hydrograph estimates.
    - The second one is the effective rainfall event time. When estimating peak runoff rates, this time is typically based upon a drainage area's rainfall response time as expressed by its Time of Concentration.
- When estimating total runoff volume, however, the effective event time must span the entire rainfall event in order for a total rainfall depth to be obtained.
- When designing runoff management practices such as infiltration basins that artificially extend an area's response time, the effective event time may include an additional period of time beyond the total rainfall duration known as the inter-event dry period.

Drainage Area

The concept of drainage area is fundamental to any rainfall-runoff analysis. It is the area that contributes runoff to a particular point in a drainage system typically referred to as the drainage area's outlet. For this reason, it may also be known as a watershed, since it represents the area that "sheds" water or rainfall to the outlet. However, this term is typically applied to larger areas draining to streams and rivers. Catchment is another term used at times instead of drainage area, as it represents the area that "catches" rainfall and delivers a portion of it as runoff to the outlet.

Both a drainage area's size and various characteristics about its soils, cover, slope, and response time are typically used to estimate runoff from rain falling within it. Of these, the drainage area size is a primary consideration. It is usually determined from a combination of topographic maps, waterway and storm sewer plans, and field reconnaissance. Most runoff estimating methods assume a linear relationship between drainage area and runoff volume. Therefore, a 20 percent error in estimating a drainage area's size will, among other impacts, directly result in a similar error in the estimated runoff volume. This relationship is important when determining the required accuracy of drainage area size computations and the required time and effort to achieve it.

Two important drainage area characteristics for estimating runoff are its shape and slope. As discussed above, a drainage area's response time will influence the rate of runoff from a given rain event, with shorter response times producing greater runoff rates than longer ones. A drainage area with generally steep slopes can therefore be expected to respond faster to rainfall and concentrate a greater amount of runoff over a given period of time. Similarly, the length that the runoff must travel to the drainage area's outlet can also affect the response time, with elongated drainage areas with relatively longer travel lengths typically producing lower runoff rates than more rounded ones with shorter travel lengths.

It is important, however, to avoid over-generalizing the effects of drainage area shape and slope on runoff rates, particularly for complex drainage areas and watersheds with multiple branches or tributaries. Each drainage area within an overall watershed has its own unique shape, slope, flow length, and complexity, all of which can have a direct effect on response time and resultant runoff rates. Therefore, a representative response time, typically expressed as its Time of Concentration, should be estimated as accurately as possible for each drainage area based upon these characteristics.

The variation in ground surface within portions within a drainage area, particularly those that create surface depressions and other irregularities, can also have a direct effect on the area's response time, runoff rate, and even runoff volume. Depending upon their depth and size, surface depressions can slow the rate of runoff movement and concentration as well as store a portion of the runoff. This not only increases the drainage area's peak runoff rate but the runoff volume as well. Such runoff delays and storage, in combination with such factors as antecedent rainfall, surface wetting, soil infiltration, and interception by vegetation, typically are greatest at the inception of rainfall and as such produce an effect known as initial abstraction. This is the amount of initial rainfall that must occur before runoff at the drainage area outlet begins. Depending on a drainage area's surface depressions and irregularities, along with its soils and covers, the initial abstraction can significantly affect the volume of runoff and the size and timing of its peak rate. Therefore, the effects of initial abstraction should not be overlooked, particularly for small rainfall depths where the initial abstraction amount is a significant percentage of the total rainfall.

In summary, the above subsection presented the following ideas and information:

- The concept of a drainage area that catches rainfall and drains the resultant runoff to its outlet is fundamental to runoff estimation.
- Most runoff estimation methods assume a linear relationship between drainage area size and runoff volume.
- In general, the slope and shape of a drainage area can influence the rate of runoff, including the peak rate.
- Localized surface irregularities, in combination with soil and cover characteristics, can store or abstract an initial amount of rainfall and both delay the start of runoff and reduce runoff volume and rates.

Soils

The surface and subsurface soils within a drainage area can play a direct role in determining the volume and rate of runoff from rainfall. As a result, various soil characteristics are included in most runoff estimating methods. These characteristics include the texture, structure, permeability, thickness, and moisture content of the various layers within the soil profile. Soil texture, structure, and thickness can help determine how much rain a soil can absorb and retain, with granular soils such as sands possessing greater storage capacity than silts and clays. Similarly, a thin layer of soil on top of bedrock will have less storage capacity than a deeper soil with similar texture. Permeability will affect the rate at which rainfall can enter and move through a soil and, therefore, the volume and rate of any resultant runoff. A soil's moisture content at the start of rainfall is not only a measure of its available storage capacity but can also influence its permeability rates. Rain falling on a pervious drainage area whose soils are saturated from antecedent rain events can produce runoff volumes and rates similar to a drainage area that is largely impervious.

Soil texture, permeability, and thickness data can be found in numerous sources, including laboratory tests of soil samples taken from various drainage area locations. County Soil Surveys, developed cooperatively by the USDA Natural Resources Conservation Service and various state agencies, are generally reliable sources of such information. Depending upon the Survey date, the drainage area size, the required degree of accuracy, and the sensitivity of soil characteristics in the selected runoff estimation method, field verification of Soil Survey information may be necessary. Such verification can also be used to assess soil structure, which can also influence resultant runoff amounts.

The relationship between soil texture and permeability should be noted. The relatively large percentage of void space within granular soils such as sands creates not only significant storage volume but also relatively high permeability rates. As a result of these two features, sands can be expected to produce less runoff volume than silts or clays, which have less void space and permeability. In certain instances, this relationship can permit a soil's permeability to be estimated from its texture.

As discussed above, soil permeability, texture, and moisture content in combination with vegetation and surface depressions and irregularities can also affect the amount of initial rainfall that is abstracted before runoff begins. This initial abstraction can significantly affect the volume of runoff and the size and timing of its peak rate. Therefore, the effects of drainage area soils on initial

Table 2-1: Summary of Ocean County, New Jersey Soil Compaction Study Results

Study Site	Mean Bulk Density (g/cm3)	Mean Permeability (in/hr)
Woods	1.42	15
Cleared Woods	1.83	0.13
Subdivision Lawn 1	1.79	0.14
Subdivision Lawn 2	2.03	0.03
Athletic Field	1.95	0.01
Single House	1.67	7.1
Source: Ocean County Soil Conservation District et al., 1993		

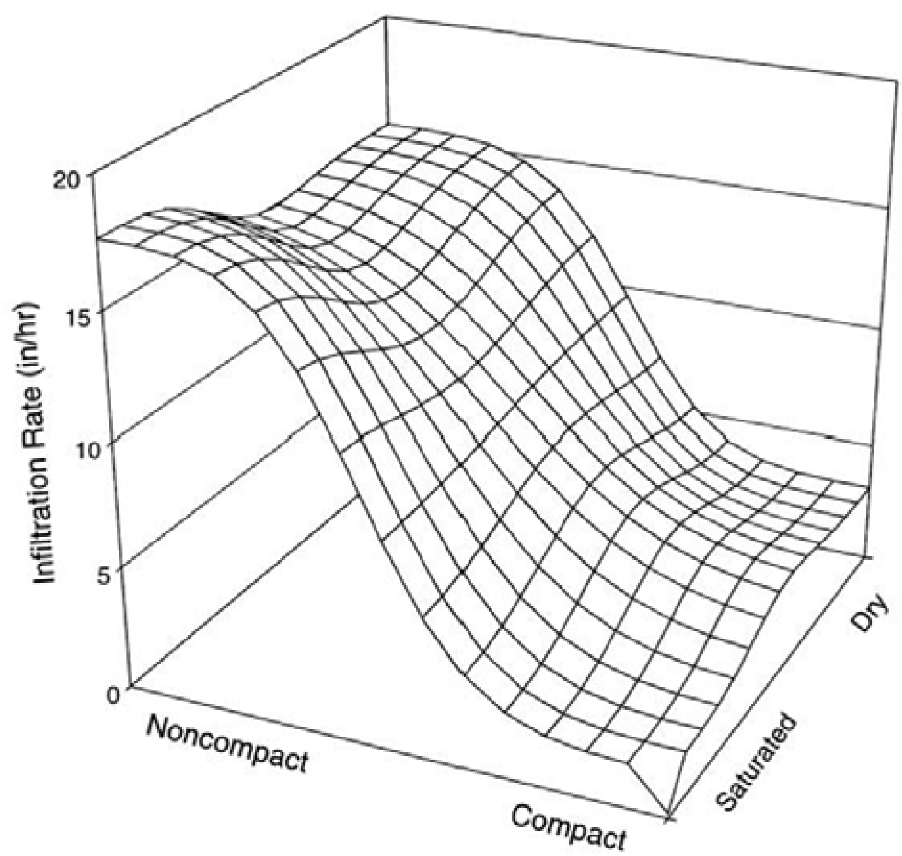
abstraction should not be overlooked, particularly for small rainfall depths.

Finally, research continues to confirm that compaction can significantly modify or damage a soil structure, resulting in decreasing storage volumes and permeability rates and increased runoff. Research conducted in New Jersey (Ocean County Soil Conservation District et al., 2001) demonstrated that soils compacted either by construction equipment or post-construction use can experience significant reductions in permeability. A summary of this research is shown in Table 2-1. It compares the bulk density (as a measure of soil structure) and permeability rates of soils with generally similar sandy soil textures at various sites. The Woods site shown in the table represents an undisturbed condition with natural soil structure. The Cleared Woods site represents a disturbed condition where the vegetation and organic ground layer have been cleared by heavy equipment without significant regrading. The Subdivision Lawns 1 and 2 and Athletic Field sites represent highly disturbed areas where both clearing and regrading by heavy equipment have occurred. The bulk density and permeability values summarized in the table are the mean of three replications in a soil layer 20 inches below the surface.

As shown in the table, the mean soil permeability of the Cleared Woods and Subdivision Lawn 1 are approximately 100 times lower than the 15 inches per hour mean permeability at the undisturbed Woods site. Greater reductions can be seen at the Subdivision Lawn 2 and Athletic Field sites, where mean permeabilities ranging from 500 to 1000 times lower than the Woods site were measured. The mean permeabilities for the various disturbed sites are similar to those found for impervious areas such as roads, highways, and parking lots (Pitt, 1991).

Further research in Alabama into the effects of compaction on both sandy and clayey soils (Pitt et al.,

Figure 2-8: Alabama Compaction Test Results for Sandy Soils



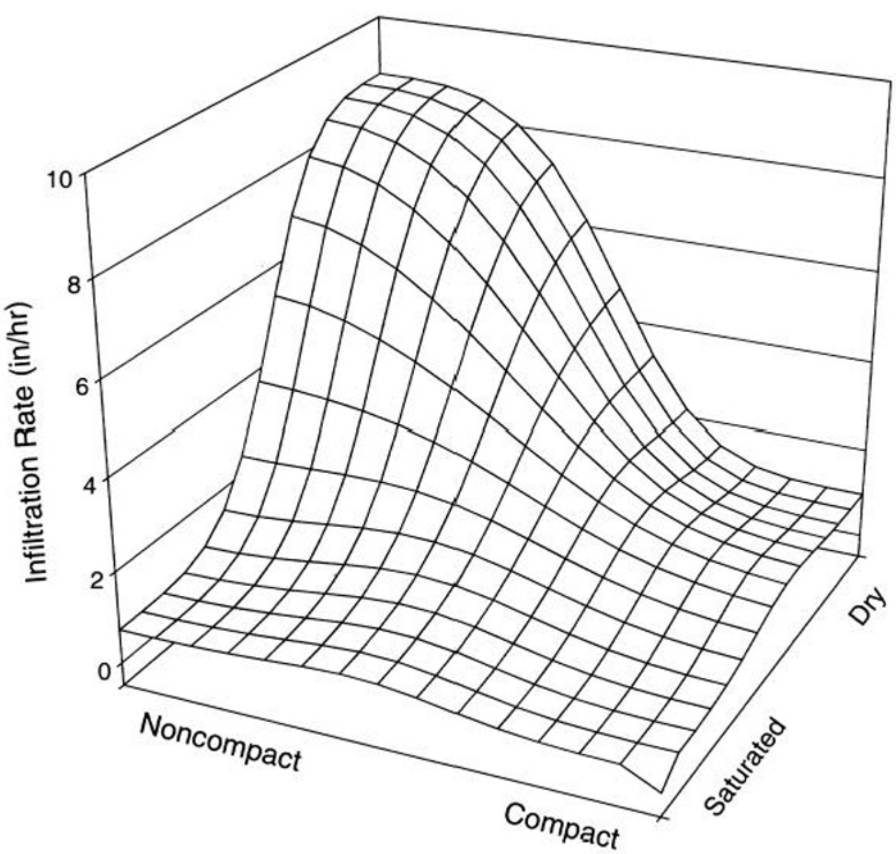
Source: Pitt et al., 1999

1999) confirmed the impacts to sandy soils previously demonstrated in Ocean County. Based upon more than 150 infiltration tests in disturbed urban soils, this research also demonstrated that such effects were generally independent of soil moisture in such soils. However, the research also found that, while compaction had similar effects on clayey soils with low moisture content, these effects were of minor significance when the moisture content approached saturated levels. A graphical summary of this research is shown in Figures 2-8 and 2-9.

From the results shown in Table 2-1 and Figures 2-8 and 2-9, it is felt that the effects of compaction on the rainfall-runoff process can no longer be ignored, particularly for sand and other coarse grained soils. As a result, inclusion of appropriate factors in runoff estimation methods is warranted when predicting runoff from a future, developed drainage area with such soils. However, this may require additional research data in order to reliably predict the degree of expected compaction and its impacts on soil permeability and runoff. Further study of the long-term effects of compaction, and whether natural weathering processes can restore some or all of the lost soil structure and permeability, are also required. Until such research is concluded, the results of the New Jersey and Alabama studies and a conservative or "safe" design approach may be used as guidance.

Potential measures to address the adverse impacts of soil compaction may be found in the results for the Single House site shown on the bottom row of Table 2-1. According to the Ocean County Study report, this site was not constructed through widespread regrading with heavy equipment typical of large tract construction, but instead through limited regrading with relatively light construction equipment. According to the results in Table 2-1, the lawn area at this site had a mean permeability rate of 7.1 inches per hour. While this is less than half the tested mean of 15 inches per

Figure 2-9: Alabama Compaction Test Results for Clayey Soil



Source: Pitt et al., 1999

hour for woods, it nevertheless represents a relatively high permeability rate, particularly in comparison with the other, more highly disturbed sites in the table. This relatively high disturbed site permeability rate may indicate that the adverse impacts of compaction may be avoided or reduced through the use of site design techniques and construction practices and equipment that minimize site disturbance, regrading, and construction equipment weight and movement.

The Alabama research also presents a potential measure to address soil compaction through the addition of large amounts of compost to the soil. Tested on a glacial till soil, this measure was shown to significantly increase soil permeability at the expense, however, of an increase in nutrients in the runoff. Such soils also produced superior turf with little or no need for maintenance fertilization.

In summary, the above subsection presented the following ideas and information:

- Soil characteristics such as texture, permeability, and thickness can greatly influence the rainfall-runoff process and are therefore included in most runoff estimation methods.
- These characteristics can affect both the amount of initial rain that must fall before runoff begins and the total volume and peak rate of runoff.
- The general relationship between soil texture and permeability may allow the latter to be estimated from the former.
- Soil moisture content at the start of rainfall can significantly modify a soil's storage capacity and permeability rate.
- Compaction can also significantly modify a granular soil's undisturbed storage capacity and permeability rate.

Land Cover

In addition to the soils at and below the land surface within a drainage area, the type of cover on the soils' surface directly affects the rainfall-runoff process and is an important factor in most runoff estimation methods. Land covers can range from none (i.e., bare soil) to vegetated to impervious. Important vegetation characteristics include type, density, condition, extent of coverage, degree of natural residue or litter at the base, and degree of base surface roughness. Important impervious surface characteristics also include surface roughness, age and condition, connectivity, and the presence of cracks and seams. Connectivity describes whether runoff from an impervious surface can flow through a direct connection to a downstream swale, gutter, pipe, channel, or other concentrated flow conveyance system, or whether the runoff can flow onto and be distributed over a downstream pervious area, where a portion can infiltrate into the soil. As a result, unconnected impervious surfaces typically produce less runoff volume than directly connected ones.

All of the above characteristics can affect the volume of resultant runoff by influencing the amount of rainfall that is either stored on the land and vegetated surfaces or infiltrated into the soil. These characteristics can also affect a drainage area's response time or Time of Concentration and, consequently, the rate and duration of runoff. For example, TC equations developed by the NRCS indicate that runoff flowing as sheet flow across relatively smooth impervious surfaces will travel approximately 10 times faster than it would across a wooded area. The surface storage and delaying effects of land cover, particularly vegetation, can also help increase the amount of initial abstraction, thereby decreasing the runoff volume from a drainage area.

Land cover data sources, frequently used in combination, include field reconnaissance, aerial photographs, satellite imagery, and geographic information system (GIS) databases for existing drainage area conditions. Land cover under proposed or future conditions can be estimated from zoning maps, development regulations, proposed land development plans, and build-out analyses.

In estimating runoff from rainfall, it is interesting to compare the different responses from the impervious portions of a drainage area with those with pervious land covers such as turf grass, woods, or even bare soil. At the start of rainfall, the initial abstractions of both the impervious and pervious surfaces must be overcome before runoff begins. While the initial abstractions for typical impervious covers like roofs, roadways, parking lots, and sidewalks are considerably less than for areas with pervious covers, they nevertheless exist (Pitt and Voorhees, 1993). However, having a lower value, the initial abstraction for the impervious surfaces is overcome first, and the impervious surfaces will begin to produce runoff. This will continue until the larger initial abstraction of the pervious covers is also overcome. At this point, both the impervious and pervious portions of a drainage area will produce runoff.

Once runoff has started, it is generally accepted that its amount will increase exponentially as rainfall continues. This nonlinear relationship between rainfall and the runoff it produces is more pronounced for pervious land covers than impervious ones, which typically have a near constant or linear rainfall-runoff response once runoff begins. These different initial abstractions and rainfall-runoff responses result in the relative percentage of runoff produced from each type of cover varying considerably, depending upon the total rainfall amount.

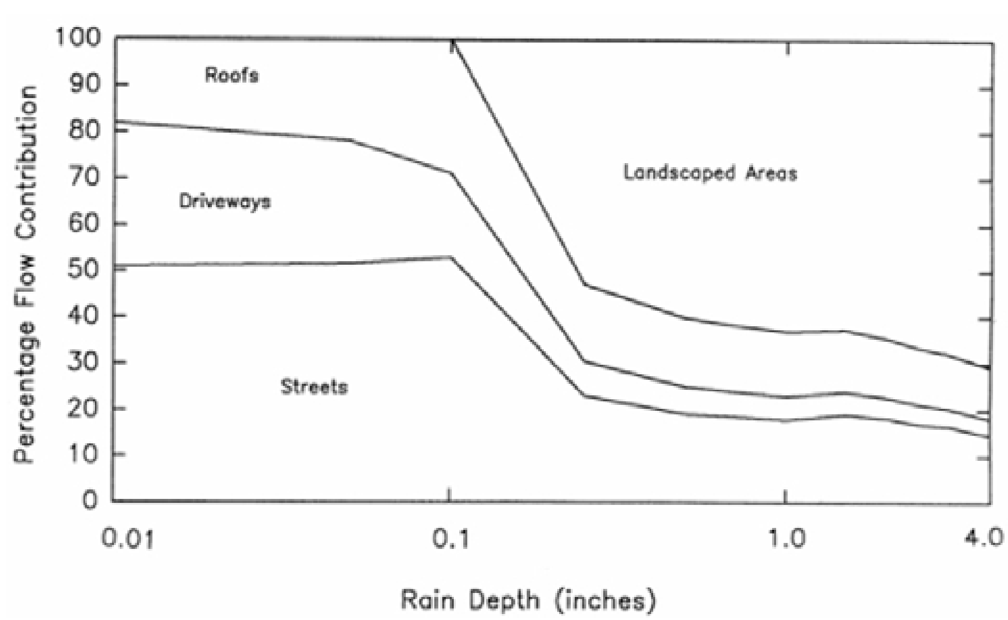
This difference is illustrated in Figure 2-10. It depicts the relative percentage of total runoff volume produced for a given amount of rain from various runoff source areas at a typical medium density residential housing site with clayey soils. As shown in the figure, site runoff would be entirely comprised of runoff from those site areas with impervious covers (i.e., streets, driveways, and roofs) from the start of rainfall until approximately 0.1 inches have fallen. However, as rainfall continues and overcomes the initial abstraction of the site's pervious landscaped areas, runoff from these areas also begins. When the rainfall has reached approximately 1 inch, approximately 50 percent of the site runoff is produced by these pervious areas. This increase in runoff percentage continues as rainfall continues, reaching approximately 70 percent at a total rainfall depth of 4 inches.

Such relationships are useful to urban stormwater management programs because they identify the critical runoff source areas that have the greatest impact on various program objectives. If a program objective is to address the runoff quality and pollution impacts caused predominantly by small, frequent rainfalls, then the control of impervious surfaces and the runoff from them is important. If flood or erosion control is critical, then all land covers may be important, since they all contribute important percentages of the total site runoff during the larger rainfall normally associated with these types of problems.

In summary, the above subsection presented the following ideas and information:

- The type, character, extent, and condition of the various land covers within a drainage area can have a significant effect on initial rainfall abstractions and resultant runoff volumes, rates, and durations.
- There are typically many sources of land cover data, including aerial photographs, GIS databases, field reconnaissance, and land development plans.

Figure 2-10: Relative Runoff Contributions from Various Source Areas at Medium Density Residential Site with Clayey Soils



Source: Pitt and Voorhees, 1993

- Pervious and impervious covers respond differently to rainfall. The relative percentage of the total runoff from each varies with the total amount of rainfall.
- Impervious areas typically produce the majority of runoff from small rainfalls, while the percentage from pervious areas typically increases with increasing rainfall.
- Runoff from impervious areas can also vary, depending upon their roughness, condition, and connectivity. Directly connected impervious areas can produce significantly more runoff volume than unconnected ones.

Runoff Estimation: Methods and Models

There are numerous methods currently available to estimate runoff from rainfall. In general, most methods will include some if not all of the parameters described in the previous section. Exactly what method to utilize and what parameters to include typically depends upon available parameter data and the desired results.

Using desired results as a basis, runoff estimation techniques can be broadly grouped into the following three categories:

1. Runoff Volume Methods
2. Peak Runoff Rate Methods
3. Runoff Hydrograph Methods

Each category will generally utilize certain parameters and equations and, therefore, will require certain types and ranges of data. A brief description of each category is presented below. As can be seen from the descriptions, the number of parameters increases as we proceed down the list.

Runoff Volume Methods

When an estimate of runoff volume is desired, typical parameters include total rainfall, drainage area size, and soil and land cover characteristics. Soil characteristics will generally include estimates of initial abstraction amounts, soil infiltration rates, and some measure of antecedent moisture condition. Infiltration rates may be fixed at a constant rate or may vary throughout the event, typically in an exponential manner. A more sophisticated method may include consideration of drainage area slope. A similarly sophisticated method may also include rainfall intensity and total storm duration, although, in general, time-based parameters are not included, particularly those based upon a single design storm. However, runoff volume estimating methods which utilize long-term rainfall data will typically consider time in the form of interevent dry periods and the amount of soil moisture depletion that may occur during each one.

Peak Runoff Rate Methods

Methods that produce estimates of peak runoff rate from a given storm event typically include all or most of the parameters utilized in runoff volume methods. However, as the term "rate" implies, time plays a more important role and, consequently, more time-based parameters are typically included. These include an estimate of the drainage area's Time of Concentration as well as the peak rainfall intensities over this period. Simplified methods utilize a single, average rainfall intensity over the entire TC while more sophisticated ones allow the use of several, shorter-term intensities within the overall TC.

Runoff Hydrograph Methods

When an entire runoff hydrograph is desired, additional time-based parameters and data are required in addition to the parameters used in runoff volume or peak runoff rate methods. First, since a runoff hydrograph is a measure of runoff rate resulting from all or a portion of a rainfall event, rainfall data throughout the entire event is required, typically divided into time periods equal to at least 20 percent to 25 percent of the drainage area's TC. In addition, some measure of the movement of runoff through the drainage area over time is also required. Once again, simplified methods typically assume a linear relationship, while more sophisticated ones utilize a nonlinear one based upon such mathematical techniques as unit hydrographs and kinematic wave equations.

In comparing the above descriptions of the three general runoff estimation methods, several observations can be made. First, as noted above, the number of time-based parameters increases as we move from estimating runoff volume to peak rates and then entire runoff hydrographs. This relationship can tell us which type of method is needed when designing or analyzing a particular stormwater management facility or practice. That is, a stormwater management measure such as an infiltration basin that is relatively insensitive to the rate of runoff inflow can often be designed from estimates of total runoff volume only. However, designing a stormwater facility such as a detention basin that is sensitive to the rate of runoff inflow will typically require a runoff hydrograph.

This relationship between stormwater facility type and required runoff method can also guide us toward the type of rainfall data that may be utilized in facility design. Since records of total storm depth are generally more available than records of incremental rainfall over short time increments, an infiltration basin designer will be much more likely to have a choice between actual long-term rainfall data and a single design storm approach than a detention basin designer would. Similarly, the designer of a stormwater facility to control the runoff from relatively small, frequent rainfalls is more likely to be able to choose between a long-term data and a design storm approach than the designer of a facility to control runoff from large, relatively infrequent events. This is because the first designer requires a relatively short period of rainfall record, which is presently more available than the longer-term records required by the second designer.

In addition, as noted above, the number and range of included parameters increases as we move from the runoff volume estimation methods to the runoff peak and then the runoff hydrograph methods. This increasing data and computational complexity can also signal a decrease in the certainty of the estimates produced by these methods. As a result, whether using long-term data or a single design storm approach, we can generally expect our estimates of total runoff volume to be more reliable and accurate than our estimates of peak runoff rate and, to an even greater extent, entire runoff hydrographs. This realization should guide our selection of design parameters and facility features so that the inherent safety of the facility design increases with decreasing estimation certainty.

Finally, as our concerns for runoff quality and the environment have grown, there has been an increasing interest in estimating the runoff from relatively small rainfalls. In recognition of this interest, it is important to note a second categorization of runoff estimation methods that is based upon the range of applicable rainfall depths. At the time of the 1994 publication of the original *Fundamentals of Urban Runoff Management*, the NRCS Runoff Equation and its variants had become the standard method for estimating runoff volume from rainfall. As clearly stated in various NRCS publications such as TR-55, this method was and remains intended for runoff depths of 0.5 inches or more. In many instances, this would require a minimum total rainfall depth of approximately two to three inches which, in many locations, would have an average frequency or recurrence interval of one year or more.

While these rainfall depths and frequencies typically represented the lower limits of interest or jurisdiction of runoff management programs in 1994, research and experience has pointed toward the need to manage the runoff from smaller rainfall amounts in order to optimize control of runoff quality and water ecosystem problems (Pitt and Voorhees, 1993). Therefore, it has likewise become important to develop and utilize newer runoff estimation methods suitable for these lower rainfall depths. Equations such as those developed by Pitt and Voorhees and by the Center for Watershed Protection for the State of Maryland have been shown to be particularly reliable for such rainfall depths. Use of the NRCS Runoff Equation for runoffs less than the official NRCS limit, which may be necessary in certain existing runoff management programs and computer models, should only be made with caution and a thorough understanding of the method's assumptions, limitations, and sensitivities. Similar caution should be used when using a method intended for small rainfalls to estimate runoff from larger events.

In summary, the above section presented the following ideas and information:

- Runoff estimation methods can be categorized by the type of result they produce.
- In general, the three basic method types are those that estimate runoff volume, peak runoff rate, and runoff hydrographs.
- Each method utilizes a certain combination of parameters, equations, and assumptions.
- As you proceed from estimating runoff volume to peak runoff rate and then runoff hydrographs, the degree of complexity and range of parameters typically increases as well, particularly of those associated with time.
- This increased complexity can also signal a decrease in reliability of results, indicating the need for increased discretion and data accuracy

to ensure effective and safe stormwater facilities and practices.

- The type of estimation method required to design a stormwater facility will depend upon the facility's sensitivity to changes in inflow over time.
- Methods that utilize long-term rainfall data and single design storms are both available. Which approach can be utilized will depend upon the range of rainfalls to be controlled, the facility's sensitivity to time, and the availability of suitable rainfall data and computer programs.

Impacts of Land Use Change

Typically, a land development project will result in modifications to several of the factors associated with the rainfall-runoff process. These can include replacing indigenous vegetation with both impervious land covers and planted vegetated covers such as turf grass. Such land covers are less permeable and have fewer surface irregularities and surface storage, resulting in increased runoff volumes and longer runoff durations. This problem may be compounded by increases in drainage area size through surface regrading and conveyance system construction, which can make a larger area contribute runoff to a particular location. Soil compaction during construction may further increase the volume of runoff from the turf grass and other constructed pervious areas.

The land cover changes described above can also cause significant reductions in initial abstraction, creating a lower rainfall threshold in order for runoff to begin. This lower threshold can be particularly damaging, for it results in runoff to downstream waterways from rainfalls that previously did not produce any runoff, hypothetically causing an infinite increase in the runoff from such rains. This also compounds the increased runoff volume impacts by creating a greater number of runoff producing storm events and increasing the frequency of runoff and pollutant loadings in downstream waterways.

In addition to being less permeable, impervious and turf grass land covers are typically more efficient in transporting runoff across their surfaces, resulting in decreases in a drainage area's Time of Concentration and a corresponding increase in runoff rates, including the peak runoff rate. Such increases, which can be compounded by the replacement of natural conveyance systems with more efficient constructed ones such as gutters, storm sewers, and drainage channels, can cause an increase in flow velocity in downstream waterways which, when combined with the increased flow duration, can create new or aggravate existing waterway erosion and scour.

Finally, the decrease in infiltration and resultant increase in runoff indicates that less rainfall may be entering the local or regional groundwater, resulting in the depletion or complete eradication of waterway baseflows and the lowering of the groundwater table. While research into these impacts has at times produced somewhat conflicting results (Center for Watershed Protection, 2003), the negative impacts to baseflows

Table 2-2: Land Development Impacts Example, Pre- and Post-Development Site Conditions

Development Condition	Site Land Cover	Average Initial Abstraction
Pre-developed	Woods	1.6
Post-developed	25% impervious and 75% turf grass	0.9

Table 2-3: Land Development Impacts Example, Pre- and Post-Development Runoff Volumes

Storm Frequency	24-Hour Rainfall (Inches)	Estimated Runoff Depth	
		Pre-Developed	Post-Developed
2-year storm	2.8	0.1	0.6
10-year storm	4.0	0.5	1.3

and groundwater levels caused by land use changes have become a generally accepted tenet of urban runoff management programs.

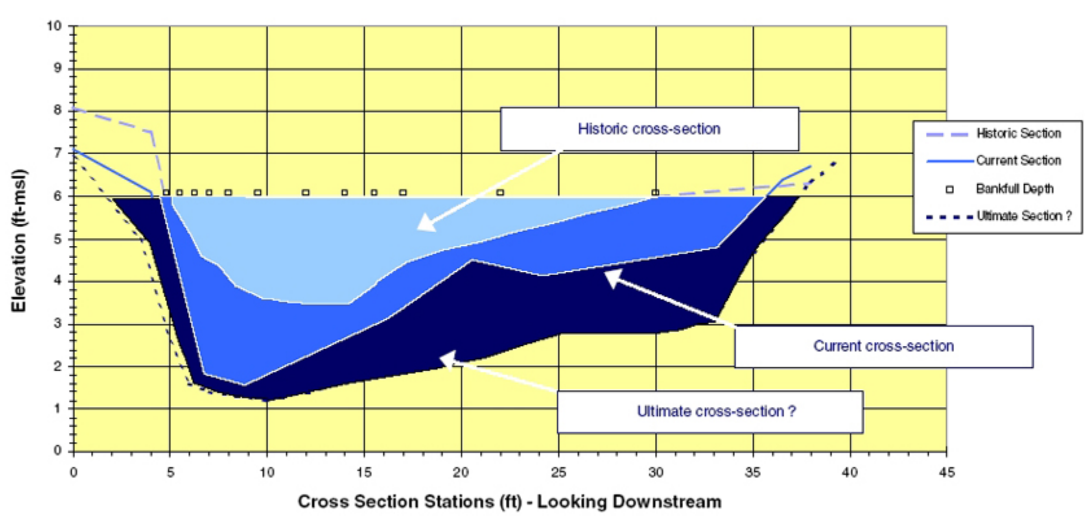
Such impacts can be quantified through a hypothetical land development example utilizing the NRCS Runoff Equation. The pre- and post-developed land uses and covers are summarized in Table 2-2. As shown in the table, the wooded land cover that exists in the pre-developed condition will be changed to a combination of 75 percent turf grass and 25 percent impervious cover that is directly connected to the site's drainage system. Our example will assume a relatively granular site soil, identified as a Hydrologic Soil Group B soil in the NRCS method, and will analyze the impacts of the site development for both a 2- and 10-year, 24-hour rainfall. The resultant pre- and post-developed runoff volumes for both storm events are summarized in Table 2-3.

A review of Table 2-2 indicates that the average initial abstraction for the post-developed site will be approximately 40 percent smaller than for the pre-developed one, decreasing by 0.7 inches from 1.6 to 0.9 inches. This means that while a minimum of 1.6 inches of rainfall is required to produce runoff from the pre-developed site, only 0.9 inches on average will be necessary under post-developed conditions. It should be noted that this post-developed initial abstraction is an average value for the combined turf grass and impervious cover site and that only approximately 0.1 inches of rain should be necessary to produce runoff from the impervious portions. This means that runoff volumes to downstream waterways are not only expected to increase but that this runoff will now be occurring from rain events between approximately 0.1 and 1.6 inches that previously produced no site runoff or waterway flow. This will significantly increase the number of times when runoff and associated pollutants will be flowing to and through downstream waterways.

A review of Table 2-3 indicates the extent of the estimated runoff volume increases that can be expected due to the proposed land use change. As shown in the table, the total 2-year runoff volume from the site is estimated to increase by 500 percent following development from approximately 0.1 to 0.6 inches. The estimated 10-year volume increase, while smaller, is nevertheless significant, increasing from approximately 0.5 to 1.3 inches or by approximately 160 percent. This also indicates that the quantity impacts of land use change are more acute for smaller, more frequent rainfalls – a distinct problem for waterways that are particularly sensitive to such storm events.

The potential impacts of this increased frequency and volume of development site runoff to downstream waterways is illustrated in Figure 2-11, which depicts the changes to a stream cross section in Maryland between 1950 and 2000 (Center for Watershed Protection, 2003). As shown in the figure, both the width and depth of the cross section have increased considerably between

Figure 2-11: Effects of Urbanization on Maryland Stream Cross Section



Source: Center for Watershed Protection, 2003 the 1950 or "Historic" configuration and the 2000 or "Current" condition. It should be noted that, over this time period, sufficient land development has occurred in the stream's drainage area to increase the total impervious land cover from approximately 2 percent to 27 percent. The "Ultimate" cross section shown in the figure is an estimate of the final cross section size in response to this degree of urbanization. Additional research indicates that stream channel areas can enlarge by two to eight times due to drainage area urbanization (Center for Watershed Protection, 2003).

In addition to channel cross section enlargement, other physical impacts of increased runoff volumes, rates, frequencies, and durations include (Center for Watershed Protection, 2003):

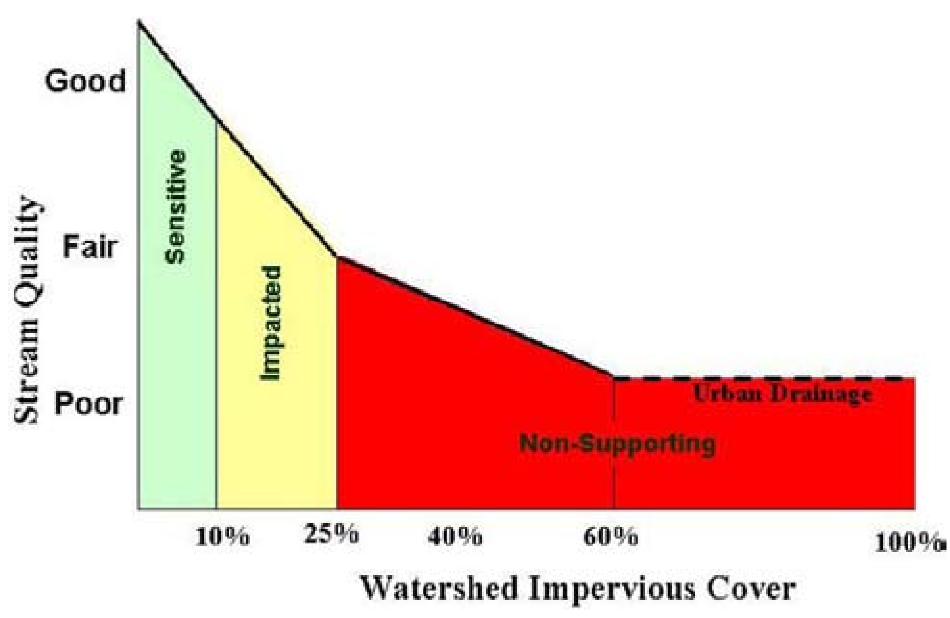
- Channel bank undercutting;
- Channel bottom incision;
- Loss of aquatic habitat;
- Increase in sediment yield and transport;
- Loss of riparian cover; and
- Increase in water temperature.

Utilizing the results from a number of research studies, the Center for Watershed Protection has developed a relatively simple model that demonstrates a direct relationship between drainage area urbanization (as measured by the percentage of impervious land cover in the drainage area) and the general quality of the stream to which the area's runoff drains. This model is depicted in Figure 2-12. It indicates that as total impervious cover in a drainage area increases, the quality of the stream decreases. This model has been widely adopted as a predictor of the adverse effects that can occur if drainage area development continues in an unmanaged or unregulated way.

In summary, the above section presented the following ideas and information:

- Land use changes can increase impervious land cover, decrease soil permeability and vegetated cover, reduce initial abstractions, and shorten runoff response times.
- Such changes can result in increased volumes, rates, durations, and frequencies of surface runoff and waterway flows.

Figure 2-12: Center for Watershed Protection's Impervious Cover Model



Source: Center for Watershed Protection, 2003

- Such increases can adversely impact waterways through channel enlargement, bank undercutting, aquatic habitat destruction, increased sediment loadings, and increased water temperatures.
- Such impacts have been extensively documented through research.

Summary and Conclusions

This chapter demonstrates how an understanding of the fundamentals of the rainfall-runoff process is critical to the development and operation of an effective urban runoff management program. Such fundamentals include:

1. The rainfall-runoff process is complex, and no perfect runoff estimation methods exist.
2. However, through informed assumptions and an understanding of the fundamentals, we can generally overcome these complexities and produce reasonable, reliable, and safe runoff estimates.
3. Several types of runoff estimation methods are available, utilizing a range of parameters and data including both actual long-term rainfall data and single event design storms.
4. The type and accuracy of the required runoff estimate and the availability of the required data will largely determine the runoff estimation method to be used.
5. The impacts of land use change include increased runoff and waterway flow volumes, rates, durations, and frequencies.
6. These increases can cause significant physical damage to waterways and aquatic habitats as well as biological, chemical, and environmental damage to ground and surface waters. Further information on these quality impacts are presented in Chapters 3 and 4.
7. Management of land use changes and preservation of the rainfall-runoff process for undeveloped conditions can prevent or mitigate such damage.
8. Structural stormwater management measures can also be used to reduce or control the runoff impacts of land use changes both during and after site construction. These measures are described in detail in Chapters 8 and 9.

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CHAPTER 3

Water Quality Impacts of Urbanization

CHAPTER 3

Water Quality Impacts of Urbanization

This chapter focuses on the physio-chemical aspects of water quality by examining the characteristics, sources, and patterns of urban runoff pollutants. Stormwater runoff from urbanized areas carries with it a wide variety of pollutants from diverse and diffuse sources. Based on data collected over many decades, throughout the country, it is apparent that there is a great deal of variability in urban runoff pollutant composition and concentrations. Representing all recognized classes of water pollutants, these runoff contaminants originate not only from land-use activities in the drainage area where runoff is collected but also occur as atmospheric deposition from areas outside the watershed of the receiving water body. In addition, exchanges between surface and groundwater can also be a pathway for pollutants. For example, landfill leachate or buried toxic waste can easily contaminate groundwater, which can then become a source of pollutants to surface waters. On the other hand, pollution can be transported via urban surface runoff and can result in the contamination of groundwater or surface receiving water bodies. The multiple sources of urban runoff pollution on, above, and below the surface represent a complex set of watershed conditions. They determine the effects that drainage from the watershed will have on natural receiving water, and represent a challenge for management.

The impact of stormwater runoff pollutants on receiving water quality depends on a number of factors, including pollutant concentrations, the mixture of pollutants present in the runoff, and the total load of pollutants delivered to the water body. Water pollutants often go through various physio-chemical processes before they can impact an aquatic biota. During their transport by surface waters and stormwater runoff, losses such as sedimentation can reduce the total stress burden on aquatic organisms, although the reduction may not be permanent (e.g., sediments can be resuspended). Physical, chemical, and biological processes can also cause transformations to different physical (particulate versus dissolved) or chemical (organic or inorganic) forms. Depending on the environmental conditions and the organisms involved, transformations can cause enhanced (synergistic) or reduced stress potential.

Water pollution is not the only condition in the watershed that causes ecological stress. Chief among other stresses is modified hydrology from increased stormwater runoff flow volumes and peak rates discharged from urbanized landscapes. Conversely, stress can come from decreased dry weather baseflows resulting from reduced groundwater recharge in urban areas. Finally, aquatic biota can be affected by the various stresses in whatever form they arrive. Biota may have an easier time dealing with a few rather than many stressors, especially when they act in a synergistic manner. Of course, populations of aquatic organisms do not live in isolation but interact with other species, especially in predator-prey relationships. These interactions have many implications for the ecosystem. For example, the loss of one species from a pollution problem will likely result in the decline or elimination of a major predator of that species. These and other physical or biological stressors will be discussed in detail in the next chapter.

Background

Stormwater Pollutant Sources

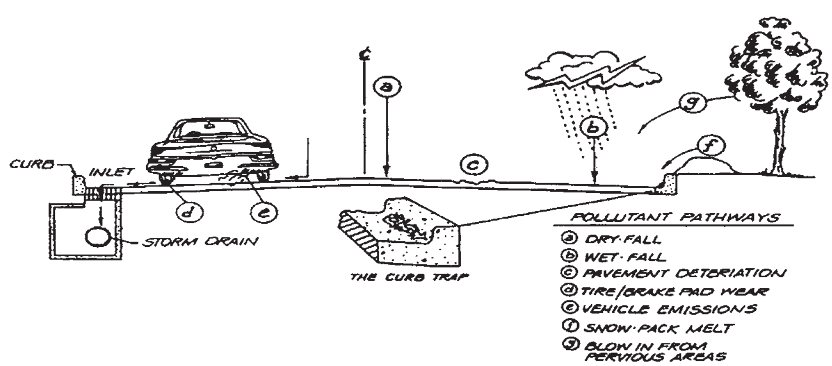
Stormwater runoff from urbanized areas is generated from a number of sources, including residential areas, commercial and industrial areas, roads, highways, and bridges. Essentially, any surface that does not have the capability to store and infiltrate water will produce runoff during storm events. These are known as impervious surfaces. As the level of imperviousness increases in a watershed, more rainfall is converted to runoff. In addition to creating greater runoff volumes, impervious surfaces (roads, parking lots, rooftops, etc.) are the primary source areas for pollutants to collect within the built environment (Figure 3-1). Runoff from storm events then carries these pollutants into receiving waters via the stormwater conveyance network. Land-use (e.g., residential, commercial, and industrial) and human activities (e.g., industrial operations, residential lawn care, and vehicle maintenance) characteristic of a drainage basin largely determine the mixture and level of pollutants found in stormwater runoff (Weibel et al., 1964; Griffin et al., 1980; Novotny and Chester, 1981; Bannerman et al., 1993; Makepeace et al., 1995; Pitt et al., 1995).

Atmospheric deposition of pollutants is typically divided into wet-fall and dry-fall components. These inputs can come from local sources, such as automobile exhaust, or from distant sources such as coal or oil power plant emissions. Regional industrial and agricultural activities can also contribute to atmospheric deposition as dry-fall. Precipitation also carries pollutants from the atmosphere to earth as wet-fall. Depending on the season and location, atmospheric deposition can be a significant source of pollutants in the urban environment. The USGS has estimated that up to 25 percent of the nitrogen entering the Chesapeake Bay likely comes from atmospheric deposition (USGS, 1999).

The types of land-use activities present in a drainage basin are also important in determining stormwater quality. The method of conveyance within the built environment is influential as well. The traditional means of managing stormwater runoff in urban areas has been to construct a network of curb-and-gutter streets, drain-inlet catch basins, and storm drain piping to collect this runoff, transport it quickly and efficiently away from the urbanized area, and discharge the stormwater into receiving waters.

Separate storm sewer systems convey only stormwater runoff. Water conveyed in separate storm sewers is frequently discharged directly to receiving waters without treatment. Stormwater can also bypass the stormwater infrastructure and flow into receiving waters as diffuse runoff from parking lots, roads, and landscaped areas. In cases where a separate storm sewer

Figure 3-1: Stormwater Runoff Pathways and Pollutant Sources



Streets provide several pathways for stormwater pollutants. Atmospheric pollutants settle or are washed onto the street during rain events (a, b). Pavement fragments also contribute to stormwater pollution (c). Vehicles contribute emissions and tire and brake pad particles (d, e). Snow collected at the street edge melts and contributes salts (f). Leaves and pollen from trees are blown into the street (g). Curb and gutter systems channel polluted stormwater directly into streams.

Source: Schueler, 1995

system is present, sanitary sewer flows are conveyed to the municipal wastewater treatment plant (WWTP) in a separate sanitary sewer system.

In a combined sewer system, stormwater runoff may be combined with sanitary sewer flows for conveyance. During low flow periods, flows from combined sewers are treated by the WWTP prior to discharge to receiving waters. During large rainfall events, however, the volume of water conveyed in combined sewers can exceed the storage and treatment capacity of the wastewater treatment system. As a result, discharges of untreated stormwater and sanitary wastewater directly to receiving streams can occur in these systems. These types of discharges are known as combined sewer overflow (CSO) events.

Urban streets are typically significant source areas for most contaminants in all land-use categories. Parking lots and roads are generally the most critical source areas in industrial and commercial areas. Along with roads, lawns, landscaped areas, and driveways can be significant sources of pollution in residential areas. In addition, roofs can contribute significant quantities of pollutants in all land-use types (Bannerman et al., 1993). The quantity of these pollutants delivered to receiving waters tends to increase with the degree of development in urban areas.

Historically, as urbanization occurred and storm drainage infrastructure systems were developed in this country, the primary concern was to limit nuisance and potentially damaging flooding due to the large volumes of stormwater runoff that were generated. Little, if any, thought was given to the environmental impacts of such practices on water quality. Due to the diffuse nature of many stormwater discharges, it is difficult to quantify the range of pollutant loadings to receiving waters that are attributable to stormwater discharges. Awareness of the damaging effects stormwater runoff is causing to the water quality and aquatic life of receiving waters is a relatively recent development, as is stormwater quality treatment.

Stormwater Runoff Pollutants

Stormwater runoff from urban areas can contain significant concentrations of harmful pollutants that can contribute to adverse water quality impacts in receiving streams. Impacts on beneficial uses can include such things as beach closures, shellfish bed closures, limits on fishing, and limits on recreational contact in waters that receive stormwater discharges. Contaminants enter stormwater from a variety of sources in the urban landscape. In general, these pollutants degrade water quality in receiving waters associated with urbanizing watersheds. Stormwater pollution is often a contributing factor where there is an impairment of beneficial use and/or an exceedance of criteria included in water-quality standards (WQS).

Research has identified stormwater runoff as a major contributor to water quality degradation in urbanizing watersheds (Field and Pitt, 1990; Makepeace et al., 1995; Pitt et al., 1995; Herricks, 1995; CWP, 2003). Stormwater or urban runoff typically contains a mixture of pollutants, including the following major constituents:

- Sediment;
- Nutrients (nitrogen and phosphorus);
- Chlorides ;
- Trace metals ;
- Petroleum hydrocarbons ;
- Microbial pollution ; and
- Organic chemicals (pesticides, herbicides, and industrial).

Sediment is one of the most common and potentially damaging pollutants found in urban runoff. Sediment pollutant levels can be measured as Total Suspended Solids (TSS) and/or Turbidity. TSS is a measure of the total mass of suspended sediment particles in a sample of water. The combination of flow and TSS gives a measure of sediment load carried downstream. Turbidity measures the scattering of light by suspended sediment particles in a water sample. Turbidity and TSS in stormwater runoff can vary significantly from region to region, as well as within a local area, depending on the sources of sediment contributing to the runoff load. The size distribution of suspended particles, as well as the composition of particulate (e.g. organic vs. inorganic) can have a significant influence on the measured turbidity or TSS of a water sample. Current research indicates that particle size distribution (PSD) may be an important parameter to measure when evaluating the sediment component in surface waters or stormwater runoff (Bent et al. 2001; US-EPA 2001; Burton and Pitt 2002).

Sediment in stormwater runoff can come from the wash-off of particulate material from impervious surfaces in already urbanized areas and/or from active construction sites in urbanizing areas. Streets, parking lots, lawns, and landscaped areas have been identified as the primary source areas for sediment in the urban environment (CWP, 2003). Construction site runoff has the potential to contain very high levels of sediment, especially if proper erosion and sediment control (ESC) best management practices (BMP) are not employed. The TSS concentration from uncontrolled construction sites can be more than 150 times greater than that found in natural, undeveloped landscapes (Leopold, 1968). Uncontrolled runoff from construction sites has been shown to have a TSS ranging from 3,000 to 7,000 mg/l (CWP, 2003). When proper ESC BMP techniques are utilized, the TSS level can typically be reduced by at least an order of magnitude, if not more (CWP, 2003).

Nutrients (nitrogen and phosphorous) are essential elements in all aquatic ecosystems. However, when these nutrients are found at excessive levels, they can have a negative impact on aquatic systems. Common sources of nutrients such as nitrates and phosphates include chemical fertilizers applied to lawns, golf courses, landscaped areas, and gardens. Residential lawns and turf areas (e.g., sports fields, golf courses, and parks) in urbanizing watersheds have been shown to be "hot spots" for nutrient input into urban runoff (CWP, 2003). In general, lawns and turf areas contribute greater quantities of nutrients than other urban source areas. In fact, research suggests that nutrient concentrations in runoff from lawns and turf areas can be as much as four times greater than those from other urban nutrient source areas (Bannerman et al., 1993; Steuer et al., 1997; Waschbusch et al., 2000; Garn, 2002).

Sources of nutrients such as nitrates and phosphates include chemical fertilizers applied to lawns, golf courses, landscaped areas, and gardens. In addition, nutrient pollution can originate from failing septic systems or from inadequate treatment of wastewater discharges from an urban WWTP. Atmospheric deposition of nutrient compounds from industrial facilities or power generation plants is also a source of nutrients in the built environment. Soil erosion and other sediment sources can also be significant nutrient sources, as nutrients often tend to be found in particulate form. Research indicates that human land-use activity can be a significant source of nutrient pollution to stream and wetland ecosystems (Bolstad and Swank, 1997; Sonoda et al., 2001; Brett et al., 2005). Many studies have linked nutrient levels in runoff to contributing drainage area land uses, with agricultural and urban areas producing the highest concentrations (Chessman et al., 1992; Wernick et al., 1998; USGS, 1999). Snowmelt runoff in urban areas can also contain elevated levels of nutrients (Oberts, 1994).

Excessive nutrient levels in urban runoff can stimulate algal growth in receiving waters and cause nuisance algal blooms when stimulated by sunlight and high temperatures. The decomposition that follows these algal blooms, along with any organic matter (OM) carried by runoff, can lead to depletion of dissolved oxygen (DO) levels in the receiving water and bottom sediments. This can result in a degradation of habitat conditions (low DO), offensive odors, loss of contact recreation usage, or even fish kills in extremely low DO situations.

Nitrate is the form of nitrogen found in urban runoff that is of most concern. The nitrate anion (NO_3) is not usually adsorbed by soil and therefore moves with infiltrating water. Nitrates are present in fertilizers, human wastewater, and animal wastes. Nitrate contamination of groundwater can be a serious problem, resulting in contamination of drinking water supplies (CWP, 2003). High nitrate levels in drinking water can cause human health problems.

Phosphates (PO_4) are the key form of phosphorus found in stormwater runoff. Phosphates in runoff exist as soluble reactive phosphorus (SRP) or ortho-phosphates, poly-phosphates, and as organically bound phosphate. The poly-form of phosphates is the one that is found in some detergents. Orthophosphates are found in sewage and in natural sources. Organically bound phosphates are also found in nature, but can also result from the breakdown of phosphorus-based organic pesticides. Very high concentrations of phosphates can be toxic.

Chlorides are salt compounds found in runoff that result primarily from road de-icer applications during winter months. Sodium chloride (NaCl) is the most common example. Although chlorides in urban runoff come primarily from road deicing materials, they can also be found in agricultural runoff and wastewater. Small amounts of chlorides are essential for life, but high chloride levels can cause human illness and can be toxic to plants or animals.

Metals are among the most common stormwater pollutant components. These pollutants are also referred to as trace metals (e.g., zinc, copper, lead, chromium, etc.). Many trace metals can often be found at potentially harmful concentrations in urban stormwater runoff (CWP, 2003). Metals are typically associated with industrial activities, landfill leachate, vehicle maintenance, roads, and parking areas (Wilber and Hunter, 1977; Davies, 1986; Field and Pitt, 1990; Pitt et al., 1995). In one study in the Atlanta (GA) metropolitan area, zinc (Zn) was found to be the most significant metal found in urban street runoff (Rose et al., 2001). Similar results were found in the Puget Sound (WA) region (May et al., 1997). A study in Michigan found that parking lots, driveways, and residential streets were the primary source areas for zinc, copper, and cadmium pollution found in urban runoff (Steuer et al., 1997).

Most of the metal contamination found in urban runoff is associated with fine particulate (mostly organic matter), such as is found deposited on rooftops, roads, parking lots, and other depositional areas within the urban environment (Furguson and Ryan, 1984; Good, 1993; Pitt et al., 1995; Stone and Marsalek, 1996; Crunkilton et al., 1996; Sutherland and Tolosa, 2000). However, a significant fraction of copper (Cu), cadmium (Cd), and zinc (Zn) can be found in urban runoff in the dissolved form (Pitt et al., 1995; Crunkilton et al., 1996; Sansalone and Buchberger, 1997).

Petroleum hydrocarbon compounds are another common component of urban runoff pollution. Hydrocarbon sources include vehicle fuels and lubricants (MacKenzie and Hunter, 1979; Whipple and Hunter, 1979; Hoffman et al., 1982; Fram et al., 1987; Kucklick et al., 1997; Smith et al., 1997). Hydrocarbons are normally attached to sediment particles or organic matter carried in urban runoff. The increase in vehicular traffic associated with urbanization is frequently linked to air pollution, but there is also a negative relationship between the level of automobile use in a watershed and the quality of water and aquatic sediments. This has been shown for polycyclic aromatic hydrocarbon (PAH) compounds (Van Hoffman et al., 1982; Metre et al., 2000; Stein et al., 2006). In most urban stormwater runoff, hydrocarbon concentrations are generally less than 5 mg/l, but concentrations can increase to 10 mg/l in urban areas that include highways, commercial zones, or industrial areas (CWP, 2003). Hydrocarbon "hot spots" in the urban environment include gas stations, high-use parking lots, and high-traffic streets (Stein et al., 2006). A Michigan study showed that commercial parking lots contributed over 60 percent of the total hydrocarbon load in an urban watershed (Steuer et al.,

Table 3-1: Pollutants Commonly Found in Stormwater and Their Forms

Pollutant Category	Specific Measures
Solids	Settleable solids Total suspended solids (TSS) Turbidity (NTU)
Oxygen-demanding material	Biochemical oxygen demand (BOD) Chemical oxygen demand (COD) Organic matter (OM) Total organic carbon (TOC)
Phosphorus (P)	Total phosphorus (TP) Soluble reactive phosphorus (SRP) Biologically available phosphorus (BAP)
Nitrogen (N)	Total nitrogen (TN) Total kjeldahl nitrogen (TKN) Nitrate + nitrite-nitrogen (NO3+NO2-N) Ammonia-nitrogen (NH3-N)
Metals	Copper (Cu), lead (Pb), zinc (Zn), cadmium (Cd), arsenic (As), nickel (Ni), chromium (Cr), mercury (Hg), selenium (Se), silver (Ag)
Pathogens	Fecal coliform bacteria (FC) Enterococcus bacteria (EC) Total coliform bacteria (TC) Viruses
Petroleum hydrocarbons	Oil and grease (OG) Total petroleum hydrocarbons (TPH)
Synthetic organics	Polynuclear aromatic hydrocarbons (PAH) Pesticides and herbicides Polychlorobiphenols (PCB)

1997). Lopes and Dionne (1998) found that highways were the largest contributor of hydrocarbon runoff pollution.

Microbial pollution includes bacteria, protozoa, and viruses that are common in the natural environment, as well as those that come from human sources (Field and Pitt, 1990; Young and Thackston, 1999; Mallin et al., 2000). Many microbes are naturally occurring and beneficial, but others can cause diseases in aquatic biota and illness or even death in humans. Some types of microbes can be pathogenic, while others may indicate a potential risk of water contamination, which can limit swimming, boating, shellfish harvest, or fish consumption in receiving waters. Microbial pollution is almost always found in stormwater runoff, often at very high levels, but concentrations are typically highly variable (Pitt et al., 2004). Sources of bacterial pollution in the urban environment include failing septic systems, WWTP discharges, CSO events, livestock manure runoff, and pet waste, as well as natural sources such as wildlife. Young and Thackston (1999) showed that bacterial concentrations in stormwater runoff were

Table 3-2: Pollutants Commonly Found in Stormwater and Their Sources

Pollutant	Potential Sources
Hydrocarbons (gasoline, oil, and grease)	Internal combustion engines Automobiles Industrial machinery
Copper (Cu)	Building materials Paints and wood preservatives Algicides Brake pads
Zinc (Zn)	Galvanized metals Paints and wood preservatives Roofing and gutters Tires
Lead (Pb)	Gasoline Paint Batteries
Chromium (Cr)	Electro-plating Paints and preservatives
Cadmium (Cd)	Electro-plating Paints and preservatives
Pesticides	Agriculture and grazing Residential and commercial use
Herbicides	Agriculture and grazing Residential and commercial use Roadside vegetation maintenance
Organic compounds	Industrial processes Power generation
Bacteria and pathogens	Human sewage Livestock manure Domestic animal fecal material
BOD	Agriculture and grazing Human sewage
Nutrients (N and P)	Agriculture and grazing Lawn and landscape fertilizer
Fine sediment	Agriculture and grazing Timber harvest Pavement wear Construction sites Road sanding

directly related to the level of watershed and impervious surface area. Mallin and others (2000) also found that bacterial pollution problems were much more common in urbanized coastal watersheds than in undeveloped catchments. There is also evidence that microbial populations can survive and possibly even grow in urban stream sediments and in sediments found in storm sewer systems, making the stormwater infrastructure a potential source of microbial pollution (Bannerman et al., 1993; Steuer et al., 1997; Schueler, 1999).

Pesticides, herbicides, and other organic pollutants are often found in stormwater flowing from residential and agricultural areas throughout the U.S. (Ferrari et al., 1997; USGS, 1999; Black et al., 2000; Hoffman et al., 2000). Among the many pesticides and herbicides commonly found in urban runoff and urban streams are the following:

- Diazinon;
- Chlorpyrifos;
- Chlordane
- Carbaryl;
- Atrazine;
- Malathion;
- Dicamba;
- Prometon;
- Simazine; and
- 2,4-D.

Toxic industrial compounds such as PCBs can also be present in urban runoff (Black et al., 2000). Studies in Puget Sound confirm these findings (Hall and Anderson, 1988; May et al., 1997; USGS, 1997; Black et al., 2000). In many cases, even banned pesticides such as DDT or other organo-chlorine based pesticides (e.g., chlordane and dieldrin) can be found in urban stream sediments. The EPA estimates that nearly 70 million pounds of pesticides and herbicides are applied to lawns and other surfaces within the urban environment of the U.S. each year (CWP, 2003). These pesticides or herbicides vary in mobility, persistence, and potential aquatic impact. Many pesticides and herbicides are known or suspected carcinogens and can be toxic to humans and aquatic biota. However, most of the known health effects require exposure to higher concentrations than are typically found in the urban environment. However, the health effects of chronic exposure to low levels of pesticides and herbicides are generally unknown (Ferrari et al., 1997).

In urban runoff, most pollutants are associated with fine sediment or other natural particulates (e.g., organic matter). This condition differs between the specific pollutants. For example, depending on overall chemical conditions, each metal differs in solubility. For in-stance, lead (Pb) is relatively insoluble, while zinc (Zn) is relatively soluble. The nutrients phosphorus (P) and nitrogen (N) typically differ substantially from one sample to another in dissolved and particulate forms.

In addition to pollutants, other water quality characteristics affect the behavior and fate of contaminants in receiving water. These characteristics include:

- Temperature – critical to the survival of cold-water organisms. Temperature also affects solubility and ion mobility;
- PH – an expression of the relative hydrogen ion concentration on a logarithmic scale of 0-14, with a pH < 7.0 being acidic, a pH of 7.0 being neutral, and a pH > 7.0 representing basic conditions;
- Dissolved oxygen (DO) – a measure of molecular oxygen dissolved in water, critical to the survival of aerobic aquatic biota. In addition, DO levels can affect the release of chemically bound constituents from sediments;
- Alkalinity or acid neutralizing capacity (ANC) – the capacity of a solution to neutralize acid of a standard pH, usually the result of its carbonate and bicarbonate ion content, but conventionally expressed in terms of calcium carbonate equivalents;
- Hardness – an expression of the relative concentration of divalent cations, principally calcium (Ca) and magnesium (Mg), also conventionally expressed in terms of calcium carbonate equivalents; and
- Conductivity – a measure of the ability to conduct an electrical current as a result of its total content of dissolved substances.

These physio-chemical characteristics can affect pollutant behavior in several ways. For example, metals generally become more soluble as pH drops below neutral and hence become more bioavailable to organisms (Davies, 1986). Alternatively, the chemical elements creating hardness work against the toxicity of many heavy metals. Low DO levels can also make some metals more soluble. Anaerobic conditions in lake bottoms often lead to the release of phosphorus from sediments, as iron changes from the ferric to the ferrous form (Welch, 1992). As discussed earlier, most of the pollutant composition of urban stormwater runoff stems from particulate material or fine sediment from surface soil erosion (e.g., construction site erosion) and from wash-off of solids accumulated on impervious surfaces throughout the urban environment (e.g., streets, highways, parking lots, and rooftops).

Pollutant Fate and Transport

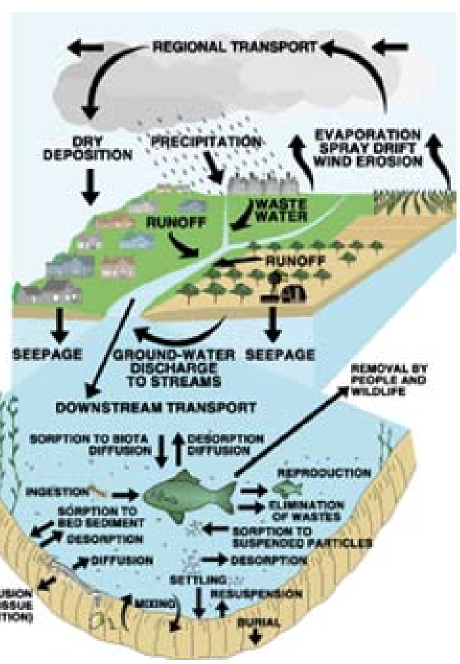
In general, the primary transport mechanism for most urban pollutants is stormwater runoff. The physio-chemical effects of watershed urbanization tend to be more variable than the hydro-geomorphic or physical habitat impacts discussed previously. As indicated above, stormwater can contain a variety of pollutants and the pollutants typically found in stormwater come from a variety of sources (see Tables 3-1 and 3-2). These pollutants most often occur as mixtures of physio-chemical constituents, which depend on the land uses found in the contributing drainage basin as well as the type and intensity of human activities present. In general, the more intense the level of urbanization, the higher the pollutant loading, and the greater the diversity of land-use activities, the more diverse the mixture of pollutants found in stormwater runoff.

The transport and fate mechanisms of stormwater pollutants in receiving waters tend to be highly variable and site-specific. Pollutants are often transported from source areas (roads, parking lots, lawns, etc.) to receiving waters via roadside ditches, stormwater pipes, or by atmospheric deposition (Figure 3-2). In general, the concentration of pollutants found in stormwater runoff is much higher than that found in receiving waters, due mostly to dilution and removal mechanisms. There is evidence of a "first flush" effect for some constituents such as metals and hydrocarbons, especially in highly impervious and connected drainage areas (Pitt et al., 1995; Sansalone and Buchberger, 1997; Pitt et al., 2004).

As was discussed earlier, most stormwater pollutants are typically found in particulate form, attached to fine sediment particles and organic matter (Pitt et al., 1995). This is especially true for nutrients, organics, and metals. In most cases, the particulate forms of toxic pollutants, such as metals tend to be less "bio-available" (Herricks, 1995).

Sedimentation is the most common pollutant fate or removal mechanism because many pollutants tend to be associated with fine particulate material and/or organic matter (Pitt et al., 1995). However, pollutants can also be transformed from particulate form to dissolved form due to changes in water chemistry (pH, hardness, DO, etc.) at the sediment-water interface. Microbial activity can also transform toxic compounds, such as heavy metals, in sediments from inorganic forms to more toxic organic forms, which also tend to be more soluble (Herricks, 1995). In addition, scouring of sediments during stormflow events and associated changes in water chemistry during these sporadic events can mobilize polluted sediments and release toxic substances into the water column where biological uptake can occur. Large quantities of sediments can be transported by stormflows in urbanizing creeks, resulting in resuspension and redeposition of pollutants. Because of the potential for accumulation of pollutants in sediment and the potential of sediments as sources of toxics, polluted sediments likely play an important role in many of the biological impacts associated with stormwater runoff. In general, most pollutants, especially metals, are found in particulate forms within the water column, or sediments and pollutant concentrations tend to be higher for smaller sediment particle sizes (Novotny and Chester, 1989; Ferguson and Ryan, 1984; Herricks, 1995; Makepeace et al., 1995; Pitt et al., 1995).

Figure 3-2: Pollutant Movement in the Hydrologic Cycle



Source: USGS, 1999

Table 3-3 summarizes urban runoff pollutant sources and shows that most pollutant categories have diverse sources. Likewise, the major sources emit contaminants in most pollutant categories. The atmosphere also contributes some pollution to runoff. Thus, urban runoff is a multifaceted and complex problem to manage.

Quantifying Urban Runoff Pollutants

Urban Runoff Measurement

The concentrations of water-quality constituents tend to be highly variable, depending on a number of environmental factors. These factors may include:

- Drainage basin area or potential runoff volume;
- Drainage system characteristics (e.g., piping, ditches, etc.);
- Drainage basin land use and land cover (LULC);
- Rainfall volume, intensity, and antecedent dry period;
- The presence of pollutant source areas or "hot spots"; and
- Pollutant deposition or build-up rates.

Water pollutants are typically quantified by concentrations and loadings. Concentration is the mass of pollutant per unit volume of water sample, usually expressed as mg/l or ug/l. It is a measure of the pollutant content at the instant the sample is taken. If the pollutant level is higher than an aquatic organism can tolerate, the concentration represents an acute effect that could be lethal or affect the performance of some physiological function as long as the concentration persists. The effects of pollutant concentrations have been established through bioassays exposing test organisms in standard laboratory procedures. However, these simple, static tests completely omit the dynamic patterns and other complexities associated with urban runoff. Toxicity of pollutants will be discussed in more detail in Chapter 4.

Loading is the mass of pollutants delivered to a water body over a period of time and is usually given on an annual basis as kg/yr or lbs/yr. When ascribed to a particular land use, loading is sometimes termed yield or simply export per unit area of the land use (kg/ha-y or lbs/acre-y). It represents the cumulative burden over the extended period and hence the potential chronic effects on receptor organisms. With few exceptions (e.g., phosphorus loading to lakes), testing has not established the biological significance of loadings and the way they are delivered to a water body. Thus, loading is mainly used to make comparisons, for example, of total pollutant burden before and after development or with and without a certain control strategy. Pollutant loadings are also the basis for regulation under the Total Maximum Daily Load (TMDL) program that is part of the CWA.

Table 3-3: Urban Runoff Pollutant Sources and Constituents

Pollutant source	Solids	Nutrients	Pathogens	Oxygen Demand	Metals	Oils	Organics
Soil erosion	x	x		x	x		
Fertilizers		x					
Human waste	x	x	x	x			
Animal waste	x	x	x	x			
Vehicle fluids	x		x	x	x	x	
Internal combustion						x	
Vehicle wear	x			x	x		
Household chemicals	x	x		x	x	x	x
Industrial processes	x	x		x	x	x	x
Paints and preservatives					x	x	
Pesticides				x	x	x	

A quantitative estimate of water quality is needed to assess impacts from development actions or to predict the benefits of a management plan. This estimation process is sometimes called water quality modeling, although the term modeling is sometimes restricted to computer-based approaches. Water quality assessments are often based on annual pollutant loading estimates, although short-term loadings or concentrations are sometimes used. Long-term loadings tend to diminish the large fluctuations to which short-term phenomena are subject. Therefore, we can generally estimate long-term loading with more assurance than concentrations. Water quality sampling methods and monitoring programs will be covered in Chapter 5.

Urban Runoff Patterns

Because of the difficulty in determining runoff pollutant concentrations during dynamic flow conditions, the expense of sampling, and the analysis required to produce even a partial picture, the accepted practice is to determine an event-mean concentration (EMC). The event-mean concentration (EMC) is the concentration of a particular constituent that is representative of a specific environmental condition, usually with respect to a specific storm event. The NURP study defined the EMC as the total mass of pollutant contained in a runoff event divided by the total volume of runoff or flow for the event. The EMC can also be found by analyzing a single sample composited from a series of samples taken at points throughout the runoff event and combined in proportion to the flow rate existing at the time of sampling. This is often termed a flow-proportional or flow-weighted composite sample (EPA, 1997). The flow or runoff pattern of an event is customarily pictured on a hydrograph, which is a graph of flow rate (water volume per unit time) versus time. The integrated area under the curve is the total event runoff volume; the product of volume and EMC is the pollutant loading for the event. The sum of loadings for all events in an interval (e.g., a year) represents the cumulative pollutant burden during that time. In addition to its expediency, basing impact assessment on the EMC is justified from a biological standpoint because the EMC best represents the cumulative toxicity that organisms are exposed to during a storm event.

Based on the inherent variability of stormwater pollutant composition, the concentrations of water quality constituents are often estimated based on probability (i.e., the ability to state the probability of exceeding any selected concentration) or using statistically valid estimations of actual concentrations. Estimating the probability of concentrations can theoretically be used to estimate maximum or any other level, but it is usually restricted to the EMC. As stated earlier, an EMC is the concentration of a particular constituent that is representative of a specific environmental condition. For example, the EMC of TSS in a stream during a storm event could be based on multiple flow-weighted composite storm samples. Generally, to estimate an EMC, a large data -set is required to establish the underlying probability distribution for the locale or, alternatively, an assumption of the distribution and a smaller local data set to fit the distribution.

Most water quality studies have demonstrated that urban runoff pollutant concentrations typically fit a "log-normal" probability distribution (i.e., their logarithms are normally distributed). This is the characteristic distribution of data in cases where the distribution range is much higher than the mean and most values are in the lower portion (Little et al., 1983).

While pollutant magnitudes in urban runoff typically follow characteristic patterns over short and long time spans, they vary greatly over space and time. The short term can be defined as a period of hours during one or a sequence of storm events. Measurements at discrete points through such a period often reveal a pattern of pollutant concentration that is higher during the beginning of the storm event and tapering off as the storm continues. The so-called "first flush" of runoff during the first minutes often contains a relatively high concentration of contaminants, which then drops substantially and fluctuates at a lower level for the remainder of the runoff event. Analysis of climatological data throughout the U.S. indicates that most of the total annual runoff is produced by numerous small storms and the initial runoff from large storms. Theoretical reasons and some empirical demonstrations indicate that the majority of pollutant loadings for some constituents are generated by these smaller flow volumes (Burton and Pitt, 2002).

The first flush sometimes does not appear, or is less pronounced, when rainfall is not intense or follows soon after an earlier storm that cleans the surfaces. In addition, recent studies have shown that the first flush effect is usually only observed in highly impervious drainage areas such as parking lots or roads (Pitt et al., 2004). It has also been demonstrated that the first flush phenomenon may only be applicable to certain pollutants, including metals, hydrocarbons, and fine sediment (Pitt et al., 2004). In some cases, a secondary spike can also appear if a sudden burst of intense rain drives material off surfaces not completely cleaned by the initial runoff. In summary, runoff concentrations can assume an almost infinite variety of patterns depending on rainfall intensity, antecedent dry period (ADP) length and conditions, pollutant deposition during the ADP, and surface characteristics in the drainage basin.

Urban Runoff Pollution Characteristics

Several studies have attempted to quantify the level of various constituents in urban runoff. As mentioned earlier, these levels tend to vary depending on the land-use and human activities found in the contributing drainage area. The earliest comprehensive study of the water quality characteristics of urban runoff was the EPA (1983) National Urban Runoff Program (NURP). Between 1978 and 1983, EPA examined stormwater quality from separate storm sewers in different land uses. The NURP project studied 81 outfalls in 28 communities throughout the U.S. and included the monitoring of approximately 2,300 storm events. The data was compiled for several land-use categories, although most of the information was obtained from residential lands. Table 3-4 summarizes the NURP findings. NURP also produced graphs for each pollutant to determine the EMC at each site and the EMC medians from all sites nationwide (EPA, 1983). These plots can help estimate concentration exceedance probabilities at other locations. Such estimates are best made with specific site data including rainfall patterns, land-use data, geological data, and other characteristics similar to those of the location of interest. Using a regional or nationwide database is less satisfactory. Local stormwater data may be available from National Pollutant Discharge Elimination System (NPDES) monitoring programs.

Since NURP, other important studies have been conducted that characterize stormwater. The USGS National Water Quality Assessment (NAWQA) Program examined runoff quality from more than 1,100 storms at nearly 100 monitoring sites in 20 metropolitan areas (USGS, 1999). Table 3-5 summarizes the general findings of the USGS studies with respect to surface water quality. These USGS studies investigated specific urban pollutants including nutrients, metals, pesticides, and herbicides. The NAWQA studies also identified a close relationship between land use and water quality in agricultural and urban areas.

As an example, the NAWQA program found that insecticides such as diazinon and malathion were commonly found in surface water and stormwater in urban areas (USGS, 1999). This research found that almost every urban stream sampled had concentrations of insecticides that exceeded at least one EPA guideline or water-quality standard. Most urban streams had concen-

Table 3-4: Pollutants Commonly Found in Stormwater and Their Sources

	NURP Mean EMC	NURP Mean EMC
Pollutant	Median Urban Site	90th Percentile Urban Site
TSS (mg/l)	141-234	424-671
BOD (mg/l)	10-13	17-21
COD (mg/l)	73-92	157-198
TP (mg/l)	0.37-0.47	0.78-0.99
SRP (mg/l)	0.13-0.17	0.23-0.30
TKN (mg/l)	1.68-2.12	3.69-4.67
NO2-N (mg/l)	0.76-0.96	1.96-2.47
Total Cu (ug/l)	38-48	104-132
Total Pb (ug/l)	161-204	391-495
Total Zn (ug/l)	179-226	559-707

Notes: EMC = Event Mean Concentration; TSS = Total Suspended Solids; BOD = Biological oxygen Demand; COD = Chemical oxygen Demand; TP = Total Phosphorus; SRP = Soluble Reactive Phosphorus; TKN = Total Kjeldahl Nitrogen

Source: NURP, 1983

trations that exceeded a water-quality guideline in 10 to 40 percent of samples taken throughout the year (USGS, 1999). Urban streams also had the highest frequencies of occurrence of DDT, chlordane, and dieldrin (all of these compounds have been banned from use in the U.S. for decades) in sediments and fish tissue (USGS, 1999). In the Puget Sound region, the mixture of pesticides found in urban streams was directly related to the type of land use found in the contributing upstream drainage area (Ebbert et al., 2000). The NAWQA studies also found that the highest levels of organochlorine compounds, including pesticides and PCBs, were found in aquatic sediment and biota in urban areas (USGS, 1999). The main source of these complex mixtures of insecticides found in urban streams was identified as business, household, or garden use in developed areas, with urban runoff being the primary transport mechanism into urban streams and other receiving waters. A study in the Puget Sound region that correlated retail sales of specific pesticides with levels of those same pesticides found in local streams confirms this finding (Bortleson and Ebbert, 2000).

The NAWQA research also found that concentrations of phosphorus exceeded the EPA target goal (TP<0.1 mg/l) for the control of nuisance algal growth in over 70 percent of the urban receiving waters tested (USGS, 1999). As mentioned above, excessive algal or aquatic plant growth due to nutrient enrichment can lead to low levels of DO (hypoxia), which can be harmful to aquatic biota. Urban runoff can contain high levels of nutrients in the form of fertilizers washed off lawns and landscaped areas. In most cases in the NAWQA studies, enrichment of receiving waters occurred in small watersheds dominated by agricultural, urban, or mixed land use (USGS, 1999). The NAWQA research also found that nitrate contamination of groundwater aquifers and drinking water supplies had the potential to be a human health risk in urbanizing areas with high nitrate concentrations in stormwater runoff.

The Federal Highway Administration (FHWA) also analyzed stormwater runoff from 31 highways in 11 states during the 1970s and 1980s (FHWA, 1995). Other regional databases also exist, mostly using local NPDES data. Other studies have confirmed the NURP findings and improved the level of knowledge with regard to stormwater pollution impacts (Field and Pitt, 1990; Bannerman et al., 1993; Makepeace et al., 1995; Pitt et al., 1995). Table 3-6 illustrates the range of pollutant levels for typical urban runoff from a number of studies.

Highway runoff is often viewed as a separate and distinct form of stormwater. Because vehicle traffic tends to be the predominant pollution source in the highway environment, runoff from roads tends to have a characteristic signature (Novotny, 2003). Several studies have been conducted to characterize highway runoff (Stotz, 1987; Driscoll et al., 1990; Barrett et al., 1998; Wu et al., 1998; Kayhanian and Borroum, 2000; Pitt et al., 2004). In general, runoff from urban highways with greater average daily traffic (ADT) volumes tends to have higher pollutant concentrations than runoff from less-traveled highways (lower ADT). Most research studies have not found any direct correlation between ADT alone and pollutant concentrations for the great majority of pollutants (Masoud et al., 2003). However, ADT is almost always one of the more influential factors in determining runoff pollutant composition and concentration. Other parameters determining the quality of highway runoff include those that control pollutant build-up and wash-off. In addition to ADT, these factors include drainage catchment area and land use, antecedent dry period between storm events, and rainfall intensity and volume. Table 3-6 shows data from highways in comparison to other urbanized areas.

In a study in Southern California (Tiefenthaler et al., 2001), samples of stormwater runoff from parking lots

Table 3-5: Relative Levels of Pollution in Streams Throughout the U.S.

WQ Parameter	Urban Areas	Agricultural Areas	Undeveloped Areas
Nitrogen	Medium	Medium-high	Low
Phosphorus	Medium-high	Medium-high	Low
Herbicides	Medium	Medium-high	Low
Pesticides	Medium-high	Low-medium	Very low
Metals	High	Medium	Very low
Toxic Organics	High	Medium	Very low
Source: USGS, 1999			

were analyzed for a number of metals including Fe, Zn, Cu and Pb as well as polycyclic aromatic hydrocarbons (PAH). These metals and PAH had the highest mean concentrations of any constituents analyzed. Zinc (Zn) was found in particularly high concentrations, which were 3 times higher after dry periods. These pollutants were found to accumulate regardless of how much the parking lot was used or maintained. In this study, all of the samples from parking lot runoff contained toxins, and all samples of parking lot runoff were toxic. (Tiefenthaler et al., 2001). In addition, the longer the antecedent dry period before a storm event, the higher

Table 3-6: Typical Levels of Metals Found in Stormwater Runoff (ug/L)

Metal	Stormwater Median (90th Percentile)[a]	Mean (sd)[b]	Median (COV) Urban Stormwater[c]	Range for Highway Runoff[d]	Range for Parking Lot Runoff[e]
Zinc (Zn)	160 (500)	215 (141)	112.0 (4.59)	56-929	51-960
Copper (Cu)	34 (93)	33 (19)	16.0 (2.24)	22-7033	8.9-78
Lead (Pb)	144 (350)	70 (48)	15.9 (1.89)	73-1780	10-59
Cadmium (Cd)	n/a	1.1 (0.7)	1.0 (4.42)	0-40	0.5-3.3
Chromium (Cr)	n/a	7.2 (2.8)	7.0 (1.47)	0-40	1.9-10
Arsenic (As)	n/a	5.9 (2.8)	3.3 (2.42)	0-58	n/a
Mercury (Hg)	n/a	n/a	0.2 (1.17)	0-0.322	n/a
Nickel (Ni)	n/a	10 (2.8)	9.0 (2.08)	0-53.3	2.1-18
Silver (Ag)	n/a	n/a	3.0 (4.63)	n/a	n/a

Notes: n/a = not available.
Sources: [a]NURP, 1983. [b]Schiff et al., 2001. [c]Pitt et al., 2002. [d]Barrett et al., 1998. [e]SCCRP, 2001.

Table 3-7: Pollutants Commonly Found in Stormwater and Their Sources – 1983 (NURP) and 1999 Databases

Pollutant	Data Source	Mean EMC	Median EMC
TSS (mg/l)	Pooled NURP	78 174	55 113
BOD (mg/l)	Pooled NURP	14 10	12 8
COD (mg/l)	Pooled NURP	53 66	45 55
TP (mg/l)	Pooled NURP	0.32 0.34	0.26 0.27
SRP (mg/l)	Pooled NURP	0.13 0.10	0.10 0.08
TKN (mg/l)	Pooled NURP	1.73 1.67	1.47 1.41
NO2-N and NO3-N (mg/l)	Pooled NURP	0.66 0.84	0.53 0.66
Total Cu (ug/l)	Pooled NURP	14 67	11 55
Total Pb (ug/l)	Pooled NURP	68 175	51 131
Total Zn (ug/l)	Pooled NURP	162 176	129 140

EMC = Event Mean Concentration
TSS = Total Suspended Solids
BOD = Biological oxygen Demand
COD = Chemical oxygen Demand
TP = Total Phosphorus
SRP = Soluble Reactive Phosphorus
TKN = Total Kjeldahl Nitrogen

Source: Smullen et al., 1999

the concentration of pollutants and the higher the toxicity found in runoff samples (Tiefenthaler et al., 2001). In an arid climate such as Southern California, pollutants tend to build up during extended dry periods and then be washed off during heavy rainfall events that are typical of the climate. In this study, a pronounced first flush of toxins was observed at the beginning of storm events (Tiefenthaler et al., 2001). More intense rains reduced pollutant concentrations, however. Regardless of the intensity of the storm event, most loose pollutants were washed from the parking lot surface in the first 15 minutes (Tiefenthaler et al., 2001). The first flush of TSS was the most evident at the relatively low rainfall intensity of 6 mm/hour (Tiefenthaler et al., 2001). The key factor influencing the first flush of TSS was found to be rainfall duration instead of intensity. TSS concentrations dropped during the course of the storm, however. During longer storms, greater rainfall intensity did not reduce zinc concentrations. Intensity only increased the concentration of pollutants in the first minute of the storm (Tiefenthaler et al., 2001). Results indicated that the most wash-off of pollutants from parking lots occurred during small storms. This especially includes Pb and Zn (Tiefenthaler et al., 2001).

In 1999, an analysis of stormwater data collected since the original NURP study was conducted to update the event-mean concentration (EMC) values for typical urban stormwater quality (Smullen et al., 1999). This data review found only a few major differences between the NURP data and the pooled data from three national databases (see Table 3-7). In general, the pooled data was very comparable with the NURP data, with a few notable exceptions. The study found that the level of TSS in runoff was significantly

Table 3-8: Summary of Event-Mean Concentration (EMC) Data for Stormwater Runoff in the U.S.

Pollutant	Data Source	Mean EMC	Median EMC	Number of Events Sampled
TSS (mg/l)	Smullen and Cave, 1998	78.4	54.5	3047
BOD (mg/l)	Smullen and Cave, 1998	14.1	11.5	1035
COD (mg/l)	Smullen and Cave, 1998	52.8	44.7	2639
TP (mg/l)	Smullen and Cave, 1998	0.32	0.26	3094
SRP (mg/l)	Smullen and Cave, 1998	0.13	0.10	1091
TN (mg/l)	Smullen and Cave, 1998	2.39	2.00	2016
TKN (mg/l)	Smullen and Cave, 1998	1.73	1.47	2693
NO2-N and NO3-N (mg/l)	Smullen and Cave, 1998	0.66	0.53	2016
Total Cu (ug/l)	Smullen and Cave, 1998	13.4	11.1	1657
Total Pb (ug/l)	Smullen and Cave, 1998	67.5	50.7	2713
Total Zn (ug/l)	Smullen and Cave, 1998	162	129	2234
Total Cadmium (ug/l)	Smullen and Cave, 1998	0.7	0.5	150
Total Chromium (ug/l)	Bannerman et al., 1996	4.0	7.0	164
PAH (mg/l)	Rabanal and Grizzard, 1995	3.5	N/R	N/R
Oil and Grease (mg/l)	Crunkilton et al., 1996	3	N/R	N/R
FC (cfu/100ml)	Schueler, 1999	15,000	N/R	34
Diazinon	US-EPA, 1998	N/R	0.025	326
Atrazine	US-EPA, 1998	N/R	0.023	327
MTBE	Delzer, 1996	N/R	1.6	592

Notes: EMC = Event Mean Concentration; TSS = Total Suspended Solids; FC = Fecal Coliform Bacteria; BOD = Biological oxygen Demand; COD = Chemical oxygen Demand; TP = Total Phosphorus; TN = Total Nitrogen; SRP = Soluble Reactive Phosphorus; TKN = Total Kjeldahl Nitrogen; PAH = Poly-aromatic Hydrocarbons; N/R = Not Reported

Source: CWP, 2003

lower than in the NURP study, perhaps indicating that erosion and sediment control (ESC) best management practices (BMP) implemented since 1983 were somewhat effective. Metals were also generally lower in the 1999 study than in the NURP data, especially lead (Pb), likely due to the elimination of leaded gasoline. This study also highlighted the fact that the variability of stormwater quality can depend on contributing land use, seasonal factors (e.g., precipitation patterns), and geographic region.

The Center for Watershed Protection (CWP) has also compiled a database of national stormwater runoff water-quality data (CWP, 2003). This data is summarized in Table 3-8.

There can be significant regional differences in urban runoff water quality due to a variety of environmental factors. To a large extent, underlying geology and soils determine the natural background level of many water-quality constituents, such as nutrients or metals. In addition, soils and topography have a strong influence on erosion potential and sediment production. One of the most influential factors impacting runoff water quality are a region's precipitation characteristics. Annual rainfall, precipitation patterns, mean storm event volume, and the range of rainfall intensities all have been demonstrated to influence runoff water quality (Driver and Tasker, 1990). For example, the western U.S. tend to have distinct „wet" and „dry" seasons, whereas the eastern U.S. and Midwest generally have more dispersed, year-round precipitation. Within the western U.S., the Pacific Northwest tends to have most of its rainfall in long-duration, low-intensity storms, whereas the Southwest tends to see more short, high-intensity storm events. Because of these factors, stormwater runoff EMC levels for nutrients, sediment, and metals have a tendency to be higher in arid or semi-arid regions and to decrease slightly when annual rainfall increases (CWP, 2004).

Table 3-9: Event Mean Concentration (EMC) Values for Stormwater Runoff Pollutants for Various U.S. Climatic Regions

Location	National	Phoenix, AZ	San Diego, CA	Boise, ID	Denver, CO	Dallas, TX	Marquette, MI	Austin, TX	MD	KY	GA	FL	MN
Mean Annual Rainfall (in)	N/A	Low (7)	Low (10)	Low (11)	Low (15)	Med (28)	Med (32)	Med (32)	High (41)	High (41)	High (41)	High (41)	Snow (*)
Pollutant													
TSS (mg/l)	78	227	330	116	242	663	159	190	67	98	258	43	112
TN (mg/l)	2.39	3.26	4.55	4.13	4.06	2.70	1.87	2.35	N/R	2.37	2.52	1.74	4.30
TP (mg/l)	0.32	0.41	0.70	0.75	0.65	0.78	0.29	0.32	0.33	0.32	0.33	0.38	0.70
SRP (mg/l)	0.13	0.17	0.40	0.47	N/R	N/R	0.04	0.24	N/R	0.21	0.14	0.23	0.18
Cu (ug/l)	14	47	25	34	60	40	22	16	18	15	32	1	N/R
Pb (ug/l)	68	72	44	46	250	330	49	38	13	60	28	9	100
Zn (ug/l)	162	204	180	342	350	540	111	190	143	190	148	55	N/R
BOD (mg/l)	14	109	21	89	N/R	112	15.4	14	14.4	88	14	11	N/R
COD (mg/l)	52	239	105	261	227	106	66	98	N/R	38	73	64	112
# Sample Events	3000	40	36	15	35	32	12	78	107	21	81	66	49
Reference	1	2	3	4	5	6	7	8	9	10	11	11	12

Notes: EMC = Event Mean Concentration; TSS = Total Suspended Solids; BOD = Biological oxygen Demand; COD = Chemical oxygen Demand; TP = Total Phosphorus; TN = Total Nitrogen; SRP = Soluble Reactive Phosphorus; N/R = Not Reported

References: 1 – Smullen and Cave, 1998; 2 – Lopes et al., 1995; 3 – Schiff, 1996; 4 – Kjelstrom, 1995; 5 – DRCOG, 1983; 6- Brush et al., 1995; 7 – Steuer et al., 1997; 8 – Barrett et al., 1995; 9 – Barr, 1997; 10 – Evaldi et al., 1992; 11 – Thomas and McClelland, 1995; 12 – Oberts, 1994

Source: CWP, 2004

In colder regions, where snow is a significant form of precipitation, snowmelt can be a major source of urban runoff pollutants (Novotny and Chester, 1981). Snow tends to accumulate during the winter, and pollutants can build up in the snowpack due to atmospheric deposition, vehicular emissions, litter, and the application of de-icing products (e.g., salt and/or sand). As a result, relatively high concentrations of some pollutants can be detected during snowmelt events and in runoff from treated roads (CWP, 2004). The main concerns with regard to the hazards of chlorides in stormwater runoff include groundwater contamination, trace metal leaching from sediments, stratification of receiving water bodies, and direct toxic effects on aquatic biota (Marsalek, 2003).

A study in Minnesota measured pollutants in urban streams and found that as much as half of the annual sediment, nutrient, hydrocarbon, and metal loads could be attributed to snowmelt runoff (Oberts, 1994). High levels of chloride (road salt), BOD, and TSS have also been reported in snowmelt runoff (La Barre et al., 1973; Oliver et al., 1974; Horkeby and Malmquist, 1977; Pierstorff and Bishop, 1980; Scott and Wylie, 1980; Novotny and Chester, 1981; Boom and Marsalek, 1988; Marsalek, 2003). Table 3-9 summarizes stormwater runoff pollutant concentrations for different climatic regions of the U.S. (CWP, 2004).

In the decades between the NURP data being collected and now, much has been accomplished with regard to urban runoff source control, the treatment of stormwater runoff, and improvements in receiving water quality. The most comprehensive analysis of stormwater runoff quality is currently underway. In 2001, the University of Alabama and the Center for Watershed Protection (CWP) were awarded an EPA Office of Water grant to collect and evaluate stormwater data from a representative number of NPDES (National Pollutant Discharge Elimination System) MS4 (municipal separate storm sewer system) stormwater permit holders. The initial version of this database, the National Stormwater Quality Database (NSQD, 2004) is currently available from the CWP.

In the NSQD project, stormwater quality data and site descriptions are being collected and reviewed to describe the characteristics of national stormwater quality, to provide guidance for future sampling needs, and to enhance local stormwater management activities in areas having limited data. Over 10 years of monitoring data collected from more than 200 municipalities throughout the country have a great potential in characterizing the quality of stormwater runoff and comparing it against historical benchmarks. This project is creating a national database of stormwater monitoring data collected as part of the existing stormwater permit program, providing a scientific analysis of the data as well as recommendations for improving the quality and management value of future NPDES monitoring efforts (Pitt et al., 2004). Table 3-10 summarizes the NSQD findings to date. Table 3-11 shows a comparison between NURP and NSQD findings. Figure 3-3 shows a sample of the NSQD findings for one common urban runoff constituent (TSS).

Urban Wetland Water Quality: Puget Sound Case Study

In a study of Puget Sound Basin freshwater wetlands (Azous and Horner, 2001), many water quality parameters exhibited upward trends with increased urbanization. Median pH levels were particularly elevated in highly urbanized wetlands while DO experienced more modest increases. Median conductivity and NH3 levels were also significantly higher in urbanized wetlands than in non-urbanized wetlands. Finally, similar rates of increase in median concentrations of total suspended solids (TSS), soluble reactive phosphorus (SRP), fecal coliforms (FC), lead (Pb) and zinc (Zn) were found with each step in the urbanization process (Azous and Horner, 2001).

In the wetlands studied, low concentrations predominated, indicating minimal water quality impacts. Concentrations of lead (Pb), however, tended to violate water quality criteria for the protection of aquatic life (Azous and Horner, 2001). In both urbanized and non-urbanized wetlands, wetland morphology type was associated with varying levels of water quality parameters. Morphology refers to the shape, perimeter length, internal horizontal dimensions, and topography of the wetland as well as to water pooling and flow patterns. Higher levels of DO, pH, conductivity, NO3+NO2-N, SRP, FC, and Pb were found in flow-through wetlands. Flow-through wetlands (FT) are channelized and have clear flow gradients, while open water wetlands (OW) contain significant pooled areas with little or no flow gradient (Azous and Horner, 2001). A large proportion of FT wetlands is found in urban areas, due to wetland

Table 3-10: Median Values for Selected Stormwater Parameters for Standard Land-Use Categories					
WQ Parameter	Residential	Commercial	Industrial	Freeways	Open Space
TSS (mg/l)	48	43	77	99	51
BOD (mg/l)	9.0	11.9	9.0	8.0	4.2
COD (mg/l)	55	63	60	100	21
FC (mpn/100ml)	7750	4500	2500	1700	3100
NH3 (mg/l)	0.31	0.50	0.50	1.07	0.30
NO2 + NO3 (mg/l)	0.60	0.60	0.70	0.30	0.60
TKN (mg/l)	1.40	1.60	1.40	2.00	0.60
SRP (mg/l)	0.17	0.11	0.11	0.20	0.08
TP (mg/l)	0.30	0.22	0.26	0.25	0.25
Cd total (ug/l)	0.5	0.9	2.0	1.0	0.5
Cd dissolved (ug/l)	ND	0.3	0.6	0.7	ND
Cu total (ug/l)	12	17	22	35	5
Cu dissolved (ug/l)	7	8	8	11	ND
Pb total (ug/l)	12	18	25	25	5
Pb dissolved (ug/l)	3	5	5	2	ND
Ni total (ug/l)	5	7	16	9	ND
Ni dissolved (ug/l)	2	3	5	4	ND
Zn total (ug/l)	73	150	210	200	39
Zn dissolved (ug/l)	33	59	112	51	ND

Notes: TSS = Total Suspended Solids; BOD = Biochemical Oxygen Demand; COD = Chemical Oxygen Demand; FC = Fecal Coliform; TKN = Total Kjeldahl Nitrogen; SRP = Soluble Reactive Phosphorus; TP = Total Phosphorus; ND = Not Detected

Source: NSQD, 2004

Table 3-11: Comparison of Median Stormwater Quality for NURP and NSQD								
WQ Parameter	Overall		Residential		Commercial		Open Space	
	NSQD	NURP	NSQD	NURP	NSQD	NURP	NSQD	NURP
COD (mg/l)	53	65	55	73	63	57	21	40
TSS (mg/l)	58	100	48	101	43	69	51	70
Pb total (ug/l)	16	144	12	144	18	104	5	30
Cu total (ug/l)	16	34	12	33	17	29	5	11
Zn total (ug/l)	116	160	73	135	150	226	39	195
TKN (mg/l)	1.4	1.5	1.4	1.9	1.60	1.18	0.60	0.97
NO2 + NO3 (mg/l)	0.60	0.68	0.60	0.74	0.60	0.57	0.60	0.54
TP (mg/l)	0.27	0.33	0.30	0.38	0.22	0.20	0.25	0.12
SRP (mg/l)	0.12	0.12	0.17	0.14	0.11	0.08	0.08	0.03

Notes: COD = Chemical Oxygen Demand; TSS = Total Suspended solids; TKN = Total Kjeldahl Nitrogen; TP = Total Phosphorus; SRP = Soluble Reactive Phosphorus

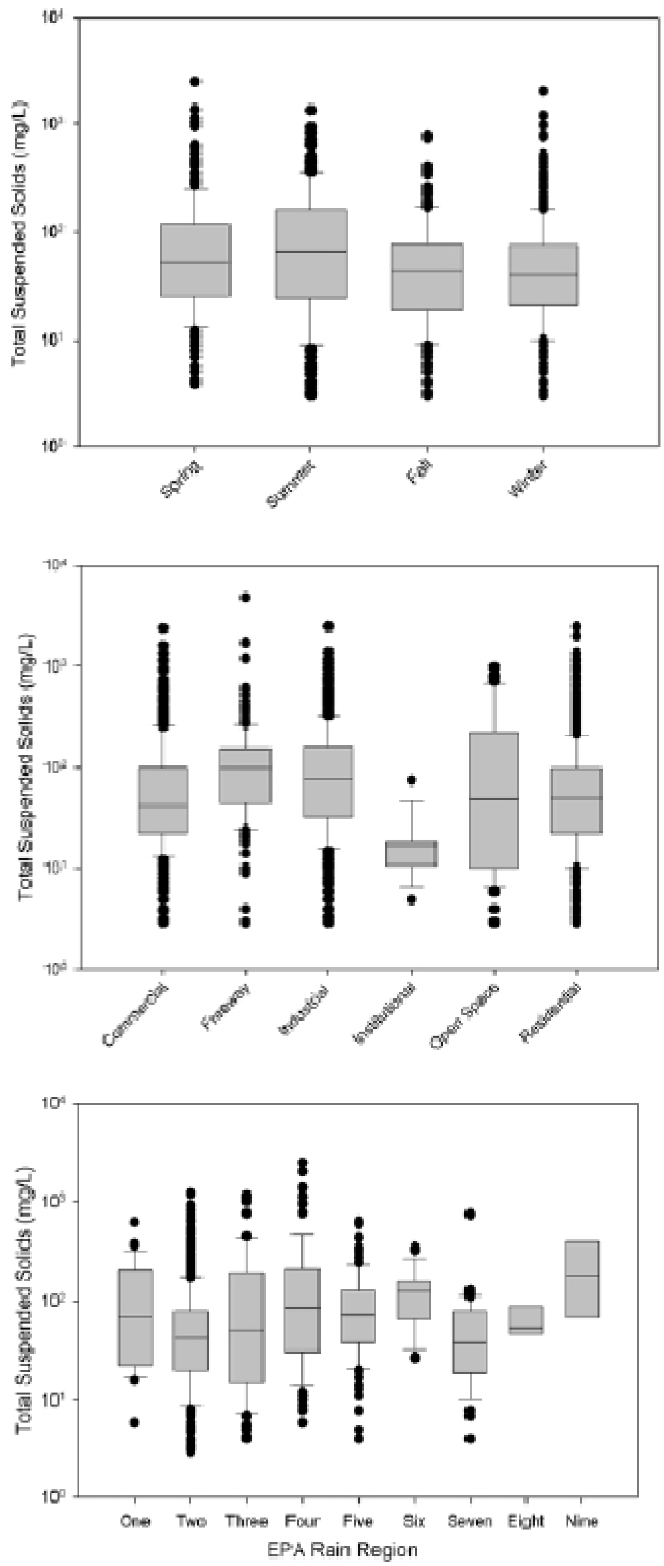
Source: NSQD, 2004

filling, stream channelization, and higher peak runoff flows, and this may help explain why pollutant levels trends are higher in these wetlands (Azous and Horner, 2001).

In the Puget Sound wetlands study, soil samples were collected once from each wetland during the summer dry period (July through September) for several years. Soil samples were taken from 3m to the side of vegetation transect lines wherever soils appeared transitional or completely different. These transitions were determined by small soil core samples or vegetation changes. Overall, two to five samples were collected from each wetland, with an average of four samples collected. The number of samples collected was related to the size and zonal complexity of the wetlands. Samples were taken from inlet zones in particular, because oxidation reduction potential (ORP) and one metal were found in significantly different levels in these locations. Soil samples were collected with a corer composed of a 10 cm (4 in) diameter ABS plastic pipe section ground to a sharp tip. A wooden rod was inserted horizontally through two holes near the top to provide leverage to twist the corer into the soil. Core samples were taken to a depth of 15 cm (6 in) and preserved by immediately placing them in bags sealed with tape. A standard 60-cm (2-ft) deep soil pit was also excavated at each sampling point not inundated above the surface. This pit was observed for depth to water table, horizontal definition (thickness of each layer and boundary type between), color (using Munsell notations), structure (grade, size, form, consistency, moistness), and the presence of roots and pores (Azous and Horner, 2001).

Sediment samples exhibited similar trends in urban and flow-through wetlands as the water quality parameters discussed previously. Median pH levels increased with each successive level of urbanization (Azous and Horner, 2001). Metals, including Pb, Zn, As (arsenic) and Cu (copper) also generally tended to increase with urbanization. As with water quality samples, sediment metal concentrations did not exceed severe effect thresholds based on the Washington State Department of Ecology. Some Cu and Pb mean and median concentrations exceeded lowest effect thresholds (Azous and Horner, 2001). While these metals tended to be found in greater concentrations in urban wetlands, they can also be found at elevated levels in non-urban wetlands.

Figure 3-3: Sample NSQD Findings



Source: NSQD, 2004

High Cu, Pb and TPH levels were seen in the two most impacted urban wetlands (Azous and Horner, 2001). Thus, local conditions may be more important factors in determining soil metal concentrations. Possible factors include the delivery of metals via precipitation, atmospheric dry-fall, dumping of metal trash, and leaching from old constructed embankments (Azous and Horner, 2001).

The impact of human activity and development on water quality varies widely between wetlands of different urbanization levels. For moderately urbanized wetlands, there is a mixed picture. Median total dissolved nitrogen concentrations (ammonia, nitrate, and nitrite) have been found to be more than 20 times higher than dissolved phosphorus, but phosphorus is the most important factor limiting plant and algal growth. As would be expected, these wetlands exhibit slightly elevated pH levels (median pH = 6.7). Dissolved oxygen is well below saturation, at times below 4 mg/l. Dissolved substances tend to be higher than in non-urbanized wetlands but are also somewhat variable. Suspended solids are only marginally higher than in non-urbanized wetlands but are also variable (Azous and Horner, 2001).

In highly urbanized wetlands, water quality samples revealed higher nutrient levels. Unlike non-urbanized or even low-moderately urbanized wetlands, these wetlands are likely to have median $NO_3 + NO_2$-N concentrations above 100 mg/l and total phosphorus (TP) over 50mg/l (Azous and Horner, 2001). In one study, FC and EC were shown to be significantly higher in highly urbanized wetlands. Many of these wetlands were within watersheds with low-density residential development (Azous and Horner, 2001).

An effort was made to correlate water quality conditions with watershed and wetland morphological characteristics. Acidity (pH), TSS, and conductivity showed the strongest ability to predict watershed and morphology characteristics. Pollutants such as TP, Zn and FC, which are often absorbed to particulates, also exhibited strong correlations with watershed conditions and morphology (Azous and Horner, 2001). On the other hand, forest cover was the best predictor of these water quality parameters. The next best land cover predictors of water quality were the percentage of impervious surface, forest-to-wetland areal ratio and morphology (Azous and Horner, 2001). A rise in the total impervious area will facilitate the delivery of TSS and increase conductivity. TSS and conductivity are directly and indirectly harmful to wetland biological communities (Azous and Horner, 2001).

These results suggest that a range of deforestation and development exists after which water quality will become degraded. Effective impervious area, which is the amount of land drained by a storm drainage system, was more predictive of water quality than total impervious area. As total impervious area approaches a range of 4 to 20 percent and forested area declines to between 0 to 15 percent, water quality will begin to decline (Azous and Horner, 2001).

Wetlands in developing areas are especially vulnerable to erosion caused by construction, which contributes to sediment levels. During these periods, both mean and median TSS values increase, although mean values show the greatest change. After construction is completed, and more surface area is covered with structures and vegetation, these values return to their approximate values before development. The sediments contributed by this erosion carry pollutants such as phosphorus and nitrogen (Azous and Horner, 2001).

Development also affects soils in wetlands. In the Puget Sound Basin wetlands, somewhat elevated pH levels prevailed. These soils were often aerobic, although many times their redox potentials were below levels at which oxygen is depleted. Metals such as Cu, Pb and Zn were higher in developing areas but did not usually approach severe effects thresholds (Azous and Horner, 2001). In a synoptic study of 73 wetlands, about 60 percent of which were urban and the rest non-urban, Pb levels were significantly different in both the inlet and emergent zones (Azous and Horner, 2001). In some soil samples, high toxicity levels were probably caused by the extraction and concentration of naturally occurring organic soil compounds during laboratory analysis. Samples from two wetlands, however, probably contained anthropogenic toxicants because the results indicated toxicity in the absence of any visible organic material (Azous and Horner, 2001).

For each region studied in the Puget Sound area, a regression was developed between the presence of crustal metals and toxic metals in relatively unimpacted wetlands. If the concentration of a toxic metal was above a 95 percent confidence level, it was probable that the metals were of anthropogenic origin. The results of this analysis echoed those described previously for urbanized wetlands. The regressions revealed a greater degree of toxic metal enrichment in the most urban wetlands (Azous and Horner, 2001).

Estimating Urban Runoff Pollutant Loading

The watershed assessment process provides the framework for evaluating watershed conditions and quantifying watershed characteristics (US-EPA, 2005). The objectives of the watershed assessment effort, pollution source information, and the water-quality data available largely determine what will be the most appropriate method for quantifying pollutant loading. In general, the approach chosen should be the simplest approach that meets the objectives of the watershed management program. Pollutant loading estimates are generally developed using a model or models.

Models can be useful tools for watershed and receiving-water assessments because they facilitate the analysis of complex systems and provide a method of estimating pollutant loading for a large array of land-use scenarios. Models are only as good as the data used for calibration and verification. There will always be some uncertainties present in all models and these uncertainties should be quantified and understood prior to using the selected model. Many models utilize literature-based values for water-quality concentrations to estimate pollutant loads (US-EPA 2005). Models have also become a standard part of most TMDL programs (US-EPA 1997). There are several recognized approaches used for estimating pollutant loadings for a drainage area or watershed basin. The three general approaches include:

- Unit-area loading;
- Simple empirical method; and
- Complex, computer-based models.

Unit-Area Loading

This method utilizes published *yield-values* to estimate pollutant loading for a specific land use. As mentioned earlier in this chapter, loading is the mass of pollutants delivered to a water body over a period of time and is usually given on an annual basis as kg/yr or lbs/yr. When ascribed to a particular land use, loading is sometimes termed yield or simply export per unit area of the land use (kg/ha-y or lbs/acre-y). Table 3-12 presents typical loadings for a number of pollutants and land uses. Although this table presents no ranges or statistics on the possible dispersion of these numbers when measurements are made, the variation is usually substantial from place to place in the same land use and from year to year at the same place.

This method is least likely to give accurate results because of the general lack of fit between the catchment of interest and the data collection location(s). To apply this method, consult a reference like Table 3-12, select the areal loading rate for each land use, multiply by the areas in each use, and sum the total loading for the pollutant of interest.

This method can be improved by producing some measure of uncertainty or error in the estimates. To do so, it is necessary to establish ranges of areal loadings from published literature or actual sampling, estimate

Table 3-12: Typical Pollutant Loadings (lbs/acre-yr) From Different Land Uses

Land-Use	TSS	TP	TKN	NH_3-N	NO_2-N and NO_3-N	BOD	COD	Pb	Zn	Cu	Cd
Commercial	1000	1.5	6.7	1.9	3.1	62	420	2.7	2.1	0.4	0.03
Parking Lot	400	0.7	5.1	2.0	2.9	47	270	0.8	0.8	0.06	0.01
High-Density Residential	420	1.0	4.2	0.8	2.0	27	170	0.8	0.7	0.03	0.01
Medium-Density Residential	250	0.3	2.5	0.5	1.4	13	50	0.05	0.1	0.03	0.01
Low-Density Residential	65	0.04	0.3	0.02	0.1	1	7	0.01	0.04	0.01	0.01
Highway	1700	0.9	7.9	1.5	4.2	n/a	n/a	4.5	2.1	0.37	0.02
Industrial	670	1.3	3.4	0.2	1.3	n/a	n/a	0.2	0.4	0.10	0.05
Shopping Center	440	0.5	3.1	0.5	1.7	n/a	n/a	1.1	0.6	0.09	0.01

Source: Based on Table 2.5 in Burton and Pitt, 2002 maximum and minimum and mean or median values of each pollutant, and then evaluate to determine if uncertainty or error could change the conclusions. Table 3-13 presents loading rate ranges based on unpublished data collected in the Pacific Northwest (PNW). The PNW regional data provided values for total phosphorus and total nitrogen for most land uses and all pollutants in road runoff, except fecal coliform. Accordingly, the regional data have narrower ranges than the remainder. Data such as that shown in Table 3-13 should be used with caution, because the concentrations of most pollutants vary considerably depending on regional characteristics in land use and climate, among other factors.

The use of published yield or unit-area loading values from specific sources, rather than for land-use categories, is also feasible. For example, a study in Maryland (Davis et al., 2001) examined the loading rates of metals (zinc, lead, copper, and cadmium) from several common sources in the urban environment. These included building siding and rooftops as well as automobile brakes, tires, and oil leakage. Loading estimates (mean, median, maximum, and minimum) were developed for each of these sources for all four metals (Davis et al., 2001). Specific data of this sort could be very useful for a variety of management scenarios.

Simple Empirical Method

The "Simple Method" was first developed by Schueler (1987) and further refined by the Center for Watershed Protection (CWP, 2003). This method requires data on watershed drainage area and impervious surface area, stormwater runoff pollutant concentrations, and

Table 3-13: Pollutant Loading (kg/ha-yr) Ranges for Various Land Uses

Land-Use Category		TSS	TP	TN	Pb	In	Cu	FC
Road	Minimum	281	0.59	1.3	0.49	0.18	0.03	7.1 E+07
	Maximum	723	1.50	3.5	1.10	0.45	0.09	2.8E+08
	Median	502	1.10	2.4	0.78	0.31	0.06	1 .8E+08
Commercial	Minimum	242	0.69	1.6	1.60	1.70	1.10	I.7E+09
	Maximum	1,369	0.91	8.8	4.70	4.90	3.20	9.5E+09
	Median	805	0.80	5.2	3.10	3.30	2.10	5.6E+09
Single family Low density Residential	Minimum	60	0.46	3.3	0.03	0.07	0.09	2.8E+09
	Maximum	340	0.64	4.7	0.09	0.20	0.27	1.6E+I0
	Median	200	0.55	4.0	0.06	0.13	0.18	9.3E+09
Single family High density Residential	Minimum	97	0.54	4.0	0.05	0.11	0.15	4.5E+09
	Maximum	547	0.76	5.6	0.15	0.33.	0.45	2.6E+I0
	Median	322	0.65	5.8	0.10	0.22	0.30	1.5E+I0
Multifamily Residential	Minimum	133	0.59	4.7	0.35	0.17	0.17	6.3E+09
	Maximum	755	0.81	6.6	1.05	0.51	0.34	3.6E+I0
	Median	444	0.70	5.6	0.70	0.34	0.51	2.1E+I0
Forest	Minimum	26	0.10	1.1	0.01	0.01	0.02	1 .2E+09
	Maximum	146	0.13	2.8	0.03	0.03	0.03	6.8E+09
	Median	86	0.11	2.0	0.02	0.02	0.03	4.0E+09
Grass	Minimum	80	0.01	1.2	0.03	0.02	0.02	4.8E+09
	Maximum	588	0.25	7.1	0.10	0.17	0.04	2.7E+I0
	Median	346	0.13	4.2	0.07	0.10	0.03	1.6E+ 10
Pasture	Minimum	103	0.01	1.2	0.004	0.02	0.02	4.8E+09
	Maximum	583	0.25	7.1	0.015	0.17	0.04	2.7E+ 10
	Median	343	0.13	4.2	0.010	0.10	0.03	1.6E+ 10

Source: Horner, 1992

annual precipitation. With the Simple Method, land use can be divided into specific types, such as residential, commercial, industrial, and roadway. Using this data, the annual pollutant loads for each type of land use can be calculated. Alternatively, generalized pollutant values for land uses such as new suburban areas, older urban areas, central business districts, and highways can be utilized. Stormwater pollutant concentrations can be estimated from local or regional data or from national data sources. Tables 3-6 through 3-11 contain the type of data required for this method.

As has been discussed, stormwater pollutant concentrations tend to be highly variable for a number of reasons. Because of this variability, it is difficult to establish different concentrations for each land use. The original Simple Method Model used NURP data for the representative pollutant concentrations. Utilizing a more recent and regionally specific database would, in general, be more accurate for this purpose. If no regional or local data exists, the Simple Method could be utilized using a median urban runoff value, derived from NURP data (US-EPA 1982), of 20,000 MPN/100ml.

Data from other sources can supplement the NURP values, and the use of EMC data from local measurements should yield superior estimates. Pollutant load values from extensive regional or local sampling programs could be the most useful. For example, water-quality studies from Western Washington and Oregon, which are compatible, have been combined to form a data set for different land use categories in the PNW Chandler (1993 and 1994) These studies found a distinction between residential, commercial, and industrial land use-related EMC values and the results of the NURP research. On the other hand, a study that only includes a small number of EMC data cannot accurately determine average runoff concentrations and may not be useful in supplementing or replacing recognized EMC values such as the NURP data. If this is the case, previously published data sets should be used instead. Additionally, it is not always advisable to obtain additional EMC data due to the additional expenses involved. It may be better to use a cost-effectiveness analysis to determine if increasing the amount of EMC data is worth it. This is especially true in light of the fact that a great deal of data is typically available, for example from municipal NPDES stormwater permit applications, that can be used to estimate runoff concentrations from a variety of land uses.

The Simple Method estimates pollutant loads for chemical constituents as a product of annual runoff volume and pollutant concentration, as (CWP, 2003):

$$L = 0.226 * R * C * A$$

where: L = Annual load (lbs)

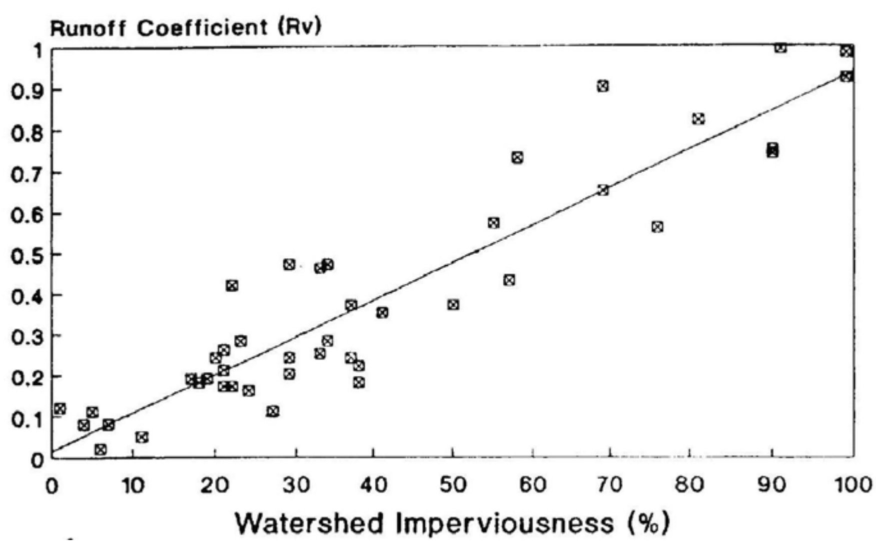
R = Annual runoff (inches)

C = Pollutant concentration (mg/l)

A = Area (acres)

0.226 = Unit conversion factor

Figure 3-4: Relationship Between Stormwater Runoff and Impervious Surface Area



Source: Schueler, 1995

For bacteria, the equation is slightly different to account for the differences in units. The modified equation for bacteria is (CWP, 2003):

$$L = 1.03 * 10^{-3} * R * C * A$$

where: L = Annual load (Billion Colonies)

R = Annual runoff (inches)

C = Bacteria concentration (#/100 mL)

A = Area (acres)

$1.03 * 10^{-3}$ = Unit conversion factor

The Simple Method calculates annual runoff as a product of annual runoff volume and a runoff coefficient (Rv). Runoff volume is calculated as (CWP, 2003):

$$R = P * Pj * Rv$$

where: R = Annual runoff (inches)

P = Annual rainfall (inches)

Pj = Fraction of annual rainfall events that produce runoff (usually 0.9)

Rv = Runoff coefficient

In the Simple Method, the runoff coefficient is calculated based on impervious cover in the sub-watershed. This relationship is based on empirical data. Although there is some variability in the data, watershed imperviousness does appear to be a reasonable predictor of Rv (Figure 3-4). The following equation represents the best-fit line for the data set ($N = 47$, $R2 = 0.71$) based on data collected by Schueler (1987). This model uses different impervious cover values for separate land uses within a sub-watershed.

$$Rv = 0.05+0.9Ia$$

Where: Ia = Impervious fraction

Limitations of the Simple Method

The Simple Method should provide reasonable estimates of changes in pollutant export resulting from urban development activities. However, several caveats should be kept in mind when applying this method. The Simple Method is most appropriate for assessing and comparing the relative stormflow pollutant load changes of different land-use and stormwater management scenarios. It provides estimates of storm pollutant export that are probably close to the "true" but unknown value for a development site, catchment, or sub-watershed. However, it is very important not to overemphasize the precision of the results obtained. The simple method provides a general planning estimate of likely storm pollutant export from areas at the scale of a development site, catchment, or sub-watershed. More sophisticated modeling may be needed to analyze larger and more complex watersheds.

In a comparison of several PNW watersheds, Chandler (1993 and 1994) found that the Schueler (1987) Simple Model loading estimates usually agreed, within a factor of two, with estimates made by much more involved and expensive modeling procedures. Chandler (1993 and 1994) utilized the Simple Model in four case-study comparisons with more complex models, including the EPA Stormwater Management Model (SWMM) and the Hydrologic Simulation Program FORTRAN (HSPF) model. Chandler (1993 and 1994) concluded that there was no compelling reason for using complex models when estimating annual pollutant loading under most situations.

In addition, the Simple Method only estimates pollutant loads generated during storm events. It does not consider pollutants associated with baseflow volume. Typically, baseflow is negligible or non-existent at the scale of a single development site, and can be safely neglected. However, catchments and sub-watersheds do generate baseflow volume. Pollutant loads in baseflow are generally low and can seldom be distinguished from natural background levels. Consequently, baseflow pollutant loads normally constitute only a small fraction of the total pollutant load delivered from an urban area. Nevertheless, it is important to remember that the load estimates refer only to storm event-derived loads and should not be confused with the total pollutant load from an area. This is particularly important when the development density of an area is low.

Computer-Based Models

There are a wide variety of computer models available today that can be used for surface water and stormwater quality assessments. Many of these models are available in the public domain and have been developed and tested by resource agencies. Regionally or locally specific versions of many of these models are also common. In comparison to the approaches outlined previously, computer-based models provide a more complex approach to estimating pollutant loading and also often offer a means of evaluating various management alternatives (US-EPA, 2005). Detailed coverage of these models is beyond the scope of this chapter. The US-EPA *Handbook for Developing Watershed Plans* (US-EPA, 2005) contains a comprehensive discussion of computer-based models in Chapter 8 of that publication.

Examples of comprehensive computerized models include Stormwater Management Model (SWMM), Better Assessment Science Integrating Point and Non-Point Sources (BASINS), the Hydrologic Simulation Program Fortran (HSPF), Source Loading and Management Model (SLAMM), Storage, Treatment, and Overflow Runoff Model (STORM), and Spatially Referenced Regression on Watershed Attributes (SPARROW). These are only a few of the computer-based pollutant-loading estimation models available (see US-EPA 2005 Table 8-4 for a more complete listing).

In general, computer-based models contain hydrologic and water quality components and have statistical or mathematical algorithms that represent the mechanisms generating and transporting runoff and pollutants. The hydrologic components of both SWMM and HSPF stem from the Stanford Watershed Model, first introduced almost 25 years ago, and produce continuous hydrograph simulations. In addition to these relatively complex computer-based models, there are numerous "spreadsheet" level models that have been developed by local and regional water-quality practitioners. In almost all cases, computer-based models need to be calibrated and validated using locally appropriate water-quality data (US-EPA, 2005), which, depending on the watershed under study, can be a time-consuming and relatively costly effort.

Most computer-based models structure the water quality components on a mass balance framework that represents the rate of change in pollutant mass as the difference between pollutant additions and losses. Additions, considered to be pollutant deposition, are computed as a linear function of time. Soil erosion is usually calculated according to the Universal Soil Loss Equation (USLE). Losses are represented by a first-order wash-off function (i.e., loss rate is considered to be a function of pollutant mass present); other losses are modeled in mathematically similar ways. For example, both organic matter decomposition and bacterial die-off are considered first-order reactions. Some models, like SWMM, have both a receiving water and runoff component. These models treat some of the transformation processes that can occur in water (e.g., dissolved oxygen depletion according to the Streeter-Phelps equation or FC die-off using the Mancini equation). However, no model can fully represent all of these numerous and complex processes.

The BASINS model is a physical process-based analytical model developed by the US-EPA and typically used for watershed-based hydrologic and water-quality assessments. For example, BASINS was used to model the East Fork of the Little Miami River (Tong and Chen, 2002). The HSPF model can be used as a component of the BASINS model (Bergman et al., 2002) or as a stand-alone model (Im et al., 2003). The SPARROW model is a statistical-regression, watershed-based model developed by the USGS (Smith et al., 1997) and used primarily for water-quality modeling (Alexander et al., 2004). Many computer-based models utilize regression equations to describe pollutant characteristics (Driver and Tasker, 1990).

There are also a number of so-called "build-up and wash-off" models that simulate pollutant build-up on impervious surfaces and use rainfall data to estimate wash-off loading. The main limitation of these models is that model-controlling factors can greatly vary with surface characteristics, so calibration with actual field measurements is needed. These models can work well with calibration and can model intra-storm variations in runoff water quality, which is a key advantage. These models are often used for ranking or prioritizing, but not for predicting actual runoff water quality. SLAMM was developed to evaluate the effects of urban development characteristics and runoff control measures on pollutant discharges. This model examines runoff from individual drainage basins with particular land-use and control practices (Burton and Pitt 2002).

Most models require substantial local data to set variable parameters in the calibration and verification phases. They also require considerable technical skill and commitment from personnel. Therefore, only those prepared to commit the resources to database development and expertise should embark on using these models. Most models used today also utilize the geographic information system (GIS) for data input and presentation of results.

In many situations, the use of computer-based models may not be merited, but in other cases, it may be helpful in determining the magnitude of the water-quality problem or aid in finding a solution. Computer models can also extend data collected and enhance findings. In addition, they can be quite useful in running a variety of scenarios to help frame the water quality problem. Examples of this include worst-case, full build-out scenarios or potential BMP scenarios to estimate the effectiveness of a range of treatment options. In any case, model selection should be linked to the project objectives and must be compatible with the data available. In almost all cases, using the simplest model that will meet the project objectives is likely the best course to take. In all cases, models should be calibrated and verified with independent, local or regionally specific data.

A good example of a watershed-scale, computer-based model dealing with multiple water-quality parameters and their impact on receiving waters is the Sinclair-Dyes Inlet TMDL Project in the Puget Sound, Washington (Johnston et al. 2003). This model has a watershed component (HSPF) linked to a receiving-water model (CH3D) that includes dynamic loading from the contributing watershed and hydro-dynamic mixing in the receiving waters of Sinclair-Dyes Inlet. The results of this model can be viewed at www.ecy.wa.gov/programs/wq/tmdl/sinclair-dyes_inlets/index.html

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CHAPTER 4

Bio-physical Impacts of Urbanization on Aquatic Ecosystems

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Bio-physical Impacts of Urbanization on Aquatic Ecosystems

The Clean Water Act (CWA) describes water quality as the combination of chemical, physical, and biological attributes of a water body. This chapter deals mainly with the biological and physical effects of watershed development on aquatic ecosystems. Physio-chemical water quality was discussed in detail in the previous chapter. The physio-chemical effects of urbanization, commonly referred to as water pollution, are discussed in this chapter only as they apply to their impact on aquatic biota. The wide array of pollutants entering aquatic ecosystems along with urban runoff can cause numerous potential biological effects. Other biological stresses often associated with modification of the hydrologic regime or changes in physical habitat also typically accompany watershed development. The goal of this chapter is to provide a synthesis of the current scientific research that covers the cumulative effects of urbanization on aquatic ecosystems, including streams, rivers, lakes, wetlands, and estuaries. Table 4-1 summarizes the impacts of urbanization on these aquatic systems.

The majority of this chapter focuses on freshwater lotic (flowing waters) or stream-river ecosystems, but lentic (non-flowing) systems, such as lakes and wetlands, are also covered, as are estuaries and nearshore areas, to a lesser extent. As Table 4-1 shows, the impacts of urbanization include chemical effects such as degraded water quality; physical effects such as altered hydrology, degraded habitat, and modified geomorphology; and biological effects including altered biotic interactions, food web (trophic) changes, chronic (sublethal) toxicity, and acute (lethal) toxicity. This chapter also presents illustrations of the complex, interdisciplinary nature of aquatic biological impacts. Subjects covered include the role of urban runoff in lake eutrophication, metals found in stormwater runoff and their effects on aquatic organisms, thermal impacts of riparian encroachment, and the fish habitat impacts of watershed development and stormwater runoff. How the many urban stressors might affect the biota in a receiving water is very complex, imperfectly understood, and hard to forecast with assurance. The multiple stressors that often accompany urbanization can interact synergistically or antagonistically. In addition, the receptor organisms under stress can interact with one another. The sum total of these interactions within an aquatic ecosystem represents the cumulative impacts of urbanization.

Background

One of the confusing aspects of water-quality management is that often only the chemical component of water quality is considered. Water-quality criteria are the main regulatory tools used in managing receiving waters. These are typically concentrations of specific chemical pollutants set so as to protect human health and beneficial uses of receiving waters (including aquatic biota) from adverse impacts. However, relying solely on these water-quality criteria to manage urban runoff is often not an effective approach, because biological and

Table 4-1: Summary of the Impacts of Urbanization on Aquatic Ecosystems		
Environmental Concern	**Potential Impact**	**Cause/Source**
Increase in runoff-driven peak or bankfull stream flows	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff volume due to an increase in basin imperviousness
Increase in runoff-driven flooding frequency and duration	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff volume due to an increase in basin imperviousness
Increase in wetland water level fluctuations	Degradation of aquatic habitat and/or loss of sensitive species	Increased stormwater runoff due to an increase in basin imperviousness
Decrease in dry season baseflows	Reduced aquatic habitat and less water for human consumption, irrigation, or recreational use	Water withdrawals and/or less natural infiltration due to an increase in basin imperviousness
Streambank erosion and stream channel enlargement	Degradation of aquatic habitat and increased fine sediment production	Increase in stormwater runoff driven stream flow due to an increase in basin imperviousness
Stream channel modification due to hydrologic changes and human alteration	Degradation of aquatic habitat and increased fine sediment production	Increase in stormwater runoff driven stream flow and/or channel alterations such as levees and dikes
Streambed scour and incision	Degradation of aquatic habitat and loss of benthic organisms due to washout	Increase in stormwater runoff driven stream flow due to an increase in basin imperviousness
Excessive turbidity	Degradation of aquatic habitat and/or loss of sensitive species due to physiological and /or behavioral interference	Increase in stormwater runoff driven stream flow and subsequent streambank erosion due to an increase in basin imperviousness
Fine sediment deposition	Degradation of aquatic habitat and loss of benthic organisms due to fine sediment smothering	Increase in stormwater runoff driven stream flow and subsequent streambank erosion due to an increase in basin imperviousness
Sediment contamination	Degradation of aquatic habitat and/or loss of sensitive benthic species	Stormwater runoff pollutants
Loss of riparian integrity	Degradation of riparian habitat quality and quantity, as well as riparian corridor fragmentation	Human development encroachment and stream road crossings
Proliferation of exotic and invasive species	Displacement of natural species and degradation of aquatic habitat	Encroachment of urban development
Elevated water temperature	Lethal and non-lethal stress to aquatic organisms – reduced DO levels	Loss of riparian forest shade and direct runoff of high temperature stormwater from impervious surfaces
Low dissolved oxygen (DO) levels	Lethal and non-lethal stress to aquatic organisms	Stormwater runoff containing fertilizers and wastewater treatment system effluent
Lake and estuary nutrient eutrophication	Degradation of aquatic habitat and low DO levels	Stormwater runoff containing fertilizers and wastewater treatment system effluent
Bacterial pollution	Human health (contact recreation and drinking water) concerns, increases in diseases to aquatic organisms, and degradation of shellfish harvest beds	Stormwater runoff containing livestock manure, pet waste, and wastewater treatment system effluent
Toxic chemical water pollution	Human health (contact recreation and drinking water) concerns, as well as bioaccumulation and toxicity to aquatic organisms	Stormwater runoff containing toxic metals, pesticides, herbicides, and industrial chemical contaminants
Reduced organic matter (OM) and large woody debris (LWD)	Degradation of aquatic habitat and loss of sensitive species	Loss or degradation of riparian forest and floodplain due to development encroachment
Decline in aquatic plant diversity	Alteration of natural food web structure and function	Cumulative impacts of urbanization
Decline in aquatic invertebrate diversity	Alteration of natural food web structure and function	Cumulative impacts of urbanization
Decline in amphibian diversity	Loss of ecologically important species	Cumulative impacts of urbanization
Decline in fish diversity and abundance	Loss of ecologically important species	Cumulative impacts of urbanization

ecological impacts can occur in an ecosystem at levels well below these chemical criteria.

This dilemma can be explained by several factors characteristic of the typical urbanized environment. As discussed earlier, water quality is assessed not just by chemical criteria, but there are physical and biological aspects to consider as well. These impacts include the modification of natural hydrologic regime, geomorphic changes in ecosystem structure, the degradation of physical habitat, disruption of ecological function or processes, and the biological changes to be discussed in this chapter.

Even from the perspective of conventional chemical toxicity alone, conventional (regulatory) water-quality criteria do not represent the complex and variable exposure patterns related to urban runoff or the cumulative impacts of long-term exposure to stormwater pollutant loadings. These criteria also do not account for any physio-chemical transformations that occur in the natural or built environment. In addition, there are numerous potential interactions within the ecosystem that cannot be accounted for using chemical criteria alone. As noted in the previous chapter, stormwater pollutant concentrations are often well below acute toxicity levels as well as below chronic toxicity levels. This is typically because the quantity of urban runoff usually dilutes pollutant levels in receiving waters (see discussion in Chapter 3). However, continued stormwater runoff inputs into streams, lakes, wetlands, and estuaries, even at low contaminant concentration levels, may eventually lead to long-term biological damage. Cumulative stress from poor water quality can result in chronic toxicity effects or bioaccumulation impacts. Pollutant accumulations in aquatic sediments can also have a long-term negative impact on benthic organisms or the embryonic stages of aquatic organisms that utilize the benthic environment.

Direct and indirect (or downstream) impacts of water quality degradation are another issue related to urban runoff impacts. In most cases, both scales of impact are present. Direct impacts are those that are present in surface waters that receive stormwater runoff directly from developed (e.g. impervious) drainage areas. Studies of direct impacts tend to focus on the hydrologic or geomorphic aspects of urban runoff. Indirect impacts are those that impact receiving waters downstream of the source, such as rivers, lakes, nearshore areas, and estuaries. In general, indirect impacts are mainly due to the physio-chemical water-quality effects of urbanization, but there is some overlap between the two scales.

Hydrologic Impacts

Landscape Alteration

Urbanization is one of the most widespread and rapidly growing forms of landscape modification affecting aquatic ecosystems. Just over 5 percent of the total surface area of the U.S. is covered by development (e.g. urbanization) related land use (EOS, 2004). Although the total land area currently occupied by urbanization (i.e., residential, commercial, and industrial development) remains relatively low in comparison to agricultural or other human land-use activities, the trend toward greater urbanization continues (Elvidge et al., 2004). According to the 2000 United States Census (USCB, 2001), approximately 30 percent of the population lives in urban areas and 50 percent in suburban areas, with the remaining 20 percent in rural areas. From an ecosystem perspective, the ecological footprint of urbanization has been shown to be significant in many cases (Folke et al., 1997). For example, it has been estimated that urbanized areas produce more than three quarters of global greenhouse gas emissions (Grimm et al., 2000). Urban development and related human activities can also produce very high local extinction rates for natural biota and can often result in the spread of exotic or invasive species (McKinney, 2002).

Urbanization can be characterized as an increase in human population density, coupled with an increase in per capita consumption of natural resources and extensive modification of the natural landscape, creating a built environment that is inherently not sustainable over the long term and often continues to expand into natural areas (McDonnell and Pickett, 1990). The landscape alterations accompanying urbanization tend to be more long lasting than other human land uses. For example, throughout much of New England, native forest cover has been steadily increasing in area over the last century, restoring areas impacted by historic logging and agriculture, whereas urbanized areas of the same region continue to persist or have significantly expanded (Stein et al., 2000). Generally, in urbanizing watersheds, water pollution and stormwater runoff are related to human habitation and the resultant increase in human land uses.

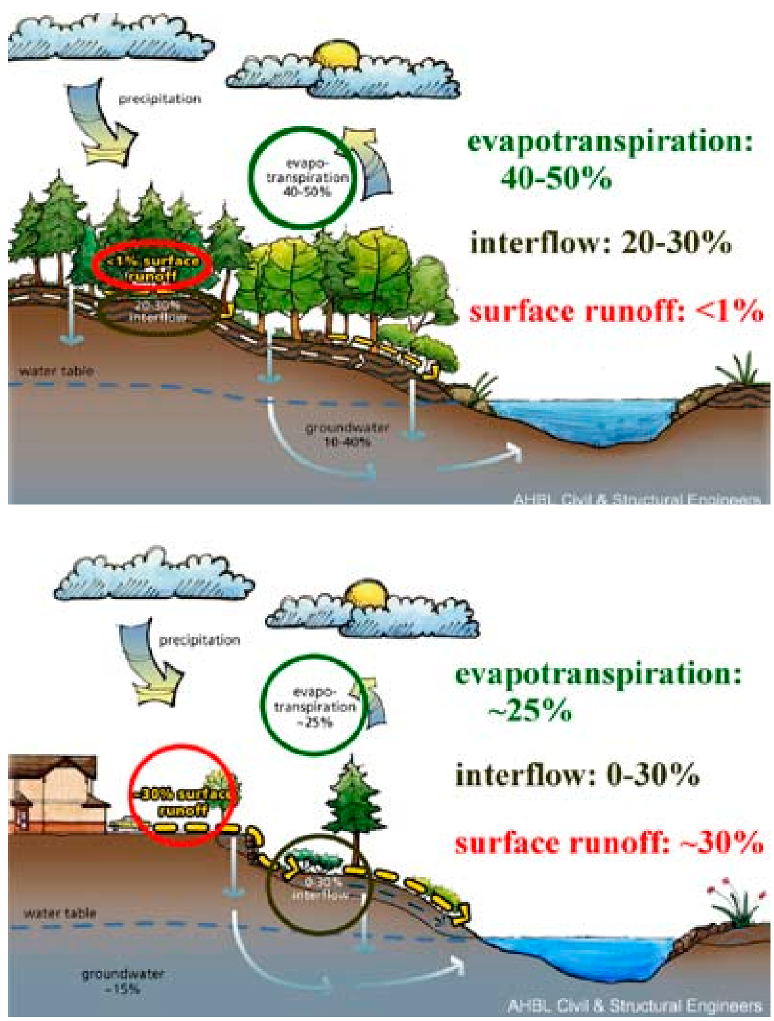
Savani and Kammerer (1961) first discussed the relationship between natural land cover and developed land use with respect to the stages of urbanization. This early research identified four stages of urbanization, each associated with characteristic changes in the hydrologic regime. These stages are rural, early urban (now called low-density suburban), middle urban (high-density suburban), and late urban. According to Savani and Kammerer (1961), during the rural stage of development, infiltration and evapo-transpiration are still the key components of the water cycle because the landscape is still predominantly unchanged from a hydrologic perspective. The early urban stage is characterized by large-lot development, where much of the natural vegetation is retained and impervious surfaces are just beginning to affect the basin hydrology. In the middle urban or suburban stage, impervious surfaces are beginning to dominate the landscape, with residential and commercial land uses being the most common. In the late urban stage, nearly all the natural vegetation has been removed, and impervious surfaces dominate the watershed landscape.

One of the most obvious manifestations of watershed development is the proliferation of impervious surfaces in the urbanizing landscape. Impervious surfaces can be broadly defined as any portion of the built environment that does not maintain the natural hydrologic regime. Impervious surfaces tend to inhibit or prevent infiltration and groundwater recharge. Impervious areas also tend to have less evapo-transpiration than natural areas. From a hydrologic perspective, development alters the natural landscape by removing native vegetation, disregarding local topography, and disturbing (through removal and/or compaction) the natural soil structure. Urbanization is typically accompanied by a reduction in rainfall interception, evapo-transpiration, and infiltration (Figure 4-1). Figure 4-2 shows the progression of impervious surface area and the changes in the hydrologic regime as development increases.

Impervious surfaces include roads, parking lots, sidewalks, driveways, and building rooftops. To a lesser extent, lawns, landscaped areas, golf courses, and parks can also be impervious (Schueler, 1995). These turf or landscaped areas are often directly connected to impervious areas and can contribute a significant fraction of the total runoff from built areas (Schueler, 1995). In addition, construction sites, agricultural croplands, quarries, and other areas of bare ground also contribute runoff volume. Impervious surface area tends to be correlated to human population density (Stankowski, 1972).

Although water resource degradation from urban runoff pollution is often considered the leading cause of ecological damage, this is not always the primary cause of water quality problems. The shift in the natural hydrologic regime from an infiltration-dominated scheme to one dominated by surface runoff resulting from watershed urbanization can have significant ramifications on river and stream hydrology (Dunne and Leopold, 1978). Due to the loss of infiltration, there is a reduction in groundwater recharge that can lead to lower dry-weather baseflows in surface waters. The relationship between imperviousness and runoff is

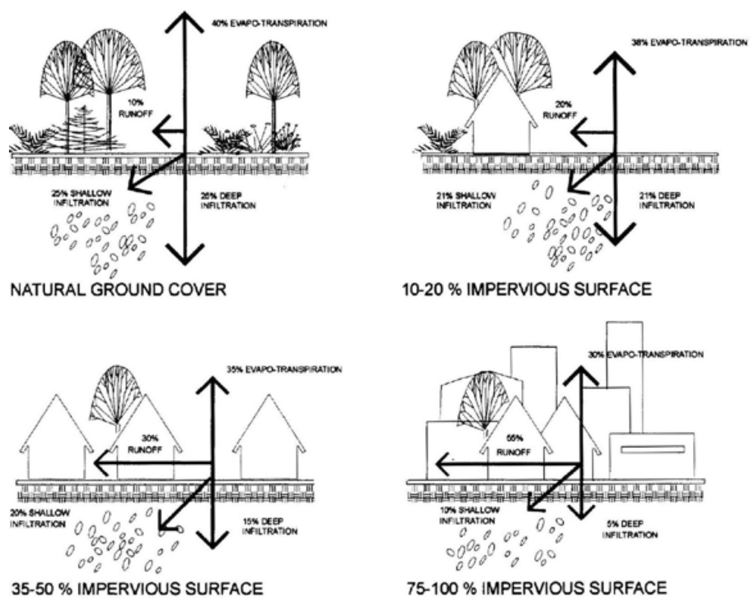
Figure 4-1: Comparison between the hydrologic regime for a natural, undeveloped watershed (upper) and an urbanized watershed in the Pacific Northwest



illustrated in Figure 4-3. The runoff coefficient reflects the fraction of rainfall volume that is converted to runoff. Runoff coefficient tends to closely track the percentage of impervious surface area in a given watershed, except at low levels of development where vegetation cover, soil conditions, and slope factors also influence the partitioning of rainfall. Impervious surfaces are hydrologically active, meaning they generate surface runoff instead of absorbing precipitation (Novotny and Chesters, 1981).

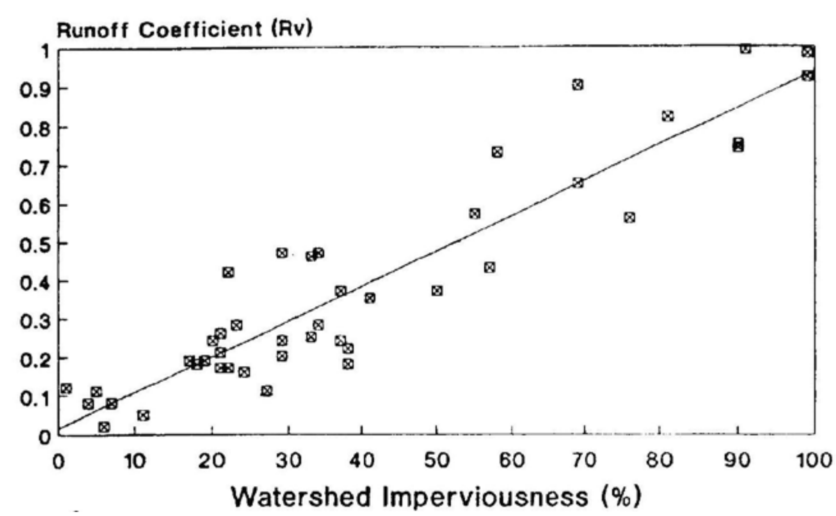
The total fraction of a watershed that is covered by impervious surface areas is typically referred to as the percent total impervious area (%TIA). The %TIA of a watershed is a landscape-level indicator that integrates several concurrent interactions influencing the hydrologic regime as well as water quality (McGriff, 1972; Graham et al., 1974; Dunne and Leopold, 1978; Alley and Veenhuis, 1983; Schueler, 1994; Arnold and Gibbons, 1996; May et al., 1997; EPA, 1997). Another impervious term commonly used in urban watershed work, especially in the modeling arena, is effective

Figure 4-2: Typical Progression of Hydrologic Changes in Urbanizing Watersheds as Imperviousness Increases With Development



Source: Schueler, 1994

Figure 4-3: Relationship Between Imperviousness and Stormwater Runoff

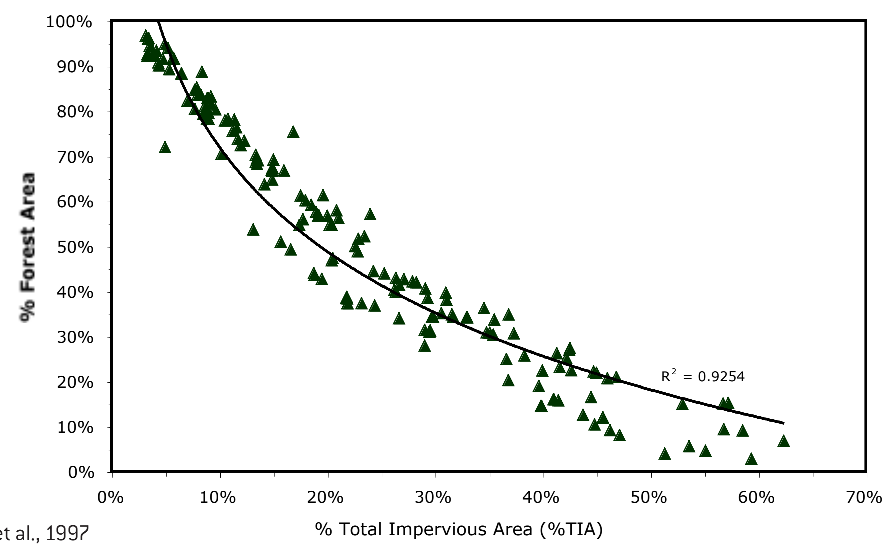


Source: Schueler, 1994

impervious area (%EIA). The %EIA is that portion of the impervious surfaces that is directly connected (via open channels or stormwater piping) to the natural drainage network (Alley and Veenhuis, 1983).

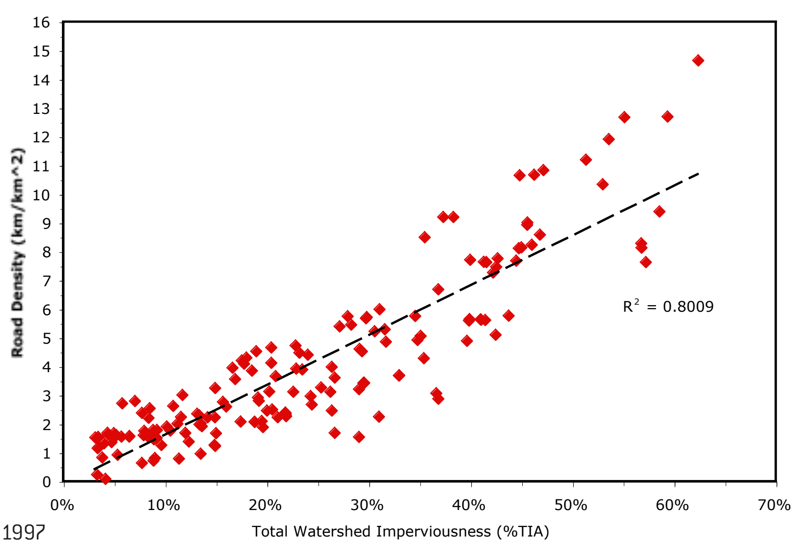
Another useful indicator of landscape-scale changes in watershed condition is the fraction of the basin that is covered by natural vegetation. In many areas, forest cover is the key parameter, but in other regions, prairie or shrub-savannah could be the key natural vegetation community. In any case, native vegetation tends to be adapted to local climate conditions and soil characteristics, making it the land cover that best supports the natural hydrologic regime. In general, urbanization tends to reduce natural vegetation land cover, while increasing impervious surface area associated with the variety of land uses present in the built environment. In most regions, the fraction of the watershed covered by natural vegetation is inversely correlated with imperviousness. For example, in the Puget Sound region of the Pacific Northwest, forest cover and imperviousness are strongly interrelated (see Figure 4-4), as are road density and imperviousness (see Figure 4-5).

Figure 4-4: Relationship Between Forest Cover and Impervious Surface Area in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

Figure 4-5: Relationship Between Road Density and Impervious Surface Area in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

Hydrological analyses suggest that maintaining forest cover is more important than limiting impervious-area percentages, at least at rural residential densities where zoning effectively limits the range of imperviousness to relatively low levels (typically < 10 percent TIA). However, without clearing limitations, the area of natural forest cover can vary widely (Booth et al., 2002). Consequently, both types of land-cover control (i.e., forest retention and impervious limitation) are likely critical to protecting aquatic resources. In rural areas, at the lower end of the development spectrum, current research indicates that retention of forest cover may be more important than limiting impervious surfaces (Booth et al., 2002). Degraded watersheds with less than 10 percent imperviousness and less than 65 percent forest cover are common ("cleared rural"); in contrast, virtually no watersheds with more than 10 percent imperviousness that have also retained at least 65 percent forest cover ("forested urban") exist in the Puget Sound region (Booth et al., 2002).

A study from western Washington illustrates the changes in hydrologic function that occur during the development process (Burges et al., 1998). To estimate the hydrologic balance for two basins in close proximity, an approach was used combining hydrologic modeling and simple monitoring. At the time of the study, both basins were in suburban areas, but one was relatively undeveloped, while the other was suburban in land use. Before being developed, the Novelty Hill and Klahanie basins were hydrologically similar. Both study basins are in the same geological region and were once largely forested. Novelty Hill was significantly deforested, and 30 percent of the area was covered with impervious surfaces. In this study, Novelty Hill had a faster flow response, higher peak flow, and longer time of discharge. Also, there was more flow response when there was preceding wetness in the soil. For the annual water balance in this basin (the difference between precipitation and catchment outflow), 69 to 88 percent of annual precipitation left as groundwater recharge or evapo-transpiration (Burges et al., 1998). Because the soil at Novelty Hill is deeper and less disturbed than at Klahanie, it takes more precipitation to saturate. In the developed Klahanie basin, 44 to 48 percent of the annual precipitation left as catchment outflow, as opposed to about 12 to 30 percent in Novelty Hill (Burges, et al., 1998). One of the most interesting findings of this study was that runoff from what are considered pervious areas such as lawns and landscaped areas accounted for 40 to 60 percent of the total annual runoff in the developed basin (Burges et al., 1998). In addition, the loss of local depressional storage likely influences hydrologic function of lawns and landscaped areas converted from natural forested areas. This study also illustrates that imperviousness encompasses much more that just paved surfaces.

Urban Hydrologic Regime

This section focuses on changes in runoff and stream-flow because they are common in urbanizing watersheds and often cause dramatic changes in basin hydrology. Hydrologic change also influences the whole range of environmental features that affect aquatic biota-flow regime, aquatic habitat structure, water quality, biotic interactions, and food sources (Karr, 1991). Although runoff and stream-flow regime are important, they are by no means the only drivers of aquatic health.

As has been discussed, urbanization alters the hydrologic regime of surface waters by changing the way water cycles through a drainage basin. In a natural setting, precipitation is intercepted or delayed by the forest canopy and ground cover. Vegetation, depressions on the land, and soils provide extensive storage capacity for precipitation. Water exceeding this capacity travels via shallow subsurface flow and groundwater and eventually discharges gradually to surface water bodies. In a forested, undisturbed watershed, direct surface runoff occurs rarely or not at all because precipitation intensities do not exceed soil infiltration rates. Figures 4-1 and 4-2 illustrates this shift in hydrologic regime.

During the initial phases of urbanization, clearing of native vegetation reduces or eliminates interception storage and the water reservoir in soils. Loss of vegetation and "duff" (mostly composting vegetative material) from the understory takes away another storage reservoir. Site grading eliminates natural depressions. Impervious surfaces, of course, stop any infiltration and produce surface runoff. Even when surfaces remain pervious, building often removes, erodes, or compacts topsoil. The compacted, exposed soil retards infiltration and offers much less storage capacity. Development typically replaces natural drainage systems with hydraulically efficient pipe or ditch networks that shorten the travel time of runoff to the receiving water (Hirsch et al., 1990).

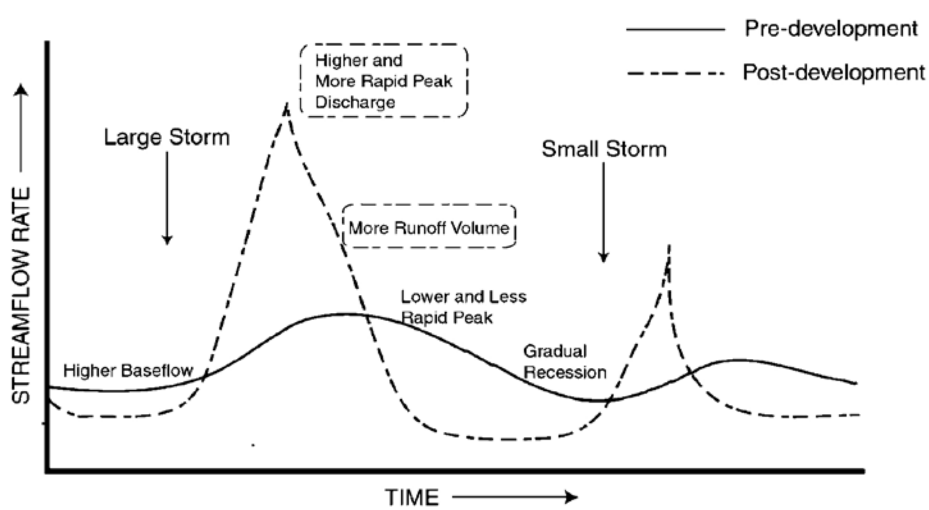
The many changes brought on by urbanization tend to alter streamflow patterns in characteristic ways.

Figure 4-6 illustrates typical hydrographs (flow rate versus time) for a stream before and after watershed urbanization. The hydrograph emphasizes the higher peak flow rate of urbanized basins compared to natural landscape conditions. The area under the hydrograph curves represents the total runoff volume, which is significantly greater for the urbanized condition. In addition, there is typically less "lag time" between rainfall and runoff when more impervious surfaces exist. The construction of an engineered stormwater drainage network also invariably increases the drainage density of urbanizing basins (Graf, 1977). Typically, these engineered conveyance systems are designed to efficiently remove water from the natural drainage network and so reduce the time necessary for overland flow to reach stream channels. The net effect of these urban watershed changes is that a higher proportion of rainfall is translated into runoff, which occurs more rapidly, and the resultant flood flows are therefore higher and much more "flashy" than natural catchments (Hollis, 1975).

In general, the hydrologic changes associated with urbanization can be traced primarily to the loss of natural land cover (vegetation and soil) and the increase in impervious surfaces in the watershed (Dunne and Leopold, 1978). The impact of urbanization and impervious surfaces on watershed hydrology has been studied for many decades. Wilson (1967) studied the impact of urbanization on flooding in Jackson, Mississippi. Early research by Leopold (1968) reported that a two- to five-fold increase in peak streamflow was common in urbanizing basins, although some streams showed an even greater rise, especially in arid areas. Seaburn (1969) studied the effects of urbanization on stormwater runoff on Long Island, New York, finding similar results. Hammer (1973) also found that peak streamflows increased with greater watershed urbanization. A decline in groundwater recharge is also common in urbanizing watersheds, due to greater impervious areas and less infiltration (Foster et al., 1994). Bharuri et al., (1997) also quantified the changes in streamflow and related decreases in groundwater recharge associated with watershed urbanization in the Midwest.

Hollis (1975) studied the impact of urbanization on flood recurrence interval. This research found that, in general, floods with a return period of one year or longer are not affected by a watershed impervious level of approximately 5 percent. In addition, small flooding events and peak streamflows may be increased by up to 10 times that found under natural conditions. Hollis (1975) found that under typical (~30 percent imperviousness) urbanized conditions, 100-year floods can be doubled in magnitude due to the greater runoff volume. Finally, the hydrologic effect of urbanization tends to decline, in relative terms, as flood recurrence intervals increase (Hollis, 1975). The findings of these studies indicate that it is not uncommon for a flood event with a 10-year recurrence interval to shift to a more frequent 2-year interval. Hollis (1975) also found that the discharge rates of small, frequent floods tend to increase by a greater percentage of pre-development rates than those of large, infrequent floods.

Figure 4-6: Changes in Stream Hydrology as a Result of Urbanization



Source: Schueler, 1994

In addition, the frequency of bankfull flows can be significantly increased in urbanizing stream basins. In western Washington State, a computer model capable of continuous simulation was used to study the hydrology of two similar watersheds (Booth, 1991). It compared a fully forested basin with a developed (approximately 40 percent impervious area) basin. The model predicted that the pre-development discharge that occurs only once in five years would occur in 39 of 40 years after urbanization. These alterations in hydrologic characteristics can result in a significant change in the disturbance regime of a typical stream ecosystem (Booth, 1991).

In a study in the Toronto area of Ontario, Canada (Snodgrass et al., 1998), the bankfull streamflow recurrence period was 1.5 years under natural conditions. Storms that result in bankfull flows were generally found to be in equilibrium with the natural resisting forces (e.g., stream bank vegetation) that tend to stabilize the stream channel. As watersheds urbanized, the streamflows that were bankfull flows occurred more frequently, up to about every 0.4 years in Toronto (Snodgrass et al., 1998).

A study in the upper Accotink Creek watershed in northern Virginia related the increase in impervious surface area from development to changes in streamflow over the period 1949 to 1994 (Jennings and Jarnagin, 2002). Over this period, the percent TIA increased from 3 percent to 33 percent. Over the same period, streamflow discharge response to precipitation events increased significantly, as did the frequency of peak events (Jennings and Jarnagin, 2002).

Other studies have shown similar results. In a stream study in Washington State, the flow rate that had been reached only once in 10 years on average before development, increased in frequency to about every two years after urbanization (Scott, 1982). In a similar study in Korea, the peak discharge of runoff increased and the mean lag time of the study stream decreased due to urbanization over a period of two decades (Kang et al., 1998).

Another important characteristic of highly impervious, urbanized watersheds is the production of runoff during even relatively small storm events. Under natural conditions, small precipitation events generally produce little, if any, runoff. This is due to the interception and evapo-transpiration of rainfall by native vegetation as well as to the absorption of rainfall by the upper soil horizon and rainfall held in natural depressions where it eventually infiltrates or evaporates. It has been estimated that natural depressional storage is typically at least 4 times that of impervious surfaces (Novotny and Chesters, 1981). A study in Australia found that the average peak discharge for urban streams was 3.5 times higher than the peak flow for rural streams (Neller, 1988).

Booth (1991) noted that in addition to high-flow peaks being amplified in urban stream hydrographs in the Puget Sound region, new peaks also appeared. These new peaks were the result of small storms, most of which produced no runoff under pre-development conditions but generated substantial flows under the urbanized condition. Therefore, it can be concluded that watershed development does more than just magnify peak flows and flooding events; it also creates entirely new high-flow events due to runoff from impervious surfaces.

Yet another characteristic of urban streams is the more rapid recession of stormflow peaks (see Figure 4-6). In addition, the baseflow conditions in urban streams are typically lower in urbanized watersheds. This has been observed for wet season baseflows in the Puget Sound region (Konrad and Booth, 2002) and in the Chesapeake Bay region (Klein, 1979). In arid regions, there may also be a noticeable decrease in dry season baseflow due to watershed development (Harris and Rantz, 1964). A study in Long Island, New York revealed the extent of seasonal hydrologic shifts in urban streams. In several undeveloped watersheds, stream baseflow constituted up to 95 percent of annual discharge. That proportion dropped to 20 percent after development (Simmons and Richard, 1982).

Rose and Peters (2001) examined streamflow characteristics that changed during the period from 1958 to 1996 in a highly urbanized watershed (Peachtree Creek), compared to less urbanized watersheds and non-urbanized watersheds, in the vicinity of Atlanta, Georgia. Data was obtained from seven U.S. Geological Survey (USGS) stream gages, 17 National Weather Service rain gages, and five USGS monitoring wells. The fraction of the rainfall occurring as runoff in the urban watershed was not significantly greater than in the less urbanized watersheds, but this ratio did decrease from the higher elevation and higher relief watersheds to the lower elevation and lower relief watersheds. For the 25 largest stormflows, the peak flows for the urban creek were 30 to 100 percent greater than the peak flows in the streams located in the less developed areas. In the urban stream, the streamflow also decreased more rapidly after storms than in the other streams. The low flow in the urban creek was 25 to 35 percent lower than in the less developed streams, likely caused by decreased infiltration due to the more efficient routing of stormwater and the paving of groundwater recharge areas.

In an extensive stream research project in Wisconsin, the observed decrease in stream baseflow was found to be strongly correlated with watershed imperviousness (Wang et al., 2001). Similarly, an urban stream study in Vancouver, British Columbia, Canada, monitored 11 urbanizing small-stream watersheds. Baseflow and groundwater recharge were consistently lower in watersheds with more than 40 percent impervious cover (Finkebine et al., 2000). Both of these studies found linkages between these shifts in hydrologic regime and both habitat degradation and the decline in biological integrity in the urbanizing streams.

Sheeder and others (2002) investigated the hydrograph responses to dual rural and urban land uses in three small watersheds. Two important conclusions were deduced from this investigation. First, in all cases, the researchers found two distinct peaks in stream discharge, each representing different contributing areas to direct discharge with greatly differing curve numbers and lags, representative of urban and rural source regions. Second, the direct discharge represented only a small fraction of the total drainage area, with the urban peak becoming increasingly important in relation to the rural peak as urbanization increases and the magnitude of the rain event decreases.

Nagasaka and Nakamura (1999) examined the influences of land-use changes on the hydrologic response and the riparian environment in a northern Japanese area. Temporal changes in a hydrological system and riparian ecosystem were examined with reference to land-use conversion in order to clarify the linkages between the two. The results indicated that the hydrological system had been altered since the 1970s, with increasing flood peaks of 1.5 to 2.5 times, and the time of peak flow appearances shortening by seven hours. The ecological systems were closely related to and distinctly altered by the changes that had occurred in the local land use. A similar study in southern California found comparable results (White and Greer, 2002).

Adjacent to water bodies, floodplain encroachment eliminates another storage zone needed to diminish high flows. When the channel cannot contain the greater flow, flooding results. Clearing riparian vegetation removes the wood supply that helps slow down the flow and, in many cases, prevent bed and bank erosion. Clearing also eliminates shade, refuge, and food supply. Urban residents and high streamflows remove remaining wood, further decreasing the stream's opportunity to dissipate energy without flooding or damaging the channel (Dunne and Leopold, 1978). In addition, any channel modifications (e.g., streambank armoring, levee construction, or diking) that inhibit stream-floodplain interactions can have serious consequences for downstream flooding.

Biological and Ecological Effects of Urban Hydrologic Change

As discussed above, the hydrologic impacts of watershed urbanization include the following:

- Greater runoff volume from impervious surfaces;
- Higher flood recurrence frequency;
- Less lag time between rainfall, runoff, and streamflow response;
- Higher peak streamflow for a given size storm event;
- More bankfull or higher streamflows – flashier flows;
- Longer duration of high streamflows during storm events;
- More rapid recession from peak flows;
- Lower wet and dry season baseflow levels;
- Less groundwater recharge; and
- Greater wetland water level fluctuation.

All of these characteristics represent alterations in the natural hydrologic regime to which aquatic biota have adapted over the long term. These are significant hydrologic changes that can negatively impact aquatic biota directly or indirectly. Direct impacts include washout of organisms from their preferred habitat and the physiological stress of swimming in higher flows. Indirect impacts are centered on the degradation of in-stream habitat that occurs as a result of the higher urban streamflows. These higher flows result in changes in channel geomorphology and physical habitat (to be discussed in detail in the next section), including stream bank erosion, stream channel instability, elevated levels of turbidity and fine sediment, channel widening or incision, stream bed scour, and the washout of in-stream structural elements (e.g., large woody debris or LWD).

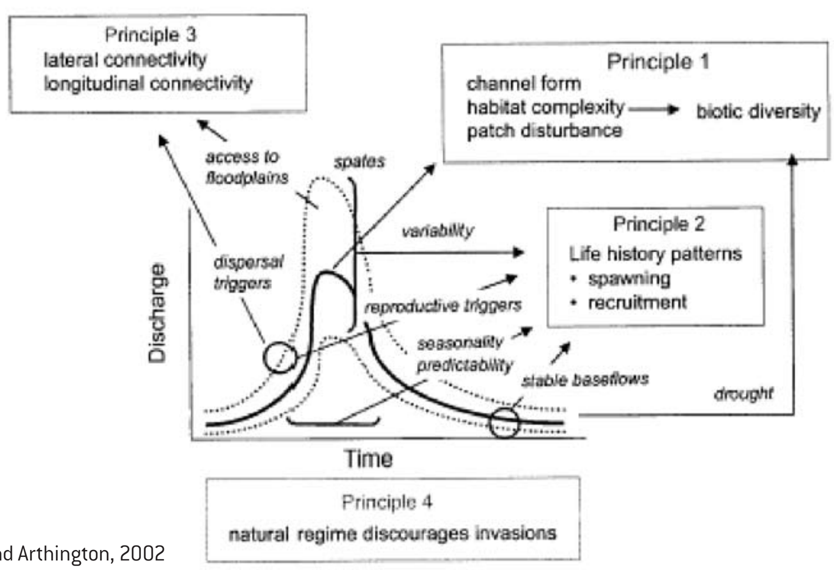
An extensive study comparing an urban (Kelsey Creek) and a non-urban (Big Bear Creek) stream in the Puget Sound region found that hydrologic changes from urbanization were the principal reasons that the urban stream failed to match its non-urban counterpart in diversity and size of salmonid fish populations and other biological indices (Pederson, 1981; Richey et al., 1981; Perkins, 1982; Richey, 1982; Scott et al., 1982). The study found that Kelsey Creek had significantly higher stormflows and flood flows, as well as lower baseflows, than Bear Creek. This shift in hydrologic regime resulted in extensive habitat degradation and stream channel alteration from the natural condition.

Another study in the Puget Sound region looked at the streamflow records of six small lowland streams over a 40-year period. Four of the study streams exhibited a significant increase in urbanization and two remained relatively undeveloped over the study period. Each of the urbanized basins experienced a significant increase in flood frequency, while the undeveloped basins showed no discernable shift in flood frequency. Salmon spawning-count data for the developed basins showed a systematic decline in salmon abundance, while the undeveloped basins showed no evidence of decline. The data implies a link between salmon population decline and either increased flood frequency or an associated degradation in habitat (Moscrip and Montgomery, 1997).

The Puget Sound Lowland Stream Research Project (May et al., 1997), one of the most comprehensive studies of the cumulative impacts of urbanization, also found that the shift in hydrologic regime in urbanizing small-stream watersheds was the primary cause of degraded habitat conditions, reduced stream biological integrity, and declining salmon diversity. In the Pacific Northwest, the importance of hydrologic alteration and its effects on stream habitats and the salmonid resource is widely recognized. A significant share of the urban runoff management effort goes into controlling water quantity to attempt to retain pre-development hydrologic patterns. With respect to resource protection, in most other urbanized areas, more attention is generally paid to quality control than to controlling quantity to maintain stream channel integrity. Yet, the same hydrologic modification problems have been noted elsewhere (Wilson, 1967; Seaburn, 1969; Hammer, 1972; Klein, 1979).

Finally, a comprehensive literature review conducted by Bunn and Arthington (2002) identifies the key principles and ecological consequences of altered flow regimes resulting from human modification of the watershed. These principles establish the linkages between flow regime and aquatic biodiversity as indicated in Figure 4-7. Their first principle is that flow is a major determinant of physical habitat in streams, which in turn determines the biotic composition of stream communities. Under this principle, channel geomorphic form, habitat structure, and complexity are determined by prevailing flow conditions. Urban examples of this have been discussed above, including the impact of flashy urban flows on benthic macroinvertebrates and native fish. The biotic communities of streams are largely determined by their natural flow regimes. This is true for aquatic insects and other macroinvertebrates (Resh

Figure 4-7: Aquatic Biodiversity and Natural Flow Regimes



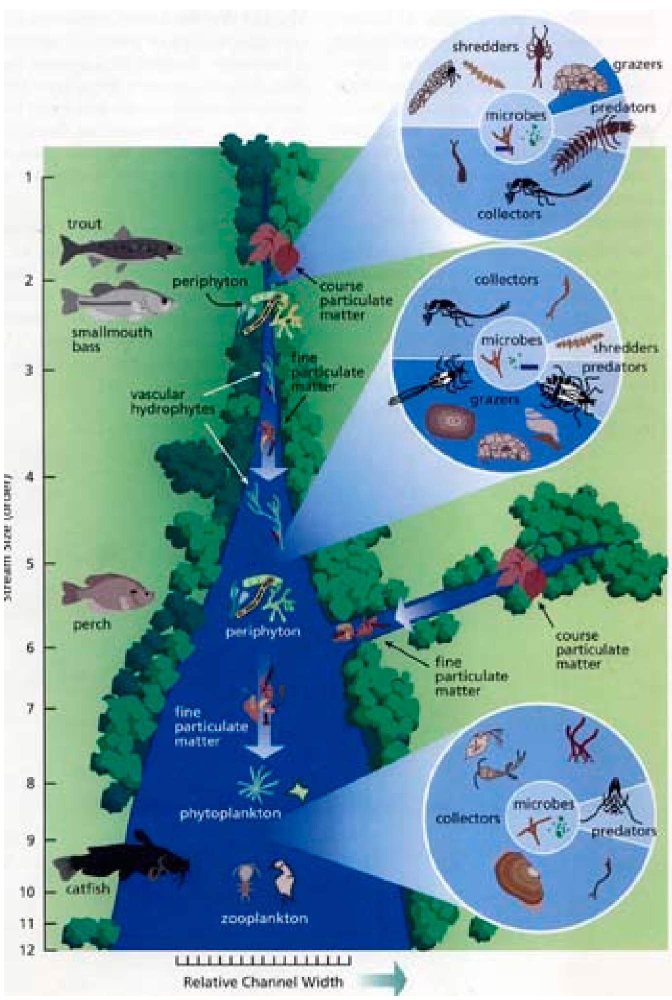
Source: Bunn and Arthington, 2002

et al., 1988) as well as fish (Poff and Ward, 1989; Poff and Allen, 1995; Poff et al., 1997).

The second principle is that aquatic species have evolved life history strategies primarily in direct response to the natural flow regime (Bunn and Arthington, 2002). For example, the timing and spatial distribution of salmon migration and spawning in the Pacific Northwest is largely determined by the natural flow regimes in each watershed (Groot and Margolis, 1991).

The third principle states that the maintenance of natural patterns of longitudinal and lateral connectivity is essential to the long-term viability of many populations of aquatic biota in flowing waters (Bunn and Arthington, 2002). Lateral connectivity refers to maintaining a connection between the active stream channel and the floodplain-riparian zone (Ward et al., 1999). This connection is often severely disrupted or lost altogether in urban streams where channelization and stream bank armoring are common. Longitudinal connectivity is disrupted by fragmentation of the riparian corridor by road or utility crossings (discussed in a later section) and the construction of in-stream migration barriers. The construction of dams and diversion structures, as well as road-crossing culverts that block fish passage, can significantly influence the viability of stream fish populations. In-stream barriers can block adult migration upstream to spawn, restrict juvenile fish access to rearing or refugia habitat, and disrupt the flow of large woody debris (LWD) and organic matter (OM) within the stream ecosystem. The river continuum concept (Vannote et al., 1980) illustrates the importance of connectivity within a stream ecosystem (Figure 4-8).

Figure 4-8: River Continuum Concept



Source: Modified from Van Note et al., 1980

The fourth and final principle states that the survival of invasive, exotic, and introduced (non-native) species is facilitated by altered flow regimes (Bunn and Arthington, 2002). The most successful exotic and invasive fish are often those that are either habitat generalists or adaptable to changing conditions (Moyle, 1986). Both these strategies are favorable to survival in urbanized hydrologic regimes. In addition, the long-term persistence of invasive fish is much more likely in aquatic systems that are permanently altered by human activity, as is the case for urbanized watersheds (Moyle and Light, 1996).

Urban Freshwater Wetland Hydrology

Wetlands provide many ecological functions for the watershed in which they are located. These functions include hydrologic, ecological, and water-quality components. Wetlands provide water storage features dispersed throughout the watershed landscape. Riparian wetlands provide natural flood storage volume. Most wetlands also provide critical storage capacity during periods of precipitation that provides for stream and groundwater recharge during dry periods. Wetlands also provide key habitat features for a variety of wildlife species.

The King County Urban Wetland Research Project studied the impacts of urbanization on freshwater wetlands in the Puget Sound lowland eco-region (Azous and Horner, 2003). Water level gages were used to determine wetland water level fluctuation (WLF). WLF is defined as the difference between base water level (BL) prior to a storm event and the crest or maximum water level (CL) for the event (WLF = CL – BL). This research found that WLF depends on a variety of watershed and wetland characteristics, but typically exceeded the natural range when basin imperviousness reached 10 percent TIA (Taylor, 1993; Azous and Horner, 2003). Similar results were found in freshwater wetlands in New Jersey (Ehrenfeld et al., 2003) and in tidal wetlands around the country (Thom et al., 2001). In a study in Saint Paul, Minnesota, Brown (1988) found that stormwater runoff quantity was related to both the amount of impervious surface area and the wetland-lake area in a basin.

In the Puget Sound urban wetland study, the WLF caused by watershed urbanization was not found to be consistently related to plant species richness but turned out to be an important factor in certain habitat types nonetheless, most notably in emergent wetlands. The frequency and duration of freshwater wetland flooding events was related to plant richness in all Puget Sound wetlands (Azous and Horner, 2003). The highest species richness at all water depths was found in wetlands with an average of less than three flooding events per month. Wetlands with a cumulative duration of flooding events lower than three days per month also had the highest species richness (Azous and Horner, 2003). While frequency affected plant richness at all water depths, duration particularly compounded the impact of frequency on vegetation found in water over two feet deep. When frequency and duration were analyzed together, it was found that the highest richness was found in wetlands with both an average of less than three events per month and a cumulative duration of flooding that was shorter than six days per month. These two factors were found to be more important than water depth in predicting plant richness (Azous and Horner, 2003).

In the Puget Sound lowland eco-region, watershed urbanization was found to have a negative impact on both native lentic and terrestrial-breeding amphibian richness. Wetlands with increasing urbanization in their contributing watersheds were significantly more likely to have lower amphibian richness than wetlands in less urbanized or natural watersheds (Azous and Horner, 2003). This relationship was linked to increased runoff into urban wetlands as well as a resultant increased WLF. When average WLF exceeded 20 cm, the number of native amphibian species declined significantly (Azous and Horner, 2003). It is thought that the greater WLF may have a disproportionate negative impact on amphibian breeding habitat and/or higher egg-embryo mortality due to desiccation of egg masses (Azous and Horner, 2003). Urbanized land-use activity in areas immediately adjacent to wetlands (within buffer zones) also decreased native amphibian richness (Azous and Horner, 2003). In general, wetlands adjacent to larger areas of forest are more likely to have richer populations of native amphibians.

Wetland WLF and flooding can also affect the richness of bird species. Increased flooding events may inundate nesting sites and disperse pollutants that bioaccumulate in birds through the aquatic food chain (Azous and Horner, 2003). Increased runoff and high WLF can alter cover, nesting habitat, and the distribution of birds' food sources. It was not possible, however, to establish that changes in population are directly related to land use since it is difficult to control for all habitat factors besides urbanization. In general, average bird species richness was inversely related to the level of urbanization (Azous and Horner, 2003).

The findings of the Puget Sound lowland eco-region urban wetland study consistently indicated that placing impervious surface on some 10 percent of a watershed creates significantly negative hydrologic, habitat, and ecological responses (Azous and Horner, 2003). To complicate the picture, development located immediately adjacent to the wetland (wetland buffer area and surrounding development), rather than away from it, can also have a significant influence on hydrologic conditions, habitat quality, and water quality (Azous and Horner, 2003).

Physical Impacts

Geomorphic Changes

Urbanization and the resultant hydrologic changes outlined above can cause significant alterations of natural stream morphological characteristics. The direct and indirect impacts of urbanization can affect longitudinal stream channel characteristics such as sinuosity and gradient. In addition, lateral characteristics such as stream channel bankfull width (BFW) and bankfull depth (BFD) can be altered as the stream expands to accommodate the higher runoff-driven flows brought on by watershed urbanization. Figure 4-9 illustrates the process of channel enlargement in urbanizing streams. Neller (1989) and Booth and Henshaw (2001) both reported that stream channels in urbanized watersheds had cross-sectional areas that were significantly larger than would be predicted based on catchment area and discharge alone.

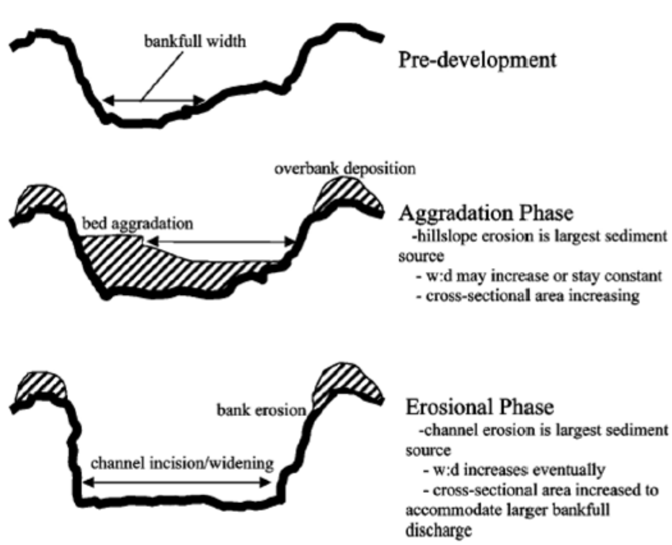
Channel enlargement can be a gradual process that follows the pace of urbanization, or it can frequently occur abruptly in response to particular storms (Hammer, 1972; Leopold, 1973; Booth, 1989; Booth and Henshaw 2001). Even in cases where the stream has been stable for many years, abrupt and sometimes massive changes in channel dimensions can occur in a single large storm once urbanization progresses to some critical level. In addition to causing accelerated channel enlargement, the higher and more frequent bankfull flows characteristic of urbanizing streams can also cause stream bank erosion, floodplain degradation, and a loss of channel sinuosity (Arnold et al., 1982).

During the construction phase of development, surface erosion of exposed areas can increase the supply of sediment available to runoff. This deposition of excess sediment can result in streambed aggradation and overbank deposition in floodplain areas. After construction is complete in a sub-basin, the external supply of sediment is reduced, but bankfull flows continue to increase as runoff from impervious surfaces increases. This can lead to increased stream bank erosion and channel enlargement as the stream tries to accommodate the increased streamflows (Paul and Meyer, 2001).

Channel enlargement tends to occur more often in urban streams that have some grade-control structures, such as in-stream LWD or road culverts. In these cases, the stream will generally erode the banks in order to widen the cross-sectional area to carry the higher urbanized flows. Culverts and other artificial grade-control structures can often cause downstream scour or upstream sediment deposition if not properly installed or maintained. Culverts in urban streams can often become migration barriers for aquatic biota such as anadromous fish or amphibians. In addition, if not properly sized for urban streamflows, culverts can cause significant localized flooding.

It has been hypothesized that urban streams will eventually adjust to their post-development hydrologic

Figure 4-9: Changes in Stream-Channel Geomorphology Due to Urbanization



Source: Neller, 1989

regime and sediment supply. There is evidence that this is the case in some regions, such as Vancouver, British Columbia, Canada (Finkebine et al., 2000) and in the Puget Sound region (Booth and Henshaw, 2001) where some urban streams seem to have stabilized several decades after build-out was completed.

In other situations, rapid channel down-cutting, known as incision, can be especially dramatic in urbanizing streams, particularly in regions with unconsolidated soils or where in-stream (e.g., LWD) structure is lost (Shields et al., 1994). In the Pacific Northwest, incision can result when increased flow and loss of LWD that dissipates energy occur in relatively steep channels with easily erodible substrate (Booth, 1991). While all channel damage is ecologically detrimental, incision is especially problematic because it removes virtually all habitat and supplies great quantities of sediment that do further damage downstream (Booth and Henshaw, 2001).

Land-use encroachment into floodplain areas and flood-control measures such as dikes and levees can also simplify and straighten a stream channel. This can exacerbate downstream channel alterations (Graf, 1975). In addition to channel modifications carried out during urban development, many streams have residual channelization impacts from past agricultural activities. Stream bank armoring or "rip-rapping" used to mitigate stream bank erosion can actually worsen downstream flooding and stream bank erosion problems. Storm event flows are unable to spread out onto the floodplain, and the increased velocities are transferred downstream along with the elevated sediment loads. There can also be a direct loss of channel migration zone (CMZ) as well as floodplain disconnection, as stream banks are armoring and development encroaches. Trimble (1997) demonstrated that channel enlargement due to the increase in watershed urbanization-driven flows caused extensive stream bank erosion, which accounted for 66 percent of the sediment transported downstream in an urban stream in San Diego, California.

Research in several locations suggests that flows larger than a two- to five-year frequency discharge can be sufficient to create large-scale channel disruption (Carling, 1988; Sidle, 1988; Booth, 1990). More than anything else, the greatly increased incidence of these flows explains the ecological vulnerability of urban streams. In addition to stream bank erosion and streambed scour or incision, higher urban streamflows can physically destroy or wash out in-stream structural elements, such as LWD. This can have a negative feedback effect on the stream channel. As higher flows wash out more and more LWD, the channel becomes even more unstable and more susceptible to further geomorphic degradation. Under these conditions, stream channels can actually "unravel" as the combined effects of channel incision, enlargement, and erosion continue to impact the stream system (Horner et al., 1997).

Two similar studies, one in Maine (Morse, 2001) and one in the Puget Sound region (May et al., 1997), demonstrated that stream bank erosion was related to the level of watershed imperviousness and linked directly to the shift in hydrologic regime. This is not to say that stream bank erosion and other geomorphic changes are only driven by urbanization. Booth (1991) and Bledsoe (2001) both reported that geomorphic change in response to urbanization depends on other factors, such as underlying geology, vegetation structure, and soil type.

Stream bank erosion and streambed scour resulting from the urban streamflow regime described previously can result in the production of excessive quantities of fine sediment (Nelson and Booth, 2002). This increase in sediment yield can be especially acute during the construction phase of development when runoff from bare ground on construction sites can carry very high sediment loads. This change in sediment transport regime can change a stream from a meandering to a braided and aggrading channel form (Arnold et al., 1982).

The shift in sediment transport regime that typically accompanies urbanization can also result in excessive sedimentation of streambed habitats. Streambeds can also become embedded and ecologically non-functional with frequent deposits of fine sediment. In the Puget Sound region, it was found that the percentage of fine sediment in stream substrates used by salmon for spawning increased along with watershed urbanization (May et al., 1997).

When a watershed is finally fully built out, this situation can actually reverse as impervious surfaces become the dominant landscape feature. Under fully urbanized basin conditions, there is often a lack of sediment delivered to stream channels (Wolman, 1967; Booth, 1991; Pizzuto et al., 2000). Under highly urbanized conditions, streambeds can become armored and are, for the most part, ecologically non-functional (May et al., 1997).

As discussed above, the geomorphologic impacts of watershed urbanization include the following:

- Stream channel enlargement and instability;
- Stream bank erosion and fine sediment production;

- Stream channel incision or down-cutting;
- Streambed scour and fine sediment deposition;
- Increase in streambed embeddedness;
- Riparian buffer (lateral) encroachment;
- Riparian corridor (longitudinal) fragmentation;
- Channelization and floodplain encroachment;
- Stream bank armoring and loss of CMZ;
- Increased sediment yields, especially during construction;
- Washout of in-stream LWD;
- Simplification of the natural drainage network, including loss of headwater channels and wetlands and lower drainage density;
- Modification of natural in-stream pool-riffle structure; and
- Fish and amphibian migration barriers (e.g., culverts and dams).

Degradation of Riparian Integrity

Riparian vegetation or the streamside forest is an integral component of all stream ecosystems. This is especially true of forested regions like the Pacific Northwest. A wide, nearly continuous corridor of mature forest, off-channel wetlands, and complex floodplain areas characterizes the natural stream-riparian ecosystems of the Pacific Northwest (Naiman and Bilby, 1998). Native riparian forests of the region are typically dominated by a complex, multi-layered forest of mature conifers mixed with patches of alder where disturbance has occurred in the recent past (Gregory et al., 1991). The riparian forest also includes a complex, dense, and diverse understory and ground cover vegetation. In addition, the extensive upper soil layer of forest "duff" provides vital water retention and filtering capacity for the ecosystem. A typical natural riparian corridor in the Puget Sound lowlands also includes a floodplain area, a channel migration zone (CMZ), and numerous off-channel wetlands. Natural floodplains, an unconstrained CMZ, and complex riparian wetlands are critical components of a properly functioning aquatic ecosystem (Naiman and Bilby, 1998). Organic debris and vegetation from riparian forests also provide a majority of the organic carbon and nutrients that support the aquatic ecosystem food web in these small lowland streams. In short, the riparian community (vegetation and wildlife) directly influences the physical, chemical, and biological conditions of the aquatic ecosystem. Reciprocally, the aquatic ecosystem affects the structure and function of the riparian community.

In addition to the characteristics of the riparian forest described above, the most commonly recognized functions of the riparian corridor include the following:

- Providing canopy-cover shade necessary to maintain cool stream temperatures required by salmonids and other aquatic biota. Regulation of sunlight and microclimate for the stream-riparian ecosystem (Gregory et al., 1991).
- Providing organic debris, leaf litter, and other allochthonous inputs that are a critical component of many stream food webs, especially in headwater reaches (Gregory et al., 1991; Naiman et al., 2000; Rot et al., 2000).
- Stabilizing stream banks, minimizing stream bank erosion, and reducing the occurrence of landslides while still providing stream gravel recruitment (Naiman et al., 2000).
- Interacting with the stream channel in the floodplain and channel migration zone (CMZ). Retention of flood waters. Reduction of fine sediment input into the stream system through floodplain sediment retention and vegetative filtering (Naiman et al., 2000).
- Facilitating the exchange of groundwater and surface water in the riparian floodplain and stream hyporheic zone (Correll, et al., 2000).
- Filtering and vegetative uptake of nutrients and pollutants from groundwater and stormwater runoff (Fischer et al., 2000).
- Providing recruitment of large woody debris (LWD) into the stream channel. LWD is the primary in-stream structural element and functions as a hydraulic roughness element to moderate streamflows. LWD also serves a pool-forming function, providing critical salmonid rearing, flow refugia, and enhanced instream habitat diversity (Fetherston et al., 1995; Rot, 1995; Rot et al., 2000).
- Providing critical wildlife habitat including migration corridors, feeding and watering habitat, and refuge areas during upland disturbance events (Gregory et al., 1991; Fischer et al., 2000; Hennings and Edge, 2003). Providing primary habitat for aquatic habitat modifiers such as beaver and

associated with salmonid populations.

Based on the results of research in the Puget Sound region (May et al., 1997), the term *riparian integrity* was adopted to describe the conditions found in natural lowland stream-riparian ecosystems. These properly functioning conditions can serve as a template for evaluation and management of riparian areas. As used here, riparian integrity includes both structural and functional elements characteristic of the natural stream-riparian ecosystem. Land-use activities and development encroachment pressure can have a negative impact on native riparian forests and wetlands, which are intimately involved in stream ecosystem functioning. Riparian integrity includes the following components:

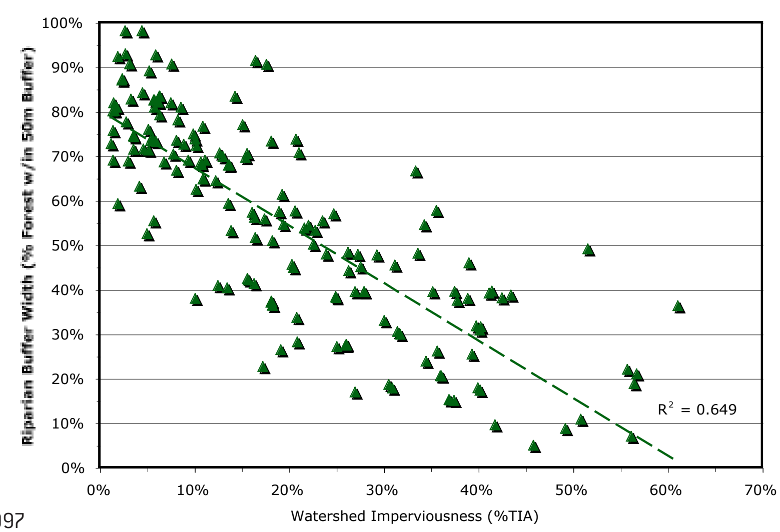
- Lateral riparian extent (so-called "buffer" width);
- Longitudinal riparian corridor connectivity (low fragmentation);
- Riparian quality (vegetation type, diversity, and maturity); and
- Floodplain and channel migration zone (CMZ) integrity.

In general, urban riparian buffers have not been consistently protected or well managed (Schueler, 1995; Wenger, 1999; Horner and May, 1999; Moglen, 2000; Lee et al., 2004). This is certainly true of the Puget Sound region (Figure 4-10). Several factors reduce the effectiveness of riparian buffers in urbanizing watersheds. The surrounding land use may overwhelm the buffer, and human encroachment continues to occur in spite of established buffer zones. Buffers that are established by regulation during the construction phase of development are rarely monitored by jurisdictional agencies. Over the long term, oversight and management of buffer areas is often taken on by property owners, who frequently are not familiar with the purpose or proper maintenance of the buffer (Booth, 1991; Schueler, 1995; Booth et al., 2002).

Ideally, the riparian corridor in a developing or developed watershed should mirror that found in the natural ecosystems of that region. Due to the cumulative impacts of past and present land use, this is often not the case (Figure 4-11). One example of this is the fragmentation of riparian corridors by roads, utility crossings, and other man-made breaks in the corridor continuity (Figure 4-12). Results from studies in the Pacific Northwest and other regions indicate that streams with a high level of riparian integrity have a greater potential for maintaining natural ecological conditions than streams with urbanized riparian corridors (May and Horner, 2000; Hession et al., 2000; Snyder et al., 2003). However, buffers can provide only a partial mitigation for urban impacts on the stream-riparian ecosystem. At some point in the development process, upland urbanization and the accompanying disturbance is likely to overwhelm the ability of buffers to mitigate for urban impacts.

There are certain problems associated with the loss of functional riparian floodplain corridors around streams

Figure 4-10: Relationship Between Riparian Buffer Width and Impervious Surface Area in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



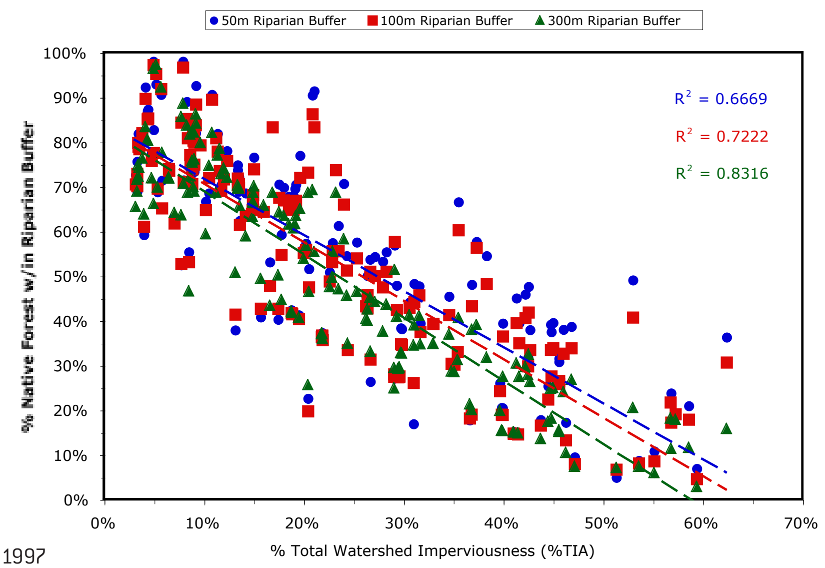
Source: May et al., 1997

in urbanizing watersheds. These include changes in food web dynamics, higher stream temperatures, loss of instream habitat complexity (LWD), invasive species, stream bank erosion and greater inputs of sediment, excessive nutrient inputs, inflows of anthropogenic pollutants, and loss of wildlife habitat.

Stream temperature is regulated mainly by the amount of shade provided by the riparian corridor. This is an important variable affecting many instream processes such as the saturation value for dissolved oxygen (DO) in the water, OM decomposition, fish egg and embryonic development, and invertebrate life history (Paul and Meyer, 2001). Removal of riparian vegetation, reduced groundwater recharge, and the "heat island" effect associated with urbanization all can affect water temperature of streams, lakes, rivers, wetlands, and nearshore marine areas.

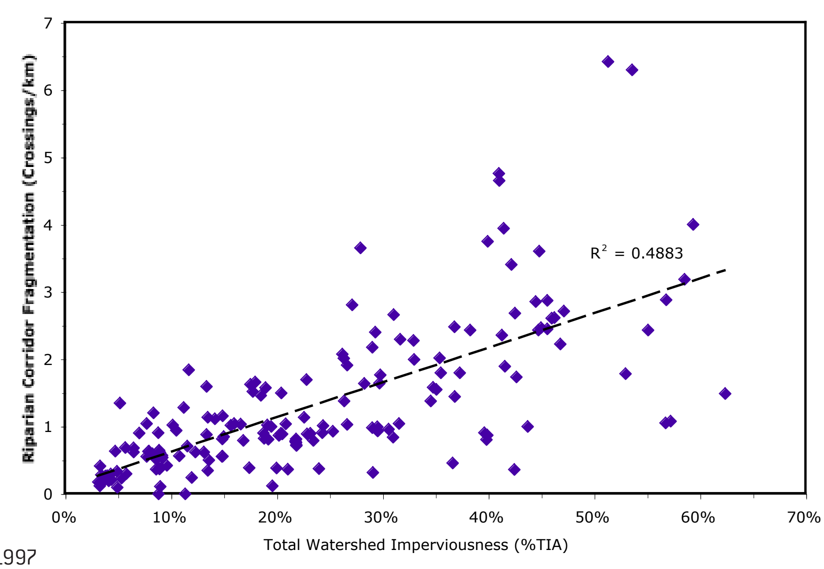
Invasive or exotic plants are another problem common to urban stream and wetland buffers. Human encroachment and landscaping activities can introduce

Figure 4-11: Relationship Between Riparian Quality and Impervious Surface Area in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

Figure 4-12: Relationship Between Riparian Corridor Fragmentation and Impervious Surface Area in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

exotic or invasive species into the riparian zone. These plants often out-compete native species, which can result in nuisance levels of growth.

Based on our current level of knowledge, the extent and configuration of urban riparian corridor buffers needed to protect the natural structure and function of the stream-riparian ecosystem cannot be described using a simple formula. Because of regional, watershed-scale, and site-level differences, as well as political issues, this is a fairly complex problem. The ecological and socio-economic value of the resource being protected should be considered when a riparian buffer or management zone is established. In addition, the local watershed, site, and riparian vegetation characteristics must be considered as well. The type and intensity of the surrounding land use should also be factored into the equation so that some measure of physical encroachment and water-quality risk is made. Finally, the riparian functions that need to be provided should be evaluated. Figure 4-13 illustrates how this might be done (Sedell et al., 1997).

Effects of Urbanization on Stream Habitat and Biota

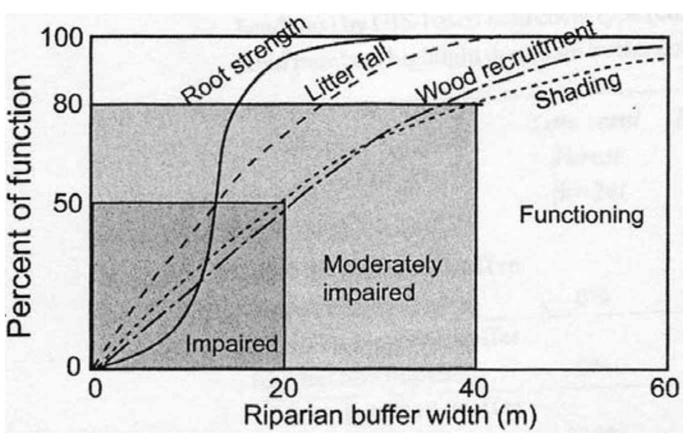
Degradation of aquatic habitat is one of the most significant ecological impacts of the changes that accompany watershed urbanization. The complex physical effects from elevated urban streamflows, stream channel alterations, and riparian encroachment can damage or destroy stream and wetland habitats. In addition to the indirect effects of habitat degradation or loss, aquatic biota can be directly affected by the cumulative impacts of urbanization.

Biological degradation is generally manifested more rapidly than physical degradation. Aquatic biota tend to respond immediately to widely fluctuating water temperatures, water quality, reduced OM inputs or other food sources, more frequent elevated streamflows, greater wetland water level fluctuations, or higher sediment loads. These stressors may prove to be fatal to some sensitive biota, impair the physiological functions of others, or encourage mobile organisms to migrate to a more habitable environment.

Ecological and biological effects of watershed urbanization include the following:

- Loss of instream complexity and habitat quality due to increase in bankfull flow frequency and duration.
- Reduced habitat due to channel modifications, and reduced baseflows causing crowding and increased competition for refuge and foraging habitat.
- Shifts in populations and communities of environmentally sensitive organisms to biota more tolerant of degraded conditions. Reduced biota abundance and biodiversity.
- Scouring and washout of biota and structural habitat elements from urban stream channels.
- Sediment deposits on gravel substrates where fish spawn and rear young and where algal and invertebrate food sources live. Reduced survival of egg and embryonic life stages.

Figure 4-13: Relationship Between Riparian Function and Buffer Width



Source: FEAMT, 1993

- Direct loss of habitat due to the replacement of natural stream channels and wetlands with engineered drainage channels and stormwater treatment ponds.
- Loss of ecologically functional pool-riffle habitat characteristics in stream channels. Loss of deep-water cover in rearing habitat and loss of spawning habitat.
- Aesthetic degradation and loss of recreational beneficial uses.
- Direct effects of suspended sediment on aquatic organisms, like abrasion of gills and other sensitive tissues, reduced light for photosynthesis, reduced visibility for catching food and avoiding predators, and transport of metallic, organic, oxygen-demanding, bacterial, and nutrient pollutants.
- Reduction in pool area and quality. Loss of refuge habitat for adult and juvenile fish.
- Loss of riparian vegetation, resulting in stream bank erosion, loss of shading and temperature regulation, reduced leaf-litter and OM input, loss of overhanging vegetation cover, and reduced LWD recruitment.
- Loss of LWD function, including hydraulic roughness, habitat formation, and refugia habitat.
- Increased summer temperatures because of lower baseflow and less water availability for heat absorption. Decline in DO from the lower oxygen solubility of warmer water.
- Less dilution of pollutants as a result of lower baseflows, which in turn results in higher concentrations and shallower flow that can interfere with fish migrations and localized movements.
- Increased inorganic and organic pollutant loads with potential toxicity impacts.
- Increased bacterial and pathogen pollution, which can result in an increase in disease in aquatic biota and humans.
- Elevated nutrient loading and resultant eutrophication of lake, wetland, and estuarine habitats. Reduced DO as a possible result of eutrophic conditions, which in turn reduces usable aquatic habitat.
- More barriers to fish migration, such as blocking culverts and diversion dams.
- Overall loss of habitat quality, complexity, and diversity due to channel and floodplain simplification or loss.

Numerous studies have documented the effect of watershed urbanization on the degradation of instream habitat and the decline of native biota. These include research from almost all parts of country and from developed countries around the world. The earliest research efforts to study the cumulative impacts of urbanization on small-stream habitat and stream biota were conducted in the Puget Sound region (Richey, 1982; Scott, 1982; Steward, 1983) and in the Chesapeake Bay region (Ragan and Dietermann, 1975; Ragan et al., 1977; Klein, 1979). These were followed by even more comprehensive studies in the same regions and in other parts of the country. This section describes the findings of this body of research (see Table 4-2 for a research summary).

As discussed earlier, one of the most common effects of watershed urbanization on instream habitat is the loss of habitat quality, diversity, and complexity. This is the so-called "simplification" of urban stream characteristics. In undisturbed, properly functioning stream systems, the natural (mainly hydrologically driven) disturbance regime maintains the stream in a state of dynamic equilibrium. This means that the stream ecosystem is stable, but not static. Changes occur on several spatial and temporal time scales (Figure 4-14).

These changes can be small and subtle, such as a riparian tree falling into a creek (LWD recruitment) and forming a new pool habitat unit as the result of the hydro-geomorphic interaction of the streamflow and the LWD. Changes can also be large and catastrophic, such as those occurring during major flooding events that can rearrange the entire channel form of a stream system. Natural streams tend to have a level of redundancy and complexity that allows them to be resilient in responding to disturbance. Streams may change over time as a result of natural habitat-forming processes (flooding, fire, LWD recruitment, sediment transport, OM and nutrient cycling, and others), but they continue to support a complex stream-riparian ecosystem and a diverse array of native biota.

As mentioned above, the first Puget Sound stream research project compared ecological and biological conditions in an urbanized stream (Kelsey Creek) and a relatively natural stream (Big Bear Creek). Urbanized Kelsey Creek was found to be highly constrained by the encroachment of urban development, with 35

Table 4-2: Summary of Research on Urban Stream Habitat, Water-Quality (WQ), and Biota					
Research Study	Habitat	WQ	Fish	Macro-invertebrates	Location
Ragan & Dietermann, 1975		x	x		MD
Klein, 1979	x	x	x		MD
Richey, 1982	x				WA
Pitt and Bozeman, 1982		x	x	x	CA
Steward, 1983			x		WA
Scott et al., 1986	x		x		WA
Jones and Clark, 1987		x		x	VA
Steedman, 1988				x	OT
Limburg & Schmidt, 1990		x	x		NY
Schueler & Galli, 1992			x		DC
Booth & Reinelt, 1993	x				WA
Lucchetti & Fuerstenberg, 1993			x		WA
Black & Veatch, 1994	x		x	x	MD
Weaver & Garman, 1994			x		VA
Lenat & Crawford, 1994	x	x	x	x	NC
Galli, 1994	x		x		DC
Jones et al., 1996	x	x		x	VA
Hicks & Larson, 1997	x				MA
Booth & Jackson, 1997	x				WA
Kemp & Spotila, 1997			x	x	PA
Maxted & Shaver, 1997	x			x	DE
May et al., 1997	x	x	x	x	WA
Wang et al., 1997	x		x		WI
Dali et al., 1998	x		x	x	MD
Harding et al., 1998	x		x	x	NC
Horner & May, 1999	x		x	x	WA
Kennen, 1999		x		x	NJ
MNCPPC, 2000	x		x	x	MD
Finkenbine et al., 2000	x				BC
Meyer & Couch, 2000	x		x	x	GA
Wang et al., 2000	x		x		WI
Horner et al., 2001	x		x	x	WA/TX/MD
Nerbonne & Vondracek, 2001	x		x	x	MN
Stranko & Rodney, 2001	x				MD
Wang et al., 2001	x		x		WI
Morse et al., 2002	x	x		x	ME

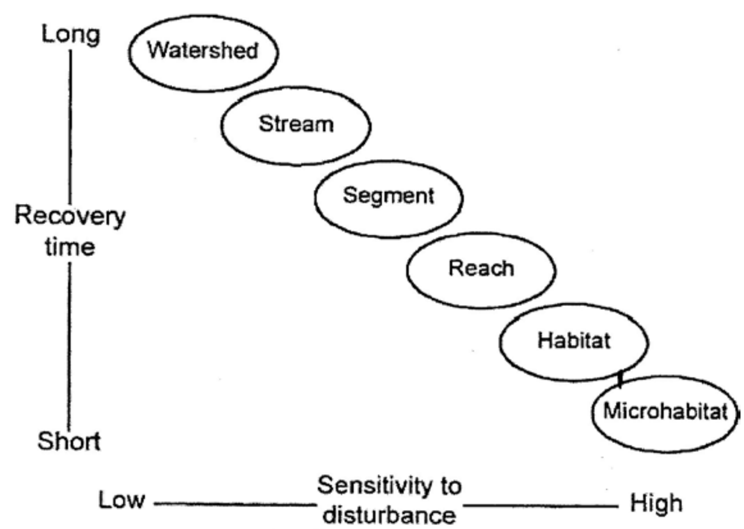
percent of the stream banks armored with "rip-rap" and the floodplain-riparian zone also highly modified. Bear Creek, on the other hand, had less than 10 percent stream bank armoring and a natural riparian corridor and CMZ. Road-crossing bridges and culverts were frequent on Kelsey Creek, but not on Bear Creek (Richey, 1982). LWD and other natural habitat complexity features common in Bear Creek were also lacking in Kelsey Creek (Steward, 1983).

In the Puget Sound comparison of urban and non-urban streams, Kelsey Creek, an urban stream, experienced twice the bed scour of its non-urban counterpart (Scott, 1982). As a consequence, sediment transport was three times as great in Kelsey Creek (Richey, 1982) and fines were twice as prevalent in its substrates (Scott, 1982). The invertebrate communities in different benthic locations produced 14 to 24 taxa in Bear Creek but only six to 14 in Kelsey Creek (Pedersen, 1981; Richey, 1982). Salmonid fish diversity also differed. Bear Creek had four salmonid species of different age-classes, whereas Kelsey Creek had only one non-anadromous species mainly represented by the 0- to 1-year age class (Scott, 1982; Steward, 1983). Although we cannot explicitly determine the relative roles of hydrology and habitat quality, much evidence shows that hydrologic alteration and the related sediment transport were most responsible for the biological effects (Richey, 1982).

Several studies in the Pacific Northwest examined various aspects of the influence of urban hydrology on salmon and salmon habitat. Data shows a significant decrease in young salmon survival in both large and small streams when events occur that are equal to or larger than the natural five-year frequency discharge. Since the frequency of events increases tremendously after urbanization, salmonids experience great difficulty in urban streams. These investigations also pointed out the relationship between urbanization level and biological integrity. The study rated channel stability along numerous stream reaches and related it to the proportion of the watershed's impervious areas. Stability was significantly higher where imperviousness was less than 10 percent (Booth and Reinelt, 1993). The study rated habitat quality along streams in two basins according to four standard measures. Marked habitat degradation occurred at 8 to 10 percent total impervious area (TIA). Population data on cutthroat trout and less tolerant coho salmon from streams draining nine catchments did not show a distinct threshold. They indicated, however, that population shifts are measurable with just a few percent of impervious area and become substantial beyond about 10 to 15 percent (Lucchetti and Fuerstenberg, 1993). Later studies in the same region confirmed this decline in salmonid abundance and diversity, as well as the degradation of salmon habitat at very low levels (5 to 10 percent TIA) of imperviousness in small urban streams (May, 1997; May et al., 1997; Horner and May, 1999).

More recent research projects in the Puget Sound region (May et al., 1997) and in Vancouver, British Columbia (Finkenbine et al., 2000) found that the degradation of instream and riparian habitat quality,

Figure 4-14: Stream Ecosystem Disturbance Regime



Source: Naiman, 1992

diversity, and complexity are common features of urban streams. There appears to be a linear decline in most measures of habitat quality in relationship to the level of watershed urbanization or imperviousness. Instream LWD, which is a critical habitat complexity element in streams in forested watersheds, tends to become scarce when %TIA approaches the 10 to 20 percent range (May et al., 1997; Horner et al., 1997; Finkenbine et al., 2000). Streambed quality also declines as urbanization increases (May et al., 1997; Horner et al., 1997; Finkenbine et al., 2000). This decline in benthic habitat is typically characterized by higher levels of fine-sediment deposition, substrata embeddedness, streambed coarsening, and frequent streambed scour events.

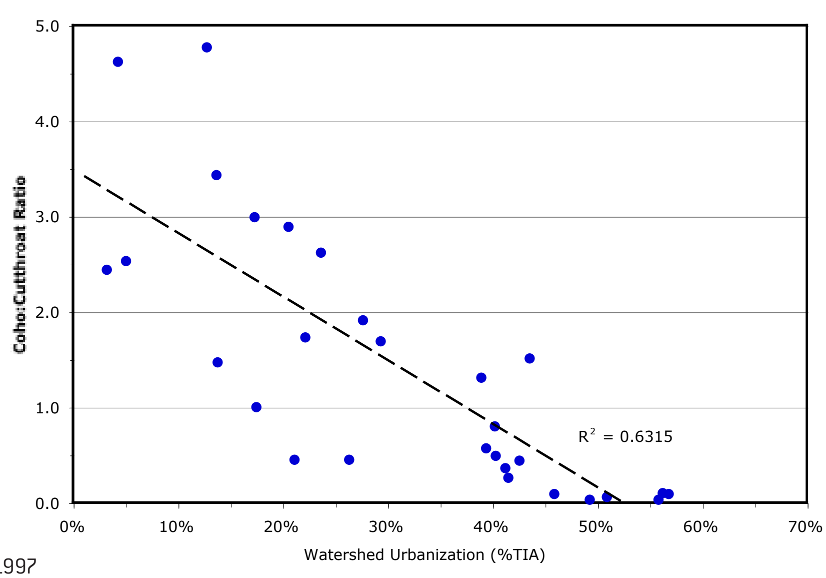
Similar to these studies in the Pacific Northwest, Morse (2003) observed that both instream habitat and water quality in small urbanizing streams in Maine declined in a linear fashion. Studies in Delaware (Maxted and Shaver, 1997), Wisconsin (Wang et al., 1997), and Minnesota (Nerbonne and Vondracek, 2001) confirm this trend. These findings have also been replicated in other countries, most notably in Australia (Davies et al., 2000) and New Zealand (Allibone et al., 2001).

This simplification of the stream channel and loss of instream habitat complexity results in a restructuring of the stream fish community in the urbanized creek. Urban impacts had a much greater impact on coho salmon (*Oncorhynchus kisutch*) than on cutthroat trout (*Oncorhynchus clarki*), which appear to be more tolerant of urban stream conditions (Scott et al., 1986). Pitt and Bissonnette (1984) and Lucchetti and Fuerstenberg (1993) also found similar results in other studies of streams in the Puget Sound lowland eco-region. Coho salmon, which normally out-compete cutthroat trout in natural streams, appear to be more sensitive to changes associated with urbanization and therefore decline in abundance as urban development increases (May, 1997; May et al., 1997; Horner et al., 1997; Horner and May, 1999). Figure 4-15 illustrates the shift in salmonid species found in urbanizing streams in the Puget Sound lowland eco-region.

Ragan and Dietermann (1975) attributed the loss of fish species diversity in urban streams in the Chesapeake eco-region of Maryland to the cumulative effects of urban development. A study in Ontario, Canada (Steedman, 1988) also found a shift in fish community structure due to the cumulative impacts of watershed land use and riparian corridor encroachment. Similar results were seen for fish community structures in New York (Limburg and Schmidt, 1990), Virginia (Weaver and Garman, 1994), Pennsylvania (Kemp and Spotila, 1997), North Carolina (Harding et al., 1998), and Georgia (Gillies et al., 2003).

A study in Mississippi found that instream habitat quality in urbanizing stream channels impacted by high-flow incision was significantly inferior to the quality of reference stream channels in undeveloped watersheds. In addition, the reference streams had greater mean

Figure 4-15: Relationship Between Stream Biological integrity, as Measured by Salmon Diversity, and Watershed Development, as Measured by Impervious Surface Area, in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

water depths, more channel complexity in the form of woody debris, and more deep pool refuge habitat than the impacted streams. Relative to the reference streams, fish assemblages in the incised stream channels were composed of smaller fish and fewer species (Shields et al., 1994).

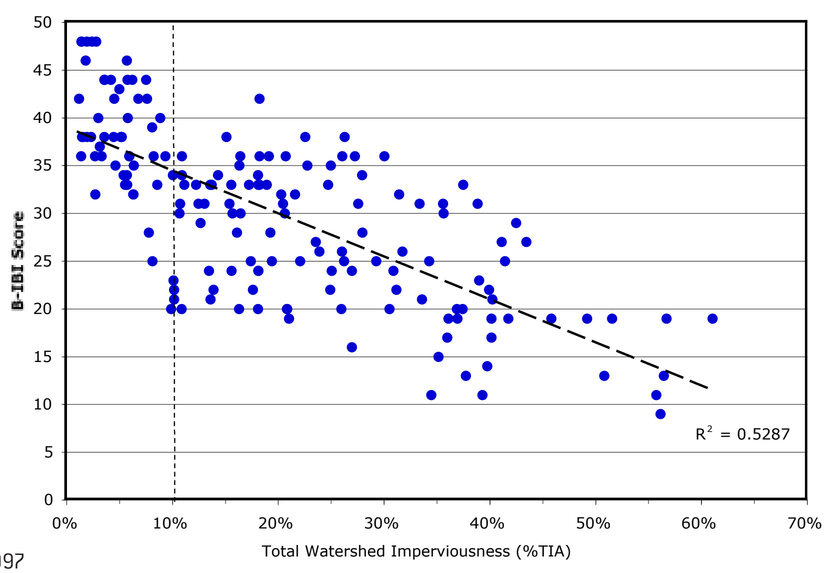
In several extensive studies of urbanizing streams in Wisconsin, a significant relationship was found between watershed land use and instream habitat as well as stream fish communities (Wang et al., 1997; Wang et al., 2000; Wang et al., 2001). In these studies, stream fish abundance and diversity both declined as watershed development increased above the 8 to 12 percent total impervious range. These studies also compared agricultural impacts to urban impacts, finding that urbanization was more severe and longer lasting. Habitat destruction and water-quality degradation were found to be the main contributing factors to the overall decline in stream ecosystem health. In addition, natural riparian vegetation (buffer) conditions had a significant influence on instream habitat conditions and appeared to at least partially mitigate some of the negative impacts of watershed urbanization (Wang et al., 2001).

A study in Washington, DC (Galli, 1991) investigated the local thermal impacts of urban runoff on stream ecosystems and reached the following conclusions:

- Air temperature was the strongest influence on stream water temperature.
- Average stream temperature increased linearly with stream sub-basin imperviousness.
- Some temperature criteria violations occurred just above 10 percent TIA and increased in severity and frequency with more imperviousness.
- All tested structural stormwater treatment facilities under best management practice (BMP) that had a surface discharge caused some violations of temperature criteria under both baseflow and storm runoff conditions.
- Based on the findings from a literature review, the investigators concluded that the thermal conditions produced by urban runoff and treatment facilities could cause succession from cold-water diatoms to warm-water filamentous green and blue-green algal species, as well as severe impacts on cold-water invertebrates and fish. A shift from cold-water community composition to warm-water organisms and exotic species is very possible in highly urbanized watersheds.

It should be noted that the life cycles of native fish can differ significantly even among closely related species. Attention must be paid to the life history specifics and habitat requirements of the various species of concern in the urban watershed being managed before any decisions are made on conservation, restoration, or mitigation of stormwater runoff impacts. Different fish carry out their migrations, reproduction, and rearing

Figure 4-16: Relationship Between Stream Biological integrity, as Measured by the Benthic-Macroinvertebrate Index of Biotic Integrity (BIBI), and Watershed Development, as Measured by Impervious Surface Area, in Urbanizing Watersheds in the Puget Sound Region of the Pacific Northwest



Source: May et al., 1997

at different times and have freshwater stages of various lengths. Management must ensure that all life stages (egg, embryonic, juvenile, and/or adult) have the habitat conditions needed at the right time and that no barriers to migration exist.

The Ohio Environmental Protection Agency (OEPA) has an extensive database relating watershed development and land use to fish abundance and diversity. This data suggests that there are multiple levels of fish response to increasing urbanization. At the rural level of development (under 5 percent urban land use), sensitive species begin to disappear from streams. In the 5 to 15 percent urban land-use range (suburban development), habitat degradation is common and fish continue to decline in abundance and diversity. In addition, aquatic invertebrates also decline significantly. Above 15 percent watershed urbanization, habitat degradation, toxicity effects from physio-chemical water pollution, and nutrient enrichment result in severe degradation of fish fauna (Yoder et al., 1999). There have been similar findings in studies in Alabama (Onorato et al., 2000) and North Carolina (Lenat and Crawford, 1994).

The cumulative effects of urbanization, including altered hydrologic and sediment transport regimes as well as channel modifications and degraded instream habitat, were also found to cause a shift in the aquatic insect communities of urban streams in the Puget Sound region (Pedersen and Perkins, 1988; May et al., 1997; Horner and May, 1999; Morley and Karr, 2002). This relationship between watershed urbanization, stormwater runoff pollution, and aquatic insect community taxonomic composition has also been observed in small stream studies in northern Virginia (Jones and Clark, 1987; Jones et al., 1994), Pennsylvania (Kemp and Spotila, 1997), New Jersey (Kennen, 1999), and Maine (Morse, 2002). These findings have also been replicated in other countries, most notably in Australia (Walsh et al., 2001) and New Zealand (Collier and Winterbourn, 2000).

Aquatic insects and other macroinvertebrates have been found to be useful indicators of environmental conditions in that they respond to changes in natural land cover and human land use (Black et al., 2004). Overall, there tends to be a decline in taxa richness or species diversity, a loss of sensitive species, and an increase in tolerant species (such as chironomids) due mainly to the cumulative impacts of watershed urbanization: altered hydrologic and sediment transport regimes, degradation of instream habitat quality and complexity, stream bed fine sediment deposition, poor water quality, and the loss of native riparian vegetation. In many cases, the myriad of aquatic insects and benthic macroinvertebrates sampled from streams or wetlands are combined into a set of indices to standardize comparisons between stream samples. Often the mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) are combined into an "EPT" index. In some cases, multi-metric indexes have been developed that include several measures of the characteristics of the stream macroinvertebrate community. The EPA Rapid Bioassessment Protocol (RBP) and the Benthic Index of Biotic Integrity (BIBI) are examples of this (Karr, 1998). Figure 4-16 illustrates the BIBI scores for urbanizing streams in the Puget Sound lowland eco-region.

Ecological Impacts of Urban Stormwater Runoff Quality

Background

In addition to the hydrologic and physical impacts of stormwater runoff generated by the urbanization process, there are water-quality impacts to aquatic ecosystems and biota that result from exposure to the pollutants found in urban runoff. Stormwater runoff from urbanized areas is generated from a number of sources including residential areas, commercial and industrial areas, roads, highways and bridges. Essentially, as discussed earlier, any surface that does not have the capability to store and infiltrate water will produce runoff during storm events. These are the previously discussed impervious surfaces. As the level of imperviousness increases in a watershed, more rainfall is converted to runoff.

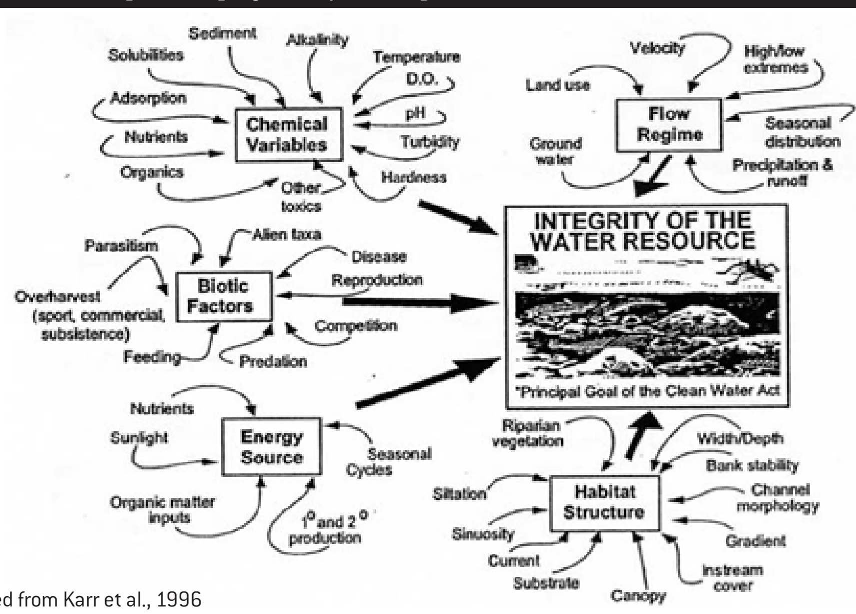
Impervious surfaces (roads, parking lots, rooftops, etc.) are the primary source areas for pollutants to collect within the built environment. Runoff from storm events then carries these pollutants into natural waters via the stormwater conveyance network. The land use (e.g., residential, commercial, and industrial) and human activities (e.g., industrial operations, residential lawn care, and vehicle maintenance) characteristic of a drainage basin largely determine the mixture and level of pollutants found in stormwater runoff (Weibel et al., 1964; Griffin et al., 1980; Makepeace et al., 1995; Pitt et al., 1995).

As was discussed in detail in the previous chapter, stormwater is a form of non-point source (NPS) pollution and typically contains a mixture of pollutants, including metals, petroleum hydrocarbons, and organic toxicants (i.e., pesticides, herbicides, and industrial chemicals). The National Urban Runoff Program (NURP) identified stormwater as a significant source of potentially toxic pollutants to receiving waters (EPA, 1983). Other studies have confirmed the NURP findings and improved the level of knowledge with regard to stormwater pollution impacts (Ragan and Dietermann, 1975; Pitt and Bozeman, 1982; Field and Pitt, 1990; Bannerman et al., 1993). Two of the most common stormwater pollutant components are petroleum hydrocarbon compounds and metals (e.g., zinc, copper, lead, chromium, etc.). Hydrocarbon sources include vehicle fuels and lubricants (Hoffman et al., 1984; Fram et al., 1987; Smith et al., 2000). Metals are also associated with vehicle maintenance, roads, and parking areas (Wilber and Hunter, 1977; Davies, 1986; Field and Pitt, 1990; Pitt et al., 1995). Pesticides, herbicides, and other organic pollutants are also commonly found in stormwater flowing from residential and agricultural areas (Pereira et al., 1996; USGS, 1997; Fan et al., 1998; Black et al., 2000; Foster et al., 2000; Hoffman et al., 2000). Studies in Puget Sound confirm these findings for our region (Hall and Anderson, 1986; May et al., 1997; USGS, 1997; Black et al., 2000). In many cases, even banned pesticides such as DDT or other organo-chlorine-based pesticides (e.g., chlordane and dieldrin) can be found in urban stream sediments. Toxic industrial compounds such as PCBs can also be present in urban runoff (Black et al., 2000). In general, the more intense the level of urbanization, the higher the pollutant loading, and the greater the diversity of land-use activities, the more diverse the mixture of pollutants found in stormwater runoff (Herricks, 1995; Makepeace et al., 1995; Pitt et al., 1995).

As discussed in the previous chapter, the transport and fate mechanisms of stormwater pollutants in receiving waters tend to be highly variable and site-specific. Pollutants are often transported from source areas (roads, parking lots, lawns, etc.) to receiving waters via roadside ditches, stormwater pipes, or by atmospheric deposition. In general, the concentration of pollutants found in stormwater runoff is much higher than that found in receiving waters, due mostly to dilution and removal mechanisms. In addition, most stormwater pollutants are typically found in particulate form, attached to fine sediment particles and organic matter (Pitt et al., 1995). This is especially true for nutrients, organics, and metals. In most cases, the particulate forms of toxic pollutants tend to be less "bio-available" (Herricks, 1995).

Because of the potential for accumulation of pollutants in sediment and the potential of sediments as sources of toxics, polluted sediments likely play an important role in many of the biological impacts associated with stormwater runoff. In general, most pol-

Figure 4-17: Stream Ecological Integrity Conceptual Diagram



Source: Adapted from Karr et al., 1996

lutants, especially metals, are found in particulate forms within the water column or sediments, and pollutant concentrations tend to be higher for smaller sediment particle sizes (DePinto et al., 1980).

As discussed earlier, physical variables such as flow regime and instream habitat are important to native biota, as are chemical factors like water or sediment quality (Figure 4-17). Human activities in urbanizing watersheds can lead to both physio-chemical pollution and biophysical alterations of stream habitats. The evaluation of cumulative ecological urban impacts can be problematic where both types of stressors occur. The relative importance of one stressor as compared to another is difficult to quantify, especially when antagonistic or synergistic effects are present. For example, effects of contaminants can also be masked by instream or riparian habitat degradation. All of these variables need to be quantified in order for a complete assessment of the impact of stormwater on human health, aquatic ecosystems, and instream biota to be developed (Horner et al., 1997).

Stormwater Toxicity in Freshwater

Current stormwater monitoring and impact assessment programs indicate that the most likely cause for degradation of biological integrity in receiving waters is a combination of physical habitat degradation, changes in the hydrologic regime, food web disruptions, and long-term exposure to anthropogenic contaminants (Pitt, 2002). However, chronic or acute exposure to potentially toxic contaminants may be especially problematic for benthic organisms such as macroinvertebrates and for organisms that have a benthic life stage (e.g., salmonids during their embryonic development stage). Acute toxicity of aquatic biota due to exposure to stormwater runoff in receiving waters is rare (Pitt, 2002).

Current research appears to indicate that even when stormwater toxicity is high, it is only for short periods of time during episodic storm events. It has been hypothesized that relatively short periods of exposure to toxic compounds at the levels normally found in stormwater are not sufficient to produce mortality in aquatic organisms. This is often based on the assumption that most of the toxic chemicals found in stormwater are found in particulate form and are not bioavailable. This school of thought holds that most of the toxicity problems observed in urban receiving waters are a result of illegal discharges or dumping and that the risk from stormwater and sediment-bound toxics is low. However, this view tends to ignore the cumulative impacts of frequent exposures of organisms in receiving waters to stormwater as well as the potential release of toxics from sediments due to changes in ambient water chemistry. In reality, urban stormwater runoff has been found to cause significant receiving water impacts on aquatic biota (Burton and Pitt, 2001).

Evaluation of stormwater or receiving water quality is a complex and expensive project. The type and quantity of stormwater constituents are highly variable, depending on land use and human activities in the source area of concern. There are also numerous confounding factors that influence how stormwater interacts with receiving waters. In addition, the relationship between observed biological effects on receiving water and possible causes (including stormwater-related toxicity) are especially difficult to identify, let alone quantify. Countless antagonistic and synergistic chemical relationships exist among the constituents in stormwater runoff and receiving waters. Physio-chemical transformations can render toxic substances harmless or create toxic mixtures from individually harmless compounds. Contaminants can also be associated with suspended sediment particles or mobilized from streambed sediments due to scour during high-flow events (Mancini and Plummer, 1986). It is likely that in most situations, multiple stressors and cumulative impacts play a significant role in the decline of biological integrity.

Many studies have shown the detrimental effects of stormwater runoff on receiving water biota. However, few studies have demonstrated a direct cause-and-effect relationship between stormwater and toxicity to aquatic biota. Beginning with the National Urban Runoff Program or NURP (EPA, 1983), numerous studies have focused on determining the chemical characteristics of stormwater. An update of the NURP stormwater data was conducted in 1999 (Smullen et al., 1999). There have also been several studies on the toxicological effects of stormwater on aquatic biota.

Pitt and Bozeman (1982) studied the impacts of urban runoff on stream water quality and biological conditions in Coyote Creek in the San Francisco Bay area. The results of this study indicated that water and sediment quality were significantly degraded by urban stormwater runoff (Pitt and Bozeman, 1982). There was also some evidence of bioaccumulation of urban pollutants in plants, fish, and macroinvertebrates resident to the system (Pitt and Bozeman, 1982).

Studies of urban streams in Bellevue, Washington examined the ecological and biological impacts of stormwater runoff (Perkins, 1982; Richey, 1982; Scott et al., 1982; Pitt and Bissonette, 1983). These studies documented the physio-chemical water quality and instream habitat degradation due to watershed development and stormwater runoff. Massive fish kills in Kelsey Creek were also observed during one of these studies. These fish kills were attributed to illegal dumping of toxic chemicals into local storm drains.

Medeiros and Coler (1984) used a combination of laboratory flow-through bioassay tests and field experiments to investigate the effects of urban stormwater runoff on fathead minnows and observed chronic effects of stormwater toxicity on growth rates in the test organisms.

Hall and Anderson (1988) studied the effects of urban land use on the chemical composition of stormwater and its toxicity to aquatic invertebrates in the Brunette River in British Columbia. This study found that land-use characteristics and the antecedent dry period between rainfall events had the greatest influence on stormwater quality and toxicity. Toxicity in this study followed the land-use sequence commercial>industrial>residential>open space (Hall and Anderson, 1988). This study also identified the "first flush" effect as being significant from a toxicity standpoint. The longer the dry build-up period between storms, the higher the pollutant load and the greater the toxicity of stormwater runoff (Hall and Anderson, 1988).

A study of stormwater toxicity in Birmingham, Alabama utilized toxicity screening as the primary detection method (Pitt et al., 1995). Of the stormwater source area samples collected, 9 percent were classified as extremely toxic, 32 percent were moderately toxic, and 59 percent showed no evidence of toxicity. Vehicle service and parking areas had the highest levels of pollutants and potential toxicants. Metals and organics were the most common toxicants found in stormwater samples.

A field study in Milwaukee, Wisconsin investigated the effects of stormwater on Lincoln Creek (Crunkilton et al., 1997). Streamside toxicity testing was conducted using flow-through aquaria with fathead minnows. In addition, instream biological assessments were conducted along with water and sediment quality measurements. The results of the flow-through tests showed no toxicity in the fathead minnows until 14 days after exposure and 80 percent mortality after 25 days of exposure, indicating that short-term toxicity testing likely underestimates the toxicity of stormwater in receiving waters.

A study in North Carolina found that stormwater runoff from vehicle service and fueling stations had consistently elevated levels of polyaromatic hydrocarbon (PAH) compounds, MTBE, and other potentially toxic contaminants (Borden et al., 2002).

Runoff from agricultural or landscaped areas can also contain significant levels of potential toxicants, especially pesticides and herbicides (Liess et al., 1999; Thomas et al., 2001; Neumann et al., 2002; Arnold et al., 2004). These toxicants are also common in stormwater runoff from residential and urban landscaped areas (Pitt et al., 1995).

Sediment contaminated by stormwater runoff also has a detrimental effect on receiving water biota. Many of the observed biological effects associated with stormwater runoff and urban receiving waters may be caused by contaminated sediments, especially those impacts observed on benthic organisms. In addition, mortality of benthic invertebrates can be high in urban streams, especially during low flow periods, suggesting that toxicity associated with exposure to contaminated sediment, concentration of toxics in the water column, and/or ingestion of contaminated OM particulate is to blame (Pratt et al., 1981; Medeiros et al., 1983; Black et al., 2000).

Studies of urban stream sediments have shown the effects of metal toxicity on early life stages of fish and invertebrates (Boxall and Maltby, 1995; Hatch and Burton, 1999; Skinner et al., 1999; Lieb and Carline, 2000). Developmental problems and toxicity have been attributed to the contaminant accumulation in sediments and the remobilization of contaminated sediments during storm events (Skinner et al., 1999). Hatch and Burton (1999) also observed significant toxicity at a stormwater outfall site where sediments were found to be contaminated by multiple stormwater-related pollutants. Lieb and Carline (2000) showed that metals were more prevalent in stream sediments downstream of a stormwater treatment pond than upstream in a natural area. However, no acute toxic effects were noted. Zinc (Rose et al., 2000) and copper (Boulanger and Nikolaidis, 2003) are the most common metals found in urban sediments contaminated by stormwater runoff. These metals can be quite mobile under typical conditions found in urban receiving waters, but in most cases, a majority of the metal ions are bound to fine sediment particles and are not generally bioavailable. Examples of elevated levels of stormwater-related toxicants accumulating in urban stream sediments are numerous (Pitt, 2002). The levels of metals in urban stream sediments are typically orders of magnitude greater than those in the water column (DePinto et al., 1980; Pitt and Bozeman, 1982; Scott et al., 1983; May et al., 1997). Similar results are found when analyzing marine sediments from urban estuaries with stormwater discharges (Long et al., 1996; Morrisey et al., 1997; Bolton et al., 2003).

Stormwater Toxicity in Estuarine-Nearshore Areas

The effects of watershed development and stormwater runoff extend into marine waters at the mouths of streams (sub-estuaries) and in the nearshore environment of coastal regions. As with freshwater receiving waters, these impacts include physical, chemical, and biological effects.

Several studies on the toxic effects of water pollution on salmon have been conducted in the Puget Sound region and the Lower Columbia River Estuary in the Pacific Northwest (McCain et al., 1990; Varanasi et al., 1993; Casillas et al., 1995; Casillas et al., 1998; Collier et al., 1998). In these studies, there were demonstrable chronic toxilogical effects (immuno-suppression, reduced disease resistance, and reduced growth) of PAHs, PCBs, and other organic pollutants seen in juvenile and adult salmon.

A study of the Hillsborough River in Tampa Bay, Florida investigated the impacts of stormwater runoff on estuarine biota (MML, 1984). Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. No significant stormwater toxicity-related impacts were noted.

In a study of multiple stormwater discharge sites in Massachusetts Bay, high levels of PAH compounds were found in receiving waters and estuarine sediments (Menzie et al., 2002). Land use was a critical factor in determining pollutant composition and concentrations, with urbanized areas (mixed residential, commercial, and industrial land uses) having the highest pollutant (PAH) levels. No toxicity testing was conducted.

A study of stormwater discharges from Chollas Creek into San Diego Bay, California, indicated measurable toxic effects to aquatic life (Schiff et al., 2003). This study found that a toxic plume from the freshwater creek extended into the estuary, with the highest toxicity observed closest to the creek mouth. The toxicity decreased with increasing distance from the mouth due to mixing and dilution. Toxicity identification evaluation (TIE) methods were used, and it was found that trace metals from stormwater runoff were most likely responsible for the plume's toxicity to the sea urchins used in this study (Schiff et al., 2003).

A study of the water quality impacts of stormwater runoff into Santa Monica Bay, California also identified toxic effects in the estuarine receiving waters (Bay et al., 2003). As in the San Diego study, the freshwater plume from an urbanized stream (Ballona Creek) was responsible for the toxicity observed in marine organisms. Stormwater-transported metals (mainly zinc) were identified as the most likely toxic constituent. The only toxic effects noted were chronic, not acute. As in the previously discussed study, the toxicity decreased with increasing distance from the mouth due to mixing and dilution (Bay et al., 2003). Sediments in estuarine areas were also found to be highly contaminated by stormwater pollutants (Schiff and Bay, 2003).

Several studies on the toxic effects of stormwater runoff on native biota have been conducted in the Puget Sound region. One of the first studies looked at the uptake of aromatic and chlorinated hydrocarbons by juvenile chinook (McCain et al., 1990). This study found no acute toxicity, but identified numerous potential chronic impacts on growth and survival. In a related study, juvenile chinook salmon from both a contaminated urban estuary and a non-urban estuary were studied for two years (Stein et al., 1995). Exposure to aromatic and chlorinated hydrocarbons was measured, and both PAH and PCB levels in fish from the urban estuary were significantly higher than in fish from the non-urban estuary. The results of these studies indicate that out-migrant juvenile salmon have an increased exposure to chemical contamination in urban estuaries during their residence time in these habitats. This exposure was determined to be sufficient to elicit biochemical responses and to have the potential for chronic toxicity effects (Stein et al., 1995).

Runoff from urban areas can also contain significant levels of pesticides and herbicides at levels that have been shown to be potentially toxic to native biota (Bortleson, 1997; MacCoy and Black, 1998; Voss et al., 1999; Black et al., 2000; Hoffman et al., 2000). In a study conducted by King County, Washington, pesticides and herbicides in runoff and urban streams were linked to retail sales of the same pesticides within the urban watersheds under study (Voss and Embrey, 2000). The most common pesticides and herbicides detected during storm events included diazinon, 2-4-D, dichlorbenil, MCPP, prometon, and trichlopyr (Voss and Embrey, 2000).

Diazinon has been shown to have neurotoxic effects on salmon (Scholz et al., 2000). At sublethal levels, it was shown to disrupt homing behavior in chinook salmon by inhibiting olfactory-mediated responses (Scholz et al., 2000). This may have significant negative consequences for the survival and reproductive success of native salmonids.

Short-term exposures to copper (such as during storm runoff events in urban areas) have also been demonstrated to have sublethal effects on coho salmon by inhibiting the olfactory nervous system (Baldwin et al., 2003). In this study, the neurotoxic effects of copper were found to be dose-dependent, having a measurable effect over a broad range of concentrations. These effects occurred rapidly upon exposure to copper. It was concluded that short-term exposures can interfere with olfactory-mediated behaviors in juvenile coho salmon and may impact survival or migratory success of native salmonids (Baldwin et al., 2003).

Impacts of Contaminated Aquatic Sediment on Benthic Organisms

At some point in their life cycle, many aquatic organisms have their principal habitat in, on, or near sediment. Examples of this include benthic macroinvertebrates that spend almost their entire larval stage in contact with sediments. In the Pacific Northwest, salmonids also spend an extensive portion of their embryonic life stage within the benthic environment of their natal stream. In addition to functioning as benthic habitat, sediments can also capture and retain pollutants introduced by urban runoff. Pollutants enter sediments in several ways. The most direct path is the settling of suspended solids. Sediments deposited by urban runoff can physically degrade the substrata by filling interstitial spaces utilized as habitat by benthic organisms or by reducing DO transfer within the benthic environment. Dissolved pollutants can also move out of solution and into sediments by such mechanisms as adsorption of metals and organics at the sediment surface ion exchange of heavy metals in water with native calcium, magnesium, and other minerals in sediments, as well as the precipitation of phosphorus (Burton and Pitt, 2002).]

Most aquatic sediments have a large capacity to receive such contaminants through these processes. Also, many of the particulate pollutants are conservative. Once in the sediment, they do not decompose or significantly change form. These conservative pollutants include refractory organic chemicals relatively resistant to biodegradation as well as all metals. Consequently, these types of pollutants progressively accumulate in sediments. Over the long term, discharge of even modest quantities of pollutants can result in sediment concentrations several orders of magnitude higher than in the overlying water. These contaminant reservoirs can be toxic to aquatic life they come in direct contact with, as well as contaminate reservoirs far beyond the benthic (bottom-dwelling) organisms by bio-magnification through the food web (Burton and Pitt, 2002).

Historically, water quality has received more attention than sediment contamination. In the past 10 to 15 years, this approach has changed because of mounting evidence of environmental degradation in areas that meet water quality criteria. However, sediment toxicity investigations are limited because we lack accepted testing methods and do not understand the factors that control contaminant bioavailability. The result is an approach that emphasizes bioassay exposure techniques, either in situ or in the laboratory, along with chemical analysis of the sediments, overlying water, and/or sediment interstitial water. Very few studies have focused on the eco-toxicology of contaminated sediments in the natural environment (Chapman et al., 1998).

Case Study: Urban Stormwater and Metal Toxicity

Metals are a significant pollution component of urban stormwater runoff and non-point source (NPS) pollution. Heavy metals are of particular interest because many cannot be chemically transformed or destroyed and are therefore a potential long-term source of toxicity in the aquatic environment (Allen et al., 2000). Although the specific metals and their concentrations may vary widely depending on the anthropogenic sources present, they are common to almost all water pollution. Many trace metals are important as micronutrients for both plants and animals, playing essential roles in metabolism and growth. These include iron (Fe), zinc (Zn), copper (Cu), and Manganese (Mn), to name a few. Nutrient requirements vary between species, life stages, and sexes, but normal concentrations of these micronutrient trace metals are low and typically fall within a narrow acceptable band. Exposure to concentrations outside the optimal range can have deleterious or even toxic effects. Other trace metals which are not essential, such as lead (Pb), cadmium (Cd), and mercury (Hg) can be toxic at very low levels, either acutely or due to chronic/long-term exposure. Aluminum (Al), chromium (Cr), and nickel (Ni) are also found in urban runoff.

Anthropogenic sources of metal pollution are common throughout the environment. These include industrial processes, mining, and urban storm runoff. Urban runoff can contain a wide variety of trace metals from sewage discharges, fossil-fuel combustion, automobile traffic, anti-corrosion products, and various industrial sources. In general, the concentration, storage, and transport of metals in urban runoff or streams are closely related to OM content and sediment characteristics. Fine sediment, especially organic material, has a high binding capacity for metals, resulting, as mentioned above, in generally higher levels of metal contamination in sediments than in the water column (Rhoads and Cahill, 1999).

Several studies have been conducted to characterize the levels of metals in stormwater runoff, receiving waters, and sediments (Bryan, 1974; Wilbur and Hunter, 1979; Pitt et al., 1995; May et al., 1997; Neal et al., 1997; Sansalone and Buchberger, 1997; Barrett et al., 1998; Wu et al., 1996; Wu et al., 1998; Allen et al., 2000). Generally, the levels of various metals in stormwater are quite variable and dependent on a number of factors, including background watershed characteristics, land use practices, and specific sources (see discussion in Chapter 3).

Certain urban-stream organisms, including algae, arthropods, mollusks, and annelids, have exhibited elevated levels of metal concentrations (Davis and George, 1987). Ecological responses to metals occur at all levels in the ecosystem and include the loss of sensitive taxa, both chronic and acute toxicity effects, and altered community structure.

One study (Pitt et al., 1995) of urban stormwater samples, using the Micro-Tox toxicity-screening procedure, found that less than 10 percent of samples were classified as extremely toxic, a bit over 30 percent were moderately toxic, and the majority (about 60 percent) showed no evidence of toxic effects. The Micro-Tox methodology was only used to compare relative toxicities of various samples and not as a measure of absolute toxicity or to predict long-term toxic effects of stormwater on receiving waters. It does point to the fact that in all but a few heavily polluted systems, the level of toxicants in urban runoff is typically near detection limits (Pitt et al., 1995).

The toxicity of metals to aquatic plants and organisms is influenced by chemical, physical, and biological factors. Water chemistry characteristics such as temperature, pH, alkalinity, and hardness all affect metal toxicity. Physical aspects of exposure, such as metal speciation, duration of exposure, intensity of exposure events, and inorganic or organic ligand binding, also have a significant bearing on metal toxicity (Davies, 1986). Bioavailability of metals, the life stage of the affected organisms, organism health, and the natural sensitivity of the species involved are also important determinants of metal toxicity. Aquatic toxicology data generally indicates that the ionic fraction of metals constitutes the primary toxic form (Roline, 1988).

Acute toxicity to aquatic organisms can be manifested as a wide range of effects, from reduced growth rate to mortality. Laboratory studies on the mechanism of toxicity of zinc to fish in general indicate that zinc causes death via gill hypoxia (excess mucous secretion and suffocation) and gill tissue necrosis (Davies, 1986). Osmoregulatory failure appears to be the most likely effect of acute copper toxicity. Lead and mercury affect the central nervous system coordination of activity in fish, as well as interfering with cellular osmoregulation (Pagenkopf, 1983). The metal species present in solution and the ambient water chemistry can have a significant influence on metal toxicity. Consideration of total metal concentration alone can be misleading because chemical speciation of trace metals significantly affects the bioavailability to aquatic organisms and thus the ultimate toxicity (Davies, 1986). For the most part, organisms assimilate uncomplexed metal ions more readily than complexed forms. Increases in pH, alkalinity, and hardness generally decrease metal toxicity. Hardness ($Ca+$ and $Mg++$) has an antagonistic effect on metal toxicity in that the calcium and magnesium ions compete with metal ions for uptake sites on the gill surfaces, thus reducing the toxic effects of the metal ions (Davies, 1986). Alkalinity reduces metal toxicity through the buffering mechanism of the carbonate system. Under pH control, the carbonate and bicarbonate ions complex metal ions into soluble or insoluble, less toxic forms (Pagenkopf, 1983). In most cases, in alkaline waters, metals do not reach toxic levels until their concentration overwhelms the natural buffering capacity of the carbonate system.

Organic ligands can also complex metal ions, thus reducing toxicity by binding metals to particulates and making them relatively non-bioavailable. Metal toxicity generally increases when ambient temperature rises, due to the combined effects of an increase in both organism metabolism and chemical activity. Light intensity may also have a synergistic affect on the toxicity of some metals.

Chronic toxicity of metals is generally most apparent in the embryonic and larval stages of aquatic organisms and the early life stages of aquatic plants. As a period of rapid development, the early life stage is the most sensitive stage of the organism's life cycle for metal toxicity in general and other toxicants as well. Embryogenesis is a particularly sensitive period for fish with regard to metals (Davies, 1986). The period of larval settlement is the critical phase in invertebrate life history, although invertebrates as a whole are generally less sensitive than fish to trace/heavy metal toxicity (Nehring, 1976; Winner et al, 1980; Pratt et al, 1981; Garie and McIntosh, 1986). Chronic and sublethal effects of metals include reduced growth rates, developmental or behavioral abnormalities, reproductive effects, interference with metabolic enzyme systems, anemia, neurological defects, and kidney dysfunction (Davies, 1986). Due to the greater sensitivity of young organisms to metals, any exposures during embryonic development or rearing periods can, apart from the immediate effects, also manifest themselves in the adult organisms. There has been some indication that fish exposure to very low levels of metals during early life stages can result in an acclimation effect, making them somewhat more resistant to future periodic exposures (Davies, 1986). As with most toxicants, metal toxicity also increases with exposure period. Therefore, the intermittent nature of urban runoff may be less harmful to some aquatic life forms than continuous exposure to elevated metal concentrations would. Bioaccumulation of metals in organisms is also highly variable, depending on the particular metal, its chemical form, the mode of uptake, and the storage mechanisms of the organism. In low alkalinity (soft) waters, most metal species are of the "free" form. In alkaline (hard) waters, more metal ions are complexed, but some portion may remain in the ionic forms, especially if the buffering capacity of the natural water is overwhelmed. System pH also plays a major role in determining the speciation of the metal forms in freshwater (Davies, 1986). The rate of chemical (metals) reactions or chemical kinetics is also important to understanding the overall metal toxicity process. Such reactions as complexation do not occur instantaneously in natural waters. In the case of stormwater, runoff time scales may not allow sufficient time for complexation to take place, thus mitigating or negating the toxicity-reducing buffering effects (Pitt et al., 1995).

The use of aquatic insects and other macroinvertebrates as indicators of the biological integrity of lotic ecosystems is not new. One of the earliest field studies (Nehring, 1976) involved using aquatic insects as biological monitors of heavy metal pollution in the analysis and prevention of fish kills. Macroinvertebrates are generally more tolerant of metal pollution than most species of fish found in western streams (salmonids, sculpins, etc.) and tend to bioaccumulate metals in proportion to the in-water concentration (Nehring, 1976). In contrast to the more mobile fish species, macroinvertebrates are relatively sessile organisms. They also constitute an important part of the lotic food web, being the primary food source of most stream fishes. This makes them a useful surrogate for the economically and culturally important fish that inhabit the streams of the western states. In addition, some species of macroinvertebrates turned out to be more sensitive to metal pollution than others. This concept of "tolerant" and "sensitive" groups/species has become an important aspect of macroinvertebrate-based indices of pollution (Winner et al., 1980). In general, stoneflies (Plecoptera) and mayflies (Ephemeroptera) are sensitive to metal pollution, caddisflies (Trichoptera) are moderately sensitive/tolerant, and midges (Chironomids) are metal pollution-tolerant (Garie and McIntosh, 1986).

Field studies into the impact of urban runoff on lotic systems often use macroinvertebrate community structure as an indicator of ecosystem degradation. Many studies have found that, although urban runoff is the causal agent of ecosystem disruption, the impacts of stormwater pollution events are not just short-term. Partitioning of pollutants, especially metals, into sediments has been shown to have long-term ecological consequences on the primarily benthic-dwelling macroinvertebrate community structure (Pratt et al, 1981). In many cases, analysis of stormwater samples will not detect significant metals either in the dissolved or particulate form, but sediment samples will show metal accumulation bound to organic and inorganic ligands (Whiting and Clifford, 1983). Urban stormwater pollution is by its nature sporadic and acts as a physical and chemical pulse on the receiving water ecosystem. Higher levels of urban pollutants, such as metals and hydrocarbons, are typically found during "flushing" storm events (Pitt et al., 1995). Also coincident with these elevated pollution level events is increased flow over the period of the storm. These "scouring," high-energy flows have been shown to have a negative synergistic impact on benthic populations (Borchardt and Statzner, 1990). Some benthic species tend to migrate downstream or "drift" during stormflow conditions or pollutant events, while others try to avoid exposure by burrowing into the substrate.

One of the first comprehensive studies of the effects of urban runoff on benthic macroinvertebrates in streams was conducted on the East Coast (Garie and McIntosh, 1986). This was a typical upstream (control) compared to downstream (impacted) site study. Lead, zinc, and chromium were the predominant metals found in the stormwater. Macroinvertebrate diversity (number of taxa) and changes in community composition were used as the primary measures of impact. The results of this study again showed that there are both "tolerant" and "sensitive" species with regard to metal toxicity and urban runoff impact. The study also confirmed that elevated pollutant concentrations during urban runoff storm events were short-term and transient in nature, and it was hypothesized that the real impact on macroinvertebrate communities lay in long-term exposure to metals accumulating in the benthic sediments. This points out one of the potential flaws of using macroinvertebrates as biological surrogates for fish in that fish are generally not exposed to the sediment chemistry that the benthos are. Another very comprehensive study conducted in the Pacific Northwest showed that, although macroinvertebrate community structure was significantly changed due to urbanization impacts, the fish population structure of impacted and control streams remained largely the same (Pedersen and Perkins, 1986). Apparently, salmonids feed on available benthos and do not select for specific trophic groups or species. This is not to say that a shift in benthic community structure is not a good indicator of urban impact, but one must be careful in extrapolating the results of one group of organisms to other biota, even if they are closely linked within the food web. The PNW study also demonstrated a lack of consistency when trying to use complex macroinvertebrate diversity indices to gauge the level of urban impact. Natural variability was generally too high and effectively masked any well-defined correlations.

Aquatic insect sampling and analysis has, however, been shown to be very useful as a tool for assessing other impacts of metal pollution. The usefulness of benthic macroinvertebrates as monitors of bioavailable metal concentration and long-term bioaccumulation of metals has been demonstrated (Kiffney and Clements, 1993). Still other studies have highlighted the synergistic (negative) impacts of metals and other habitat degradations on aquatic ecosystems in general (Clements, 1994; Hoiland and Rabe, 1992). Finally, the persistence of sediment metal levels and resultant long recovery times has been shown for macroinvertebrate communities exposed to prolonged pollution inputs in the field (Chadwick et al., 1985).

Urban Runoff and Eutrophication

Watershed urbanization generally leads to higher nutrient (phosphorus and nitrogen) concentrations in stormwater runoff (Omernik, 1976). Phosphorus is generally found in particulate form, but the more bioavailable, dissolved forms are also common. Nitrogen is typically found in the nitrate or ammonium form. Sources of nutrients in urbanizing catchments include lawn and garden fertilizers, wastewater (failing septic systems and WWTP discharges), and fine sediment from erosion or street runoff. Although nutrient pollution is often associated more with agricultural activities, urbanization can contribute significant quantities of nutrients to receiving waters (Omernik, 1976).

Eutrophication is the process through which excess nutrients cause overall algal biomass increases, especially during "bloom" periods. This is due to increased loading of the nutrient that had previously been in shortest supply relative to need. This limiting nutrient is usually either phosphorus or nitrogen, but most often, and most consistently, it is phosphorus in freshwater lakes. In estuarine or marine nearshore areas, nitrogen is typically the limiting nutrient. In addition to promoting larger quantities of algae, nutrient enrichment typically changes the composition of the algal community. One-celled diatoms give way to filamentous green forms, followed by blue-green forms (some toxic) with a larger nutrient supply (Welch, 1980; Welch et al., 1988; Welch et al., 1989; Welch et al., 1992).

As discussed earlier, urban areas have a number of nutrient sources, and nutrient loadings increase with the development level. Eutrophication degrades lake and estuarine ecosystems in several ways. The filamentous algae are poorer food than diatoms to herbivores because of their structure and, sometimes, bad taste and toxicity. Filamentous algae clog water intakes and boat propellers and form odorous masses when they wash up on beaches. They also reduce water clarity, further limiting beneficial uses. When a large biomass dies at the end of the bloom, its decomposition by bacteria creates high oxygen demand, which can result in severely depressed DO levels (Welch, 1980; Shuster et al., 1986; Walker, 1987). In addition to algal blooms and the associated negative impacts, eutrophication may result in an overall increase in other nuisance plants, including a variety of submerged or emergent aquatic macrophytes. Some of these plant communities may include invasive species such as hydrilla, Eurasian milfoil, purple loosestrife, and reed canary grass (Welch 1980).

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CHAPTER 5

Aquatic Monitoring and Program Design

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Obtaining conclusive data on urban water resources and stormwater runoff is difficult and expensive. Therefore, monitoring programs that collect data must be carefully designed to be cost-effective. This chapter suggests a general process for designing programs, whether the monitoring subject is natural or runoff water, sediments, or biological community characteristics or organisms. The recommended system thus applies to the design of monitoring programs whose elements are detailed in Chapters 6 and 7.

This process originated in research to improve monitoring program design in urban runoff and related fields (Reinelt, Horner, and Castensson, 1992; Reinelt, Horner, and Mar, 1988; Mar et al., 1986). Burton and Pitt (2002) have written very extensively on all aspects of monitoring program design and execution. Their Chapter 4 covers the program design aspects of monitoring and presents a number of case studies. Some citations to that work are given in this and the two following chapters. Those who wish to pursue any subject in detail should consult Burton and Pitt, using their helpful 37-page index.

The suggested analytical process has five steps:

1. Specify monitoring program objectives;
2. Determine the level of effort to devote to the analysis;
3. Perform a systematic analysis appropriate to the problem and objectives;
4. Use the analysis results to tentatively specify monitoring program elements; and
5. Evaluate the tentative monitoring program for cost-effectiveness and finalize according to evaluation results.

Each of these steps entails numerous tasks and decisions that are essential to arriving at a well-founded monitoring program design. Ultimately, the purpose of working through these steps is to determine the best combination of program elements that will achieve the objectives with an acceptable level of assurance in the results at known cost.

Monitoring Program Design Steps

Step 1: Specify Monitoring Program Objectives

Establishing objectives for environmental monitoring is essential, even though they cannot always be specified in great detail. Thoughtful statements, agreed upon by all concerned, should guide the monitoring program design and conduct. Objectives follow from the nature of the problem or the particular decision-making need that requires data collection.

Every monitoring program should, if possible, formulate objectives at two levels, general and specific. General objectives describe what must be accomplished to solve the overall problem or meet the need. Examples are:

- Determine if a water body meets water quality standards applying to it.
- Define water quality conditions in a lake prior to shoreline development.
- Determine long-term trends in sediment accumulation of metals in a poorly flushed bay.

- Find and quantify the contamination source that has closed a shellfish bed to harvesting.
- Calibrate and verify a specific runoff rate simulation model.
- Assess the relative health of a benthic macroinvertebrate community.

Specific objectives relate directly to measurements and produce results to meet the general objectives. Some examples related to the fourth general objective above are:

- Determine the annual fecal coliform loadings contributed by agricultural, septic drain field, and urban runoff sources.
- Identify individual sources of fecal coliform loading that should be addressed to achieve 70 percent reduction in total annual loading.

These objectives can be stated in more detail and more specifically when Step 3 of the general design process is completed. For example, more analysis could permit recasting the second specific objective as:

- Rate individual sources of fecal coliform loading in terms of annual loading generated and potential for reduction, and identify the best combination of sources to address that would lead to a 70 percent reduction in total annual loading.

This chapter emphasizes the point that an environmental monitoring program should be regarded as a type of scientific experiment. The rigor of the scientific method promotes careful design of the program to make as sure as possible that it will yield answers with a known level of certainty. Accordingly, it is sometimes appropriate to state objectives in terms of a scientific hypothesis. An example would be:

- Determine the best way to target source controls to pollution sources, with the "null hypothesis" being that the annual loadings of a pollutant do not significantly differ between Source A and Source B, and the "alternative hypothesis" being that they do differ with statistical significance.

Step 5 explores this subject further. It supplies methods that allow the monitoring program designer to judge the likelihood of the hypotheses being successfully tested with the funds available.

Objectives form the foundation for the entire monitoring program. They should be consulted frequently in its development to make sure that decisions made along the way comply with the intentions represented by objectives. At the same time, they should not be overly rigid, in case circumstances change. In some cases, new information will make it possible to sharpen the objectives. In others, the findings through analysis may point out a lack of realism in the initial objective statements and suggest how they should be modified accordingly. Ultimately, the assessment in Step 5 will determine if the objectives are achievable with an acceptable level of assurance within the funds available. If this is unlikely, it is much better to change them to something that can be met and still fulfil a purpose than to proceed with a program that has little probability of success.

Using this process ensures careful decision-making at each step and counters the tendency to use a generic monitoring strategy that may not relate to the program goals. Exercising discipline to make careful assessments is the best way to be cost-effective in monitoring.

Step 2: Determine the Level of Effort

The effort put into monitoring program design can range from relatively simple and inexpensive to thorough and costly, depending on the objectives for the particular program. The development of specific objectives is an iterative process of adjusting goals in light of the quantity and type of information available, the detail of additional information needed, the resources of the designers, and the urgency to begin monitoring.

Available information can help target new monitoring and substantially reduce costs. Therefore, designers should incorporate this information in their analysis, using techniques in this manual. Some problems may not be worth extensive effort, while others require it. For example, existing data can help determine whether there is a problem in a particular location without necessitating the expense and effort of sampling. On the other hand, monitoring to allocate resources for solving problems in a large, complex watershed may require a substantially greater level of analysis.

Even if little guidance information exists and the designer has limited time and resources, at least the basic analytical process should be applied. After developing a preliminary information base, the designer can always review the systematic analysis of the problem and objectives in more detail later.

Step 3: Perform a Systematic Analysis

As the core of the process, Step 3 requires the most effort. The analyst should give priority to key factors thought to pertain to the issue. This systematic analysis is often referred to as a watershed analysis. The term "watershed" broadly signifies an area that drains a land surface to a point of interest. While a watershed can be a small catchment with a simple drainage system, for now we will limit the term to include only landscapes of some size and complexity draining through a network of artificial and natural conveyances to a natural water body. Thus, the analysis involves surveying watershed characteristics, identifying the most critical potential problems and sources, and highlighting the most critical places, times, and biological units that manifest these problems.

A watershed inventory involves collecting the level of data appropriate to the needs of the project. While the level of detail may vary, the inventory should include developing a basin map; identifying such features as land uses, soils, topographic information, and hydrologic data; and identifying potentially critical locations relative to the objectives adopted in Step 1. For example, if these objectives pertain to pollution sources, some possible locations of interest are earth-moving sites, industrial areas, and major traffic concentrations. If the objectives focus on aquatic resources, areas to recognize might be sites like fisheries and other productive resource areas, rare or endangered resources, and stream reaches vulnerable to major channel damage. Obtaining any available data on these features, as well as field reconnaissance, are key tasks in a watershed inventory. This includes identifying any already existing site data for possible use in Step 5 of the process.

Identifying critical problems and sources should be a systematic process of formulating a broad list and then narrowing it by prioritizing items, with the level of effort chosen in Step 2 dictating the scale of the analysis. For example, to find the principal sources of water quality deterioration in a river draining a large watershed, we may suspect that certain areas and activities need attention. However, this conclusion should be tested through some quantified, comparative estimates of pollution quantities, using models like those outlined in Chapter 3. Such models may be overly generalized, simplified, and not calibrated locally, but their purpose is not to reach a final decision but to guide the design of a monitoring program. Even with little effort, the simplest model can often bring objectivity and rigor to the analysis.

Identifying critical places, times, and receptor organisms presents a more difficult problem. We must at least conceptualize the relationship between problem and timing and the potential damage for habitats, species, and life stages. While models can sometimes help, they are usually too simple or inconvenient to be sufficient. Ideally, the specialists (e.g., water quality engineer, hydrologist) will work closely with an ecologist familiar with the water body, its ecology, and its natural history, to judge these critical factors.

It is advisable to review the original objectives for their continued appropriateness. Objectives will likely need to be modified or specified with increased knowledge.

Step 4: Tentatively Specify Monitoring Program Elements

General Considerations

If performed properly, the systematic analysis of Step 3 will provide sufficient information to give provisional shape to the monitoring program. In this step, it should be tentatively specified:

- What to sample;
- Where to sample;
- When to sample;
- How many samples to take at each site on each occasion (replicates);
- How to sample; and
- What to analyze in samples.

The philosophy behind this process is to base decisions on these program aspects on case-specific objectives and analysis. Working from prescribed sampling scopes and frequencies and standard lists of analyses should be avoided.

The list above represents the monitoring program elements that require tentative but concrete decisions, so a cost-effectiveness evaluation according to Step 5 can be conducted. Along with these basic decisions, some attention should be given at this point to the following additional elements:

- How to handle samples;
- Data quality objectives;

- Quality assurance/quality control checks; and
- How to analyze data.

Tentative judgments on these elements of a monitoring program will be further evaluated in Step 5 for their ability to meet the objectives with a known level of assurance and cost. These will be finalized after any necessary adjustments have been made and become the monitoring program design. The following two chapters on physical and chemical monitoring and biological community and toxicity assessment present details on each element, as appropriate to each monitoring topic.

What to Sample

This monitoring program element is the most straightforward one, specifying the medium or media to sample. It refers to the water body, runoff stream, sediment, habitat, biological community, or organism(s) from which samples are to be drawn.

Where to Sample

The question where to take samples must be carefully considered with respect to the established objectives and the analysis performed under Step 3. Obviously, the more locations samples are collected from, the higher the cost will be. Thus, the decision on each sampling point must be taken with an eye to its contribution to fulfilling the objectives.

In the study of urban waters, where to sample is often dictated or strongly guided by the objective of comparing two or more spatial conditions, e.g., a location affected by runoff discharge versus an unaffected one, a location served by a best management practice (BMP) versus one that is not. In experimental parlance, the affected spot is often referred to as the "treatment" site, in the sense that its condition is "treated" by the discharge or the BMP, while the other location is frequently termed the "control" or "reference" site. Accordingly, a comparative monitoring program can be termed a "control/treatment design." The term "control" stems from laboratory experimentation, where the investigator generally has a much higher degree of influence on the situation than in studies in the natural environment. The term "reference," which implies a basis for comparison rather than maximum influence, is thus often preferred in environmental monitoring. These concepts illustrate once again that a monitoring program should be regarded as a type of scientific experiment.

The comparison of two or more conditions can, and often should, be approached very systematically as a "paired watershed" monitoring program design in which the treatment site (or each of several treatment sites) is paired with the reference site. The rationale, as well as the methods, for paired watershed monitoring have been thoroughly developed over the last 15 years. The appendix to this chapter summarizes them in some detail and gives references for those interested in more information. Ultimately, this technique entails not only spatial considerations (i.e., where to sample) but also temporal ones (i.e., when to sample). This program element will be discussed below.

When to Sample

Fulfilling objectives and keeping monitoring costs within budget depends as strongly on decisions about when to sample as on where to sample. Also, the comparison objectives for urban water monitoring studies are often temporal rather than (or in addition to) spatial; i.e., the task is to compare two or more conditions separated in time. Examples would be: comparing the health of a macroinvertebrate community before and after major development in the watershed, or comparing storm runoff event mean concentrations of pollutants before and after the institution of BMPs.

As pointed out in the appendix to this chapter, it is advisable to employ, whenever possible, a reference station or stations and a control/treatment design in conjunction with temporal comparisons. The reference serves as a basis to distinguish natural variability or some other source of effect from the treatment that is the actual focus of the study. This monitoring program design lowers the risk of attributing an outcome to the treatment that is in fact the result of one or more extraneous factors. Designs of this type are sometimes referred to as before/after, control/treatment designs (BACT).

Beyond comparisons over a certain time period, there are other important considerations that enter into the decision when to sample:

- Seasonal considerations – Numerous conditions potentially instrumental to urban water monitoring objectives vary over the year. E.g., (1) many lakes stratify thermally in the summer and, depending on the climate, in the winter as well, with key implications for physical, chemical, and biological conditions; (2) living organisms pass through annual life cycles that must be recognized in scheduling their sampling; (3) pollutant delivery to water bodies accelerates in high runoff periods, while pollutants already present can concentrate in less diluted form during dry periods.

- Diurnal considerations – Some key conditions vary over the course of the day. These include (1) water temperature, dissolved oxygen, and sometimes pH in response to the photosynthetic and respiratory activities of aquatic life; (2) the variation in routine most organisms display over the course of a day, which may influence the ability to sample them.
- Considerations related to flow variation – Flow can vary depending on stochastic environmental conditions (e.g., runoff pattern in response to rainfall) and more predictable circumstances under human management (e.g., flow release from a dam).

All these factors may influence the decision when to sample. It is almost always inappropriate to devise a regular sampling schedule, as often occurs when schedules are adapted to the staff's regular work schedule.

Replicates

A replicate is a duplicate sample collected and handled in exactly the same way as the initial sample. Replicates are an important part of monitoring programs, not only for quality assurance/quality control (QA/QC) purposes, but also to define potential variability introduced from various sources. This subject will be elaborated on later in the chapter. Suffice it to say for now that, just like each sampling location and occasion, each replicate adds cost. Hence the number of replicates must be carefully considered in relation to the overall objectives, specific QA/QC requirements, and the budget.

How to Sample

This question regards the choice of sampling gear and its operation, clearly factors that affect costs and the achievement of objectives. These subjects are covered in depth in Chapters 6 and 7.

What to Analyze in Samples

The question of monitoring topics also directly affects costs and outcomes. It is covered in detail in the next two chapters.

How to Handle Samples

Once decisions are made about how samples will be collected and analyzed, handling them correctly is generally tightly standardized. Proper handling procedures ensure against alteration of samples between the time of collection and analysis, which would give false results. Chapters 6 and 7 cover the subject as appropriate to each monitoring type.

Data Quality Objectives

Data quality objectives, sometimes abbreviated as DQO, are statements (generally quantitative) representing the standards to which data will be held for acceptance and consideration in data analysis. They can be regarded as part of the set of overall objectives stated in Step 1. DQO achievement is assessed through quality assurance/quality control checks. The next two chapters on specific monitoring types give examples.

Quality Assurance/Quality Control

The effectiveness and credibility of any monitoring program depends on its quality assurance/quality control program, the control exercised on a data collection to assure, to the extent possible, a sound basis for drawing conclusions. The QA/QC program provides quantitative measurements of the "goodness" of the data. The most fundamental QA/QC concepts, applying in one way or another to every type of aquatic monitoring program, are:

- Representativeness – Results are representative when they truly reflect the population of interest, as framed in the objectives. The term "population" is used here in the general sense, referring to the aggregate of units from which samples will be drawn. Some examples are: (1) For a general stormwater runoff monitoring program, the population would be the full range of runoff events (the "units") over the whole duration of flow; whereas for a first-flush program, the population would encompass the full range, but over just the rising limb of the hydrograph (or some selected fraction of it). (2) For a river sediment contamination monitoring program, the population would be the complete distribution of sediment types (size fractions, etc.) in the reach of interest. (3) For a stream riparian cover assessment program, the population would be the cover types (vegetation, surfaces, etc.) of the lands extending a selected distance to either side of the stream centerline (or bank-full location) over the reach of interest. The objectives help determine what would be representative within their boundaries. There is clearly a tension in monitoring program design between selecting a representative number of units on the one hand, while staying within the bounds of feasibility and affordability on the other. The analysis outlined in Step 5 is an aid in resolving this tension.

- Accuracy – Accuracy is the agreement between the measurement of a variable in a sample and its true value. The term "error" refers to the discrepancy between the measured and true values (Error = Measured value – True value). Relative error expresses the error as a percentage deviation from the true value:

  Relative error (percent) = (Error/True value) x 100

QA/QC programs assess accuracy by testing samples that have set values of the variable being measured. These tests should be done blind (without the analyst knowing the value) to avoid bias.

- Precision – Precision is the agreement among replicate measurements. It is measured in absolute terms as the standard deviation of the set of replicates. More useful is relative deviation, which is the standard deviation being expressed as the percentage of the mean of the replicate values:

  Relative deviation (percent) = (Standard deviation/ Mean of measured values) x 100

Precision can, and generally should, be assessed with both field and laboratory replicates. Field replicates are separate samples collected simultaneously at the same source location and analyzed separately. They are used to assess total sample variability (i.e., field plus analytical variability). Laboratory replicates are repeated analyses of a variable performed on the contents of a single sample. They are used to assess analytical precision. Duplicate analyses of a single sample usually suffice for well-proven procedures in the laboratory. The extent of replication depends on overall objectives, data quality objectives, and monitoring program optimization for cost effectiveness (see Step 5). In the absence of any specific considerations arising from these factors, it is common for 5 to 10 percent of field samples and 5 to 10 percent of laboratory procedures to be duplicated to get some measure of precision.

Additional QA/QC terms and procedures apply to specific monitoring types, as covered in Chapters 6 and 7. Laboratory physical and chemical analyses of water and sediment samples in particular are subject to extensive further QA/QC checks.

Regardless of the monitoring type, QA/QC is best advanced by having well-qualified and -trained personnel involved at every point in the process. The quality of laboratory service should be ensured by writing detailed contract specifications, including the QA/QC checks to be performed, standards for acceptance, and action to be taken should results be unacceptable.

Data Analysis

Thinking about data analysis early is essential to ensure that measurements produce data that can be assessed to achieve program objectives. For example, it would be fruitless to collect sediment samples along a longitudinal transect (parallel to flow) and later try to define the extent of variability in the vicinity of a particular discharge. The connection between what one is trying to learn and how one is going to use data to gain that knowledge is not always as obvious as it should be. At this stage, each stated objective should be coupled with the data analysis scheme set out in the preliminary monitoring program design. The design should be finalized after Step 5 has been performed. It is, of course, still possible to implement additional data analysis procedures that may suggest themselves later on, but starting out with a sound basic plan is highly advisable.

The data analysis options in the areas of graphing, statistics, and multivariate analyses are almost unlimited. Exploring them all is far beyond the scope of this book. Chapters 6 and 7 present some techniques that have proven to be useful.

Step 5: Evaluate the Tentative Monitoring Program and Finalize

General Considerations

This step is an evaluation of the tentative monitoring program according to Step 4. It takes into account the total numbers of samples and analyses anticipated, as well as the allocation of effort among sampling locations and occasions, replicates, and analyses. These factors directly determine the program's cost and probable effectiveness. Monitoring programs frequently fail to provide the desired information, even when performed flawlessly, because the samples are insufficient to achieve an accepted level of statistical assurance. This failure results from a high variability in flow and natural aquatic systems that complicates monitoring. For example, variability prevents us from ascertaining with a high level of statistical confidence that an average water quality condition meets a certain criterion or that a new discharge creates a change in a biological community.

Sources of variability include spatial differences in a landscape or water body, differences over time (temporal variability), and measurement errors. Careful consideration of seasonal, diurnal, and flow-related factors in relation to the objectives, as discussed under Step 4, can help reduce this variability. Better techniques, if available, can reduce measurement errors. Otherwise, replicate samples will have to be collected in order to quantify the measurement error component. One strategy in dealing with sources of natural spatial and temporal variability, unless they are enormous, is to increase sample numbers, but this strategy raises cost.

The basic task in this step is to determine the number of samples (stations, occasions, and replicates) needed to meet the objectives, considering variability and budget limits. Using the optimal number of samples to reach a conclusion will result in either the maximum confidence level for a set budget, or in minimum cost for a set assurance level. These options, which represent two ways to maximize the monitoring program's cost-effectiveness, can only be applied if some data are already available to give statistical measures of central tendency (e.g., mean or median) and variance. In that case, statistical methods can be applied to the optimization problem.

In some cases, uncontrollable natural variability will be too great to achieve confidence in a certain program element within a feasible budget. In this case, the designer will have to either delete this element or reduce costs in other areas and redirect resources. The options are to reduce the sampling stations, occasions, replicates, the number of analyses prescribed, the costs of various program elements, or some combination thereof. This decision is often unpalatable because it can demand, for example, cutting geographic coverage or not analyzing for a water quality measure that is traditionally included. However, the designer must choose and target the program according to objectives and circumstances, rather than conduct a program that gives inconclusive or misleading answers.

Mar et al. (1986) present some straightforward strategies for optimizing monitoring program designs in common situations, which will be summarized in the following sections. Other situations may arise in designing aquatic monitoring programs, and different statistical methods exist to handle the various scenarios. Burton and Pitt (2002) summarizemany of these circumstances and techniques in their Table 5-3, with elaboration in the accompanying text. Other references with extensive coverage of the subject are Gilbert (1987) and Zar (1998).

Determining a Mean Value

Determining a mean value applies, for example, when an average water quality condition is compared to a regulatory criterion. In basic statistics, t-distribution defines the confidence interval for the mean of a normally distributed population (set of values) as estimated from a data set. The t-distribution is used to determine sample numbers if the data are demonstrated or assumed to have a normal probability distribution (Figure 5-1(A)), or if they can be transformed (e.g., by taking their logarithms) to yield a normal distribution.

Figure 5-2 presents the results of an analysis based on the t-distribution for three confidence levels. The curves show the number of samples required as a function of precision. Precision here is the ratio of the difference in the estimated and actual mean ($x - \mu$, the error that will be accepted) to the standard deviation (σ, the variation or "noise" in the data). To use the graph, the monitoring program designer consults available data to get estimates of the mean and standard deviation and decides on the acceptable error and confidence level. For the case of an acceptable error equal to the standard deviation (precision = 1) and an 80 percent confidence, for example, four samples suffice. Demanding a precision of 0.1, however, requires hundreds of samples.

If no data are available for this exercise, monitoring program designers have several options. They can conduct a pilot program to obtain a limited data set; however, this choice would require spending time and money. The alternative is to use data from a similar location or estimate values using professional judgment. Either course has obvious drawbacks in accuracy, but both are usually superior to making an educated guess of the sample numbers and allocation with qualitative, but no quantitative, analysis. Even that option is better, though, than blindly specifying monitoring program elements without any analysis.

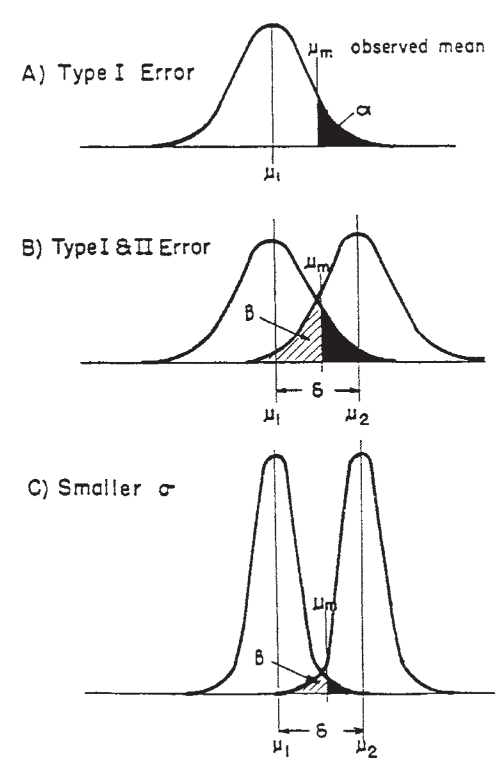
Detecting Change

Detecting change applies, for example, when the size or composition of a biological community is evaluated at two different points in time. Programs designed to detect change require different statistics than those that simply identify means. This type of problem is phrased as a statistical hypothesis test in which the null hypothesis (H_o) is that the populations are from the same distribution at both points in time; the alternative hypothesis (H_1) is that they are from different distributions.

Figure 5-1 illustrates terminology needed for this type of evaluation. The shaded area of Figure 5-1(A) represents the probability (a) of a Type I error (H_o was rejected when it was, in fact, true). Figure 5-1(B) shows distributions at both points in time, in which the difference in means represents an apparent change of magnitude (δ). The hatched area represents the probability (α) of a Type II error (H_o was accepted when it was, in fact, false). The quantity (l – ß) is termed the power of a statistical hypothesis test. Figure 5-1(C), in comparison to the other two graphs, illustrates the variation effect, as represented by the standard deviation, on power. For a given change, δ, the power increases as the standard deviation decreases.

Figure 5-3 provides a graphic way to establish the number of samples needed to detect change. To use the

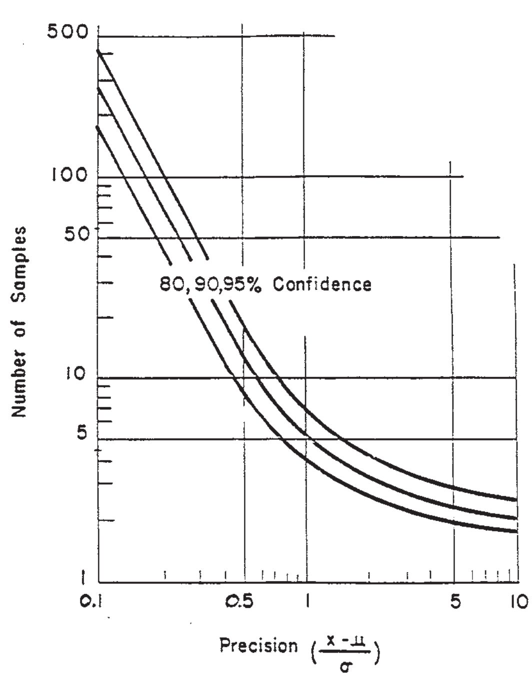
Figure 5-1: Hypothesis-Testing Fundamentals



μ_1 and μ_2 are the means of two populations of measurements; their difference represents a change δ; α and β are the probabilities of the Types I and II errors, respectively.

Source: Mar et al., 1986

Figure 5-2: Number of Samples Versus Precision



Precision is the ratio of the allowable error to the standard deviation of the population of measurements.

Source: Mar et al., 1986

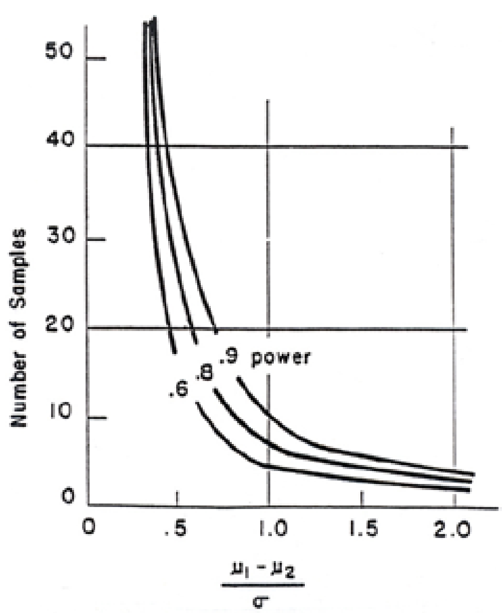
graph, the monitoring program designer consults any available data to estimate the standard deviation and decides on the magnitude of change to be detected ($\delta = \mu_1 - \mu_2$) and the power. Suppose, for example, that the objective is to detect a change of 5 units in a population previously characterized to have a standard deviation of 8 units with statistical power of 0.8. The ratio of change to standard deviation is 0.625, requiring 20 samples. This plot shows that to detect changes of less than 50 percent of the standard deviation, the program requires a large number of samples.

Monitoring Costs

The statistical methods previously illustrated show how to measure the value of added information in the form of more samples in promoting program effectiveness. To optimize the program, cost estimation must accompany these methods. Given the cost and value of added data, a trade-off analysis can be performed to obtain the most cost-effective program within the existing con-straints. Costs are accounted as follows:

$$TC = C_o + (T)(C_t) + (S)(T)(C_s) + (R)(S)(T)(C_r)$$

Figure 5-3: Power Curves Relating Number of Samples to Change to be Detected



σ is the standard deviation

Source: Mar et al., 1986

where: TC = Total cost;

C_o = Fixed overhead cost;

C_t = Fixed cost for each sampling occasion;

C_s = Cost associated with visiting each sampling station;

C_r = Cost to collect and analyze each sample;

T = Number of sampling occasions;

S = Number of sampling stations; and

R = Number of replicates on each occasion at each station.

C_o represents such costs as maintaining staff and space for the overall program to support all the work outside of going out to take samples and analyzing them (e.g., equipment inventory, monitoring program design, data analysis, reporting, administration). C_t is the cost of mobilizing for a sampling date (e.g., acquiring sampling supplies, paying a daily vehicle charge). C_s represents the expenses of travel to field locations and time spent collecting samples and delivering them to the site of analysis. Finally, C_r is the price of analyzing one sample, plus any cost, other than staff time, of the sample collection and handling process (e.g., chemical preservative). It should be noted that, regardless of these considerations concerning finances and ability to draw conclusions in the face of natural variability, some field (as well as laboratory) replicates must be taken for QA/QC purposes. If cost accounting can be done in this manner, monitoring program optimization can be performed relatively easily.

Note that (R)(S)(T) = the total number of sam-ples. For a given total, the three quantities can be varied so long as their product remains the same. If measurement error is larger than natural varia-tion, then adding replicates would reduce uncer-tainty more than adding stations or occasions. However, if spatial or temporal variation domi-nates, adding stations or occasions, respectively, would be a better strategy.

Optimization Examples

Example 1

The first example concerns selecting sample numbers to estimate mean values. Suppose three variables (A, B, and C) are to be monitored to establish their annual means at a site with 90 percent confidence. Table 5-1(a) gives variability from hypothetical pilot data and measurement costs of each. Overhead cost is equal among designs and is not considered in the calculations. This example illustrates optimizing the monitoring program for a given budget (Designs 1 and 2) and a given level of assurance (Design 3):

- Design 1 – Collect three samples for each variable (fixed cost of $420). What is the minimum error that can be attained for each variable?
- Design 2 – Collect four samples for each variable (fixed cost of $560). What is the minimum error that can be attained for each variable?
- Design 3 – A fixed error of = 40 percent of the mean is required for each variable. What design (sample numbers) provides this level of certainty at the minimum cost?

Table 5-1(b-d) summarizes the evaluation. Comparing Design 2 to Design 1 indicates that an in-creased but equal number of samples would only slightly improve the estimate of the mean for each variable. The estimate for variable B would still be highly uncertain relative to the others. However, as Design 3 shows, allocating more samples to the variable with the greatest variation, and with the most improvement results per dollar spent, provides an overall more cost-effective design. With this design, the estimate of the mean of variable B is expected to improve greatly, with some increase in the error for variable A but little for C, at about the same cost as Design 2.

In most actual cases, a simple analysis like this is insufficient, since uncertainties can result from several factors, including measurement er-rors and spatial and temporal variability. Generally, an analysis should be performed to inves-tigate each cost and variance component and their effects on the design to allocate the effort among

Table 5-1: Monitoring Program Optimization Hypothetical Example for Estimation of Means

(a) Sample Costs and Variability			
Input Data	Variable A	Variable B	Variable C
Total cost per sample	$100	$10	$30
Standard deviation (% of mean)	10	100	20
(b) Design 1 (optimization for fixed budget, three samples of each variable)			
Variable	Cost	Precision[a]	Error (% of Mean)[b]
A	$300	2.4	24
B	$30	2.4	240
C	$90	2.4	48
Total	$420		
(c) Design 2 (optimization for fixed budget, four samples of each variable)			
Variable	Cost	Precision[a]	Error (% of Mean)[b]
A	$400	2.2	22
B	$40	2.2	220
C	$120	2.2	44
Total	$560		

(d) Design 3 (optimization for fixed allowable error of = 40 percent of the mean of each variable)				
Variable	Error (% of Mean)	Precision[c]	Number of Samples[a]	Cost
A	40	4.0	3	$300
B	40	0.40	18	$180
C	40	2.0	3	$90
Total				$570

Notes: [a] From Figure 5-2 for 90 percent confidence.
[b] Error = Precision x Standard deviation.
[c] Precision = Error/Standard deviation.

sampling locations, occasions, and repli-cates. Example 2 illustrates this type of analysis.

Example 2

The second example concerns selecting sample numbers to determine whether or not the mean value of a variable at a particular location (one station) changes over time. A pilot sampling program estimated the standard deviation at 8 units, 10 sampling occasions, and two replicates. The standard deviation is estimated to decrease or increase by 0.5 unit with each added or subtracted sampling occasion, respectively, and to decrease or increase by 0.2 unit with each added or subtracted replicate, respectively. Change should be detectable with statistical power of 0.8. Cost components are: C_o = \$15,000; C_t = \$300; C_s = \$150; and C_r = \$500. This example illustrates optimizing the monitoring program for a given budget (Design 1) and a given level of detectability (Design 2):

- Design 1 – What is the optimum allocation of sampling occasions and replicates to minimize the detectable change for a budget of \$30,000?
- Design 2 – What is the optimum allocation of sampling occasions and replicates to minimize the cost of detecting a change of 5 units or smaller?

Table 5-2(a–b) summarizes the evaluation. In Design 1, reducing occasions in favor of replicates did not lower detectability. Increasing occasions to 16 with a single sample taken each time reduced detectability substantially. Lack of replication generally would not be acceptable for QA/QC purposes, and a replicate sample would be collected and analyzed on at least one sampling occasion, preferably two. A small budget increase or slight loss of detectability would be necessary to accommodate the replication. In Design 2, reducing occasions in favor of replicates raised cost, whereas increasing to 12 occasions without replication yielded the lowest cost. The same qualification stated for Design 1 pertains to replication.

Table 5-2: Monitoring Program Optimization Example for Determination of Change

Sampling Occasions	Replicates[a]	Total Samples[a]	Standard Deviation (Sd)	δ/sd[b]	δ
(a) Design 1 (optimization for fixed budget of \$30,000)					
10	2.0	20	8.00	0.63	5.0
8	2.9	23	8.83	0.57	5.0
14	1.2	17	6.15	0.67	4.1
16	1.0	16	5.21	0.68	3.5
(b) Design 2 (optimization for fixed detectability of δ = 5 units)					
Sampling Occasions	**Replicates[a]**	**Total Samples[a]**	**Standard Deviation (Sd)**	**δ/sd[b]**	**Total cost**
10	2	20	8.00	0.63	\$28,150
8	3	24	8.80	0.57	\$29,550
12	1	12	7.20	0.69	\$24,750

Notes: [a]Replicates were calculated using the total cost equation for the fixed budget, which yielded replicates per sampling occasion, generally not an integral number. In reality replicates would be randomly assigned to sampling occasions to produce the approximate specified allocation between occasions and replicates.
[b]From Figure 5-3 for 0.8 statistical power.

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Paired Watershed Study Design

Description of the Approach

The paired watershed approach is applicable to the assessment of both wet weather effects and technology performance and represents a means of connecting the two. The basic approach requires at least two watersheds, control and treatment, and two periods of monitoring, calibration and treatment. A control watershed is one that experiences essentially no change during both monitoring periods. It is subject to year-to-year or seasonal variations in large-scale factors, such as meteorological changes and natural biological cycles, that are beyond the control of study personnel. A treatment watershed is one in which a planned change (the "treatment") has been imposed between the two monitoring periods. This change can consist of a land development project that could potentially affect an aquatic ecosystem or application of a source control, or a structural best management practice (BMP) that could mitigate a negative effect. One control watershed can serve as a basis of comparison for treatments in different watersheds, and both types of watersheds can be replicated if desired.

During the calibration period, the two types of watersheds are treated identically, and paired data are collected. The basis (and implicit assumption) of the paired watershed approach is that these data represent a quantifiable relationship between the two types of watersheds and that this relationship is valid until a major change is made in the treatment watershed(s). The relationship is expressed as a linear regression of a variable measured in the (future) treatment case on the variable measured in the control case. It is further presumed that a new relationship will be established after application of the treatment and that this relationship can be quantified through post-treatment monitoring. The difference between the two relationships, if demonstrable through statistical analysis, will constitute a measure of the treatment effect (e.g., the impact of the development or the effectiveness of the management technique). This protocol is derived from work sponsored by the U.S. Environmental Protection Agency Office of Wetlands, Oceans, and Watersheds and performed by the Rural Clean Water Program at North Carolina State University (Clausen and Spooner 1993).

Advantages and Disadvantages

It should be noted that the data for the paired watersheds do not have to be statistically the same during the calibration period. Rather, it is the relationship between the paired observations that should remain the same over time, except under the influence of the treatment. Often, in fact, the paired data sets differ considerably. This difference, which is virtually inescapable in environmental systems, substantiates the value of a paired watershed approach: the technique does not assume initial equivalence in the two situations subject to comparison, which would rarely occur in reality; it does, however, assume a predictable relationship between the two.

The paired watershed approach has several other advantages besides avoiding the need to find systems that are initially similar in all important respects. Naturally variable factors such as weather are statistically controlled over the years of study, so that observed change can be attributed to the treatment in a cause-and-effect fashion. There is no need to measure all factors that could conceivably cause change, since their effects are embedded in the relationship derived during calibration. Also, the study can be completed in a shorter time than is generally possible in trend studies.

At the same time, the approach has some disadvantages and limitations. Response to the treatment is likely to be gradual, extending the length and cost of the study. Moreover, it is vulnerable to out-of-the-ordinary events like floods. Although, as mentioned above, paired watersheds do not have to be initially identical, effective application of the procedure generally requires that they be similar and in close proximity. The results will be compromised if the control watershed changes significantly during the course of the study.

Some statistical problems can arise, but they are usually avoidable if recognized and ameliorated by the monitoring program design. If, for example, the calibration period is too short, serially correlated data can result, meaning that successive observations are not independent of one another. This autocorrelation tends to increase variance and thus affect the number of observations needed to detect a difference (Gilbert, 1987; Ott, 1995). However, it is a more limiting factor in trend studies than in the demonstration of a differing relationship before and after treatment. Generally, it can be overcome by extending the calibration period over a full season, a year, or longer, depending on the objectives of the study. Another problem can occur if the treatment effect is strong enough to cause variances between the calibration and treatment periods to be unequal. Unequal variances violate the underlying assumption of the analysis of variance (ANOVA) procedure, which is applied to demonstrate the difference between the two relationships. Fortunately, ANOVA is robust, even with considerably heterogeneity of variances, so long as sample numbers are nearly equal (Zar, 1998).

Example Applications

Following are some applications for a paired watershed study design.

- Response of aquatic biota to watershed development during construction, after construction, or both;
- Comparison of sediment transport and deposition with two construction phase erosion and sediment control strategies;
- Detectability of a lawn pesticide in water and sediments with and without homeowner education;
- Storm peak flows, discharge volumes, and stream bed incision with advanced versus conventional runoff retention/detention;
- Water pollutant baseflow and storm event mean concentrations downstream of a constructed wetland with urban runoff treatment compared to downstream of an urban area without treatment;
- Fish presence and abundance downstream of a constructed wetland compared to the situation without treatment;
- Phosphorus loadings to a lake from a development with extensive roof runoff infiltration versus loadings from a development with piped roof drainage;
- Flow quantity, water pollutants, and benthic invertebrate community measures in a stream draining a watershed with state-of-the-art source and treatment controls compared to a stream draining a watershed with the legal minimum stormwater management;
- Fecal coliform concentrations in shellfish tissue from marine bays receiving flow from watersheds with and without intensive animal waste management efforts; and
- Rapid bioassessment attributes of a stream with a wide, continuous, naturally vegetated riparian zone compared to one with a narrow, disrupted, poorly vegetated riparian area.

Procedure

Monitoring Program Objectives

As with all monitoring programs, the development of objectives to guide the program design is the essential first step. For paired watershed studies, the requirements for effective utilization of the procedure, as well as its limitations as set out in this protocol, have to be recognized and formulated objectively. Once developed and refined, objectives should be used as in any other monitoring program to specify the program elements.

Watershed Selection

It is recommended that paired watersheds be selected to:

1. Be initially similar in physiographic and biological features, such as size, general morphology, slope, location, soils, and land cover;
2. Be similar in past, present, and future human influence, except for the treatment being tested;

3. Be in a steady state at the outset of the study, meaning that they have not experienced substantial change over a number of years prior to the study;
4. Be small enough to make uniform treatment throughout the treatment watershed possible; and
5. Have a stable channel at the measurement point, especially for flow monitoring.

Frequently, circumstances (e.g., a development proposal that has been approved for a specific plot of land) will dictate which watershed is to be the control and which one is to be the treatment location. If there is flexibility, any possible bias can be avoided by assigning control and treatment status randomly, for example by coin toss.

Calibration Period

Perform the monitoring program designed for the calibration period and obtain a data set of paired observations in the control and future treatment watersheds. Analyze the data as follows. It is most convenient to use a computerized statistical analysis package for most of the calculations.

1. Test to determine whether or not the data are normally distributed using a procedure such as the Schapiro-Wilk test (Zar, 1998). If the distribution is not normal, logarithmically transform the data and test for normality again. Most water quality and hydrologic data associated with stormwater runoff have been found in previous investigations to be log-normally distributed (Novotny and Olem, 1994). In the absence of testing, this distribution is usually a safe assumption for this type of data.
2. Using the log-transformed data, test for the equality of variances between watersheds using the F-test (Zar, 1998).
3. Examine residual plots to check for independence of errors (Zar, 1998).
4. Derive regression equations for each measurement variable (e.g., flow, water quality concentration or mass loading, a biological variable) in the form:

   $T = b_0 + (b_1)(C) + e$

   where T and C are the logarithms of values of measurement variables for future treatment and control watersheds, respectively; b_0 and b_1 are regression coefficients representing intercept and slope, respectively; and e is the residual error.
5. Test the statistical significance of the regression relationships by ANOVA. The test assumes that regression residuals are normally distributed, have equal variances between treatments, and are independent, as tested in Steps 1-3. If a relationship is not significant, either additional calibration monitoring should be performed to attempt to derive a significant relationship, or the variable should be discarded in favor of another one with a significant relationship.
6. Test to determine if sufficient calibration sample numbers have been collected to detect a difference of a given size, should one occur during treatment.

a. Decide on the fraction, f, of the mean value of the measurement variable during the calibration period that should be detectable after treatment. Base the selection on the objectives of the study, experience, and feasibility. The smaller the desired detectable difference, the more samples will be required in both calibration and treatment periods.

   For example, if a pollutant event mean concentration (EMC) in a stream is $EMC_1 = 30$ μg/L before treatment, and the goal is to reduce EMC to $EMC_2 = 10$ μg/L by installing a BMP, f = 0.67 represents the change that needs to be detected to determine whether or not the goal was achieved. There is no point in specifying a smaller f, at the cost of more sampling, for this objective.

b. Express the difference, d, that is to be detected in EMC_1 as $d = (f)(\log EMC_1)$, since the data have presumably been log-transformed.

c. Obtain the mean square residual variance, S_{yx}^2, from the regression significance test performed in Step 5.

d. Obtain the F-statistic from statistical tables. For n_1, the degrees of freedom for the numerator mean square, use (a −1), where a is the number of watersheds. If a control/treatment pair is being studied, (a − 1) = 1. If a control and two treatment watersheds are being investigated, (a − 1) = 2, etc. For n_2, the degrees of freedom for the denominator mean square, use $(n_1 + n_2 - a)$, where n_1 is the number of samples taken during the calibration period and n_2 is the number of samples to be taken during the treatment period (assumed at this point to be equal to n_1 for one treatment watershed, 2 x n_1 for two treatment watersheds, etc.). For a, the probability of the Type I error, it is most common to use 0.05.

e. Compute the ratio S_{yx}^2/d^2.

f. Compute the quantity: $[(n_1)(n_2)/(n_1 + n_2)]\{1/[F(1 + (F/(n_1 + n_2 - 2)))]\}$.

g. If the ratio computed in Step d is greater than the quantity computed in Step e, there are an insufficient number of samples to detect the specified difference. In that case, it is necessary to elect some combination of the following strategies: (1) specify a larger detectable difference, if consistent with objectives; (2) schedule more calibration period samples; and/or (3) schedule more treatment period samples.

7. Test to determine if residual errors about the regression are smaller than the expected BMP effect, which indicates how much deviation from the calibration regression is necessary for the treatment data to be significantly different.

Treatment Period

Perform the monitoring program designed for the treatment period and obtain a data set of paired observations in the control and treatment watersheds. Analyze the data as follows, again using a convenient statistical analysis package.

1. Derive new regression equations representing the treatment period for each measurement variable (e.g., flow, water quality concentration or mass loading, a biological variable) in the same form as for the calibration period.
2. Perform the same tests on the data as specified in Steps 1-3 for the calibration period.
3. Test the statistical significance of the treatment regression relationships by ANOVA. The test assumes that regression residuals are normally distributed, have equal variances between treatments, and are independent, as tested in Step 2. If a relationship is not significant, either additional treatment monitoring should be performed to attempt to derive a significant relationship, or the variable should be discarded in favor of another one with a significant relationship.
4. Perform an analysis of covariance (ANCOVA) comparing the calibration and treatment regression relationships. This analysis will demonstrate the significance of the differences between calibration and treatment regression equations overall and between their slopes and intercepts. The treatment effect is considered to be significant if these differences are significant, but insignificant (at least under the test conditions) otherwise.

Displaying and Interpreting Results

It is useful to graph deviations from expected values as if there were no treatment effect as a function of time during the treatment period. There is often interest in expressing the percentage difference in mean values with and without treatment, especially to express the effectiveness of a BMP. The analyses should be performed as follows:

1. Compute expected values without treatment from the calibration regressions.
2. Subtract the values from Step 1 from the observed values in the treatment watershed.
3. Plot versus time, obtaining a graph that visually illustrates the trend created by the treatment effect.
4. Compute the means of expected values without treatment, found in Step 1, and the means of observed values in the treatment watershed. Then find the percentage increase or decrease compared to the expected mean represented by the observed mean. It is not appropriate to make this calculation based on the observed values in the control watershed during either the treatment or calibration period, because generally, the control and treatment watersheds are not equivalent, even without the treatment effect.

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CHAPTER 6

Physical and Chemical Monitoring

Physical and Chemical Monitoring

Flow Monitoring

Introduction

Flow, or discharge, is a basic hydraulic characteristic affecting morphological development of stream channels, flooding behavior, bed and bank erosion, and sediment deposition. It therefore is a principal governing factor in habitat development. Furthermore, flow directly affects aquatic organisms through its velocity, against which fish must swim and non-motile organisms maintain their attachment. Flow is expressed in terms of its instantaneous rate, volume per unit time in units such as meters/second (m/s) and feet/second (ft/s), and the total volume over a designated period of time.

Measurements of flow rate, volume, or both are needed for many purposes in urban water resources work. Hydrologic studies require these data to determine the generation of surface runoff in response to precipitation. Water quality investigations need flow measurements to estimate pollutant mass loading (mass per unit time), the product of pollutant concentration and flow rate. Biological tasks sometimes need flow information for purposes such as assessing if minimum instream flows required for biota are being provided.

Performing these monitoring tasks can require flow measurement in controlled or uncontrolled open channels. In hydraulics, an "open channel" is any conveyance where flow is not constrained or under pressure. Therefore, closed pipes and culverts are open channels if they are not flowing full, a normal situation in runoff conveyance systems. When the geometry is regular and absolutely stable, the conveyance is termed "controlled," as in pipes, culverts, many lined ditches, and channels where a weir or flume can be installed. Otherwise, the channel is "uncontrolled," the situation usually found in natural streams.

Controlled open channel flow monitoring is performed using some type of flow meter. Flow meters actually detect stage and convert that reading to flow rate using an equation. In controlled flow, the equation is a standard formulation for the primary flow control device.

Flow meters are also used in long-term gauging of uncontrolled channels, if there is a way to relate stage sensed by the meter to discharge. In streams and rivers, the relationship is generally, according to a stage-discharge relationship developed at a location, expected to remain stable. If a continuous gauging record is not needed to achieve the monitoring objectives, a staff gage in conjunction with a stage-discharge relationship provides the observer with a flow rate estimate. In smaller conveyances, discharge is usually estimated from stage readings using the standard equation of open channel flow, Manning's equation. For short-term monitoring and developing a stage-discharge relationship in uncontrolled channels, the options are measuring with a current meter or a tracer.

There are also situations in urban water resources monitoring programs where pipes are submerged or flow full; in other words, they are not open channels. In these situations, which are often termed surcharged pipes, different equipment and measures must be used. While many types of flow meters exist for these pressurized flows, the most appropriate ones for situations encountered in urban water resources monitoring are ultrasonic Doppler and electromagnetic devices.

Flow Surveys in Uncontrolled Open Channels

Current Meter Method

Current meters measure flow velocity. Their use involves measurements in a number of segments across the channel and at one or two depths, depending on the total depth. Flow rate can then be estimated as the product of velocity and segment cross-sectional area and summed over all segments. A series of such determinations over time allows approximation of total flow volume during the period. The current meter method is most appropriate in natural streams and other relatively wide-open channels that can be divided into a number of segments. The technique is less accurate in very narrow channels that cannot be so subdivided, because the banks create edge effects that exert disproportionate influence. If narrow channels cannot be controlled using a weir or flume, flow rate can be estimated using Manning's equation (covered under *Using Flow Meters in Controlled and Uncontrolled Open Channels*).

Relating flow rate estimates over a range of flow conditions to water surface elevation produces a stage-discharge relationship. This relationship then allows calibrating a flow meter for continuous gauging or a staff gage for non-automated readings. The problem is that it is difficult to perform enough current meter surveys to generate a complete stage-discharge relationship, particularly with the difficult working conditions and relative rarity of high-flow events.

Newer current meters are digital, while mechanical meters are still in use too. Whatever current meter is selected, it should be able to measure velocities down to 0.03 m/s (0.1 ft/s) in depths as little as 0.1 m (0.3 ft), preferably less. All meters must be recalibrated at least once a year.

Mechanical current meters are simple and durable instruments. Some newer mechanical meters measure stage and velocity simultaneously, eliminating the need for calculating a stage-discharge relationship (Burton and Pitt, 2002). Direct-reading, digital instruments automatically calculate flow rate in the segment using mean velocity at each measurement point and the segment cross-sectional area of the subsection. The most advanced digital current meters use Doppler measurements of sonic pulses reflected as sound waves from particles in the water moving toward the meter. These meters have been more expensive and less durable than alternative instruments but have improved in these respects recently (Burton and Pitt, 2002). Figure 6-1 pictures a propeller-type instrument with digital readout. Refer to Appendix A to this chapter for the recommended measuring procedure using a current meter.

Proper site selection improves the accuracy of flow measurements at all discharge levels. Consider the following criteria when establishing a discharge measurement station. However, all criteria listed can rarely be met. Be aware of the site's limitations and possible effects on measurement. The station should be located in a channel reach (i.e., longitudinal section) with the following characteristics:

Figure 6-1: Turbo-Propeller Digital Current Meter

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- Generally, the channel should be straight for 100 m (328 ft) upstream and downstream of the measuring location; for smaller streams of only a few meters in width, the straight section should be at least 20 times the width upstream and downstream.
- Flow should be confined to one channel at all discharge stages (i.e., the channel should contain no surface or subsurface bypasses).
- The bed should be subject to minimal scour and relatively free of plant growth.
- Banks should be stable, high enough to contain maximum flows to be measured, and free of brush.
- The station should be located at a sufficient distance upstream so that flow from tributaries and tides does not affect stage-discharge measurements.
- All discharge stages should be measurable within the reach, but it is not necessary to measure low and high flows at the same cross section within the reach.
- The site should be readily and safely accessible.

The specific cross section in which a station is located within a channel reach should have the following characteristics:

- Banks should be relatively high and stable.
- The channel should be straight with parallel banks.
- Depth and velocity must meet minimum requirements for the method and instruments used.
- The bed should be relatively uniform, with minimal boulders and without heavy aquatic growth.
- Flow should be uniform and free of eddies, slack water, and excessive turbulence.
- Sites should not be located downstream of areas with rapid changes in stage or velocity.

Tracer Methods

Tracers include biodegradable, non-toxic, fluorescent dyes and salts that are detectable by photometric and conductometric measurements, respectively. Rhodamine WT fluorescent dye has been a common choice, because it has a lower detection limit, is less toxic, has lower sorption to particles, and decays more slowly than other options (Burton and Pitt, 2002). Flow rate is calculated from the tracer's travel time or degree of dilution.

Although tracer surveys can be less convenient and more time-consuming in natural waters compared to current meter methods, they are more precise (Burton and Pitt, 2002). Tracers can be indispensable in shallow streams, especially those with irregular bottoms where traditional current meters are difficult or impossible to use. Other applications of tracers are the measurement of transport and diffusion of discharge into receiving waters and the determination of retention time.

Flow Meters in Controlled and Uncontrolled Open Channels

Where the water surface in an open channel is perfectly parallel to the channel bottom, the flow is termed "normal." A primary control device, such as a weir or flume, is usually needed to produce normal flow. Where it exists, an automatic, recording flow meter programmed with a standard weir or flume equation can be used to register flow rates and volume. Otherwise, the channel is uncontrolled, and the water surface has an irregular profile. In this situation, using a flow meter requires programming into the meter a stage-discharge relationship derived either from a current meter or tracer survey or, more commonly, Manning's equation.

Flow meters detect stage in several different ways. Most common in urban water resources monitoring are meters that sense depth by releasing a regularly spaced stream of air bubbles at the channel invert and detecting the back pressure resisting the bubble release, which varies with depth as a consequence of the static head of water (Figure 6-2). This type of meter is relatively easy to use and is usually not affected by wind, turbulence, foam, air temperature gradients, or drying between events. However, it is susceptible to error when current velocity exceeds 1.5 to 1.8 m/s (5 to 6 ft/second), a result of the Bernoulli effect of pressure drop around an obstruction in high velocity flow. Also, this meter should not be used when the channel bottom slope exceeds 5-7 percent.

Three other methods of sensing depth are also in fairly frequent use (see Figure 6-2): (1) the shaft encoder, a counter-weighted float on a pulley that sends an electronic signal to a data logger; (2) the pressure transducer, which converts static pressure to an electronic signal transmitted to a data logger, and (3) the ultrasonic sensor. These flow meters can also control an automatic sampler to collect flow-proportional composite samples. The shaft encoder requires a standpipe housing removing it from the influence of velocity. Pressure transducers can be upset by contaminants, drying between events, and sudden temperature changes. A number of agents can interfere with an ultrasonic sensor, including surface-fouling materials or organisms, wind, noise, turbulence, foam, and air temperature gradients. Compensation routines and shifting from sound to the electromagnetic spectrum can alleviate some of these problems. All these options should only be used in preference to a bubbler-type meter where these interferences are absent or can be countered in some way. They do offer possible alternatives for high velocity flows.

In non-normal flow, Manning's equation is usually employed to estimate flow rates from a stage record, in preference to developing a stage-discharge relationship with current meter measurements. As pointed out earlier, covering the full flow range and accuracy in relatively small channels is problematic with current meters. The fundamental form of Manning's equation is:

$$Q = A\,R^{0.67} s^{0.5}/n$$

where: Q = Flow rate (m^3/s);

A = Channel cross-sectional area (m^2);

R = Hydraulic radius (m) = A/wetted perimeter;

s = Water surface slope (m/m); and

n = Manning's roughness coefficient (dimensionless).

In the English system of units, a multiplier of 1.49 on the right side of the equation gives Q in ft^3/s if geometric variables are in ft or ft^2:

$$Q = 1.49\,A\,R^{0.67}\,s^{0.5}/n$$

Because of uncertainties in estimating slope, depth, and n, the latter of which comes from textbook tables, the accuracy obtained from using Manning's equation is not as great as with a weir or flume. In addition, care should be taken when using the equation for low-flow events, when channel roughness, and therefore n, have more influence than in higher flow. When using Manning's equation as the basis for flow rate determination, place the selected stage sensor in the channel where:

- The cross section is uniform;
- The slope and roughness are constant;
- The channel is free of rapids, bends, abrupt falls, contractions, expansions, and backwater; and
- The channel is straight for at least 60 m (200 ft) upstream.

Because of uncertainties in estimates of slope, depth, and roughness, flow rate determinations using Manning's equation can lack accuracy, especially in an irregular geometry like a natural stream. Also, using Manning's equation is best avoided when measuring flow in channels that convey high solids loads, generally flow quite shallow, or both.

The options are to use a weir or flume, preferably, or to calibrate a stage-discharge relationship from current readings. These options are easier if the conveyance is above ground. If the flow in question is confined in a

Figure 6-2: Bubble Flow Meter, Shaft Encoder, and Pressure Transducer

Bubbler Flow Meter

Shaft Encoder

Pressure Transducer

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pipe or culvert, it may be possible to obtain a flume that can be inserted in an existing manhole or, more expensively, to install a plastic manhole that contains a flume. These devices are made in a number of different flume sizes and types.

Weirs and flumes are devices designed to establish a predictable and accurate relationship between flow and stage by controlling hydraulic conditions at the measurement point. Basic textbooks on fluid mechanics and hydraulics present standard equations to convert stage readings to flow rate. They also cover control device installation and other conditions that must exist to maximize accuracy.

A weir is a planar object, usually a vertical plate, built across the channel so that water flows over the top edge or through a regular opening (notch). The three most common opening geometries are rectangular, trapezoidal (known as a Cipoletti weir), and triangular (V-notch) (Figure 6-3). Each type has a unique discharge equation linking flow to water height above the notch low point, which is programmed into the software of the flow meter. Weirs are easily fabricated from inexpensive materials and can be used in an irregularly shaped channel, a situation in which a standard equation would be difficult to apply reliably. On the other hand, weirs can retain sediments that alter the environment, and they can cause flooding and overflow past the notch with higher than expected flow, invalidating the weir equation. Also, free fall of water over the weir is a prerequisite for validity. Weirs are often inappropriate in a natural stream because of potentially negative biological effects, especially blockage of fish movements.

A flume is a specially built reach of channel, sometimes a prefabricated insert, that has a converging section, a throat, and a diverging exit section. The flume's area, slope, or both are designed to differ from those of the channel, inducing velocity increase and corresponding water surface level change. Flumes are less subject than weirs to problems with sediment deposition, flooding, and overflow. Several flume configurations are available off the shelf, with the H flume being the most common because of proven performance, relatively low cost, and a wide range of relatively accurate measurement capability at flows commonly experienced in urban water monitoring (approximately 0.01 to 1 m^3/s = 0.35 to 35 ft^3/s). H flumes (Figure 6-3) are available from suppliers for typically encountered ranges. Designs are also available for smaller flows (HS flume) and larger ones (HL flume). Like weirs, each flume type has a characteristic discharge equation that can be programmed in the flow meter. It is usually impossible or inadvisable to mount a flume in a natural stream.

Surcharged Pipe Instrumentation

Ultrasonic Doppler velocity sensors measure the shift in frequency of waves reflecting off particles in the flow and convert the measurements to an estimate of the average particle velocity. High or changing solids concentrations, air bubbles, and foulants can interfere with them.

Electromagnetic velocity sensors operate under Faraday's principle, in which a conductor (water) moving through an electromagnetic field generates a voltage proportional to the velocity. They are less subject to the problems affecting ultrasonic meters but can be upset by electrical noise.

Figure 6-3: V-Notch Weir and H-flume (with Instrument Shelter)

V-Notch Weir

H-flume (with Instrument Shelter)

There is no alternative to using one of these instruments in a surcharged case. They can also be used in open channel flow and are generally more accurate than flow meters relying on a stage-discharge relationship. However, their purchase cost is higher.

Water Quality Monitoring

Introduction

The essential tasks in sampling natural waters and runoff are to obtain a sample that properly represents the water of interest according to the program objectives, and to prevent its deterioration and contamination before and during analysis. These tasks break down into sample collection, sample handling, sample analysis, and quality assurance/quality control (QA/QC). Sample collection, in turn, involves considerations of what, where, and when to sample; how many samples to take; and how to sample. Thoughtful, thorough planning and performance of these steps should produce representative samples and fulfil the objectives. The following section outlines how to organize these tasks. At the end, it also covers data analysis.

In preparing for sampling, a good, helpful laboratory can save a lot of work and ensure that field personnel are properly equipped to take valid samples in return for the business. Many states now certify laboratories for water quality analysis. A lab should be chosen based on experience or trusted recommendations. When preparing for sampling, ask the laboratory to provide the proper sample containers for the analyses they will perform and (of crucial importance) to clean those containers as designated in U.S. Environmental Protection Agency (1983) and American Public Health Association (1998) procedures to avoid contamination that will invalidate the sample. Obtain one or more coolers that will be adequate in size to transport all samples on ice after collection until they reach the laboratory.

A single container can typically be used to hold samples that will be analyzed for several variables with compatible preservatives. For example, conductivity, pH, total suspended solids, and turbidity analyses can usually be performed on samples from one container, and all nutrient analyses can usually be performed on samples from a second container. Appendix B specifies sample containers for common water quality variables as well as other information that will be considered later.

Preparations should also consider QA/QC, covered in detail below. At the preparation stage, it is necessary to pick up extra containers for field replicates and field blanks. A field replicate is a repeated sample, taken at exactly the same spot in exactly the same way, immediately after the primary sample. The general rule is to select randomly 5 to 10 percent of samples for field replication. The random selection can be made by assigning each sampling location and occasion an identifying number and then using a random number generator on a calculator to pick the 5 to 10 percent to be replicated. A field blank is simply a container of distilled water that is carried into the field and returned to the laboratory without disturbance. Its purpose is to indicate if transport has introduced contamination to samples. The field blank should be part of the lab's standard QA/QC procedures for pathogen samples and sometimes for nutrient work.

Sampling personnel should give close attention to safety considerations. Some key ones are:

- Do not allow effluents, contaminated receiving waters, sharp underwater objects, or chemical reagents to contact skin; use rubber boots and gloves.
- Do not enter confined spaces, which may have inadequate air flow and concentrated harmful gases. If the objectives require sampling in such areas, obtain the services of a crew with special training and all of the right equipment.
- Use a proper tool to remove manhole covers, and never leave an open manhole unattended.
- Wear a hard hat if there is any possibility of falling objects.
- Wear a reflective vest if there is traffic near the sampling area, and set up rubber traffic cones if necessary to divert vehicles far enough away.
- When sampling, do not enter a channel with a velocity greater than 75 cm/s (2.5 ft/s) or deeper than waist height; sample from a bank or bridge instead. Have a safety rope ready in all cases when personnel enter the water.

Sample Collection

What and Where to Sample

Once monitoring objectives are well defined, what and where to sample are fairly straightforward considerations; one samples the water body or runoff stream at the place or places where information can answer the questions represented by the objectives. The principal consideration in this regard comes up when paired sampling is performed. As pointed out in Chapter 5, pairing monitoring stations is advantageous in reducing or eliminating the confounding effects of variability on interpreting results. Paired stations can be on different water bodies or different points on the same water body, one affected and one unaffected by a certain condition. Paired stations must be selected carefully to be as similar as possible in all respects except the effect being studied. Refer to the appendix to Chapter 5 for guidance in station selection and other aspects of paired monitoring program designs.

When to Sample

Deciding when to sample is a key and intricate consideration in obtaining representative samples that will serve the defined monitoring program objectives. Natural water bodies and stormwater runoff experience substantial variability over time. This variability must be accounted for to get reliable answers to the questions the program sets out to answer. The chief sources of temporal variability over extended time spans are seasonal changes and stochastically varying meteorological events. Over shorter time intervals, variability is a function of such phenomena as diurnal light and temperature fluctuations and differing precipitation intensity during the course of storms.

To make good decisions about when to sample, there is simply no substitute for working out beforehand what one wants to learn and, following this, what conditions must be observed to gain the desired knowledge. In most cases, an appropriate sampling schedule will not be uniform over time, since the events creating the conditions of interest are very rarely uniform either. It will therefore almost always be necessary to emphasize certain periods over others to accomplish the objectives within cost limitations. In a stormwater runoff study, for instance, emphasis might be placed on times of highest runoff, when pollutant delivery is greatest, and of lowest flow, when pollutants concentrate most.

Since stochastic meteorological events drive many of the cases of interest in urban water resources monitoring, randomly selecting sampling occasions should be seriously considered. For estimating total suspended solids concentrations and mass loadings, Leecaster, Schiff, and Tiefenthaler (2002) compared two random sampling program designs versus three schemes stratified by season or storm size. They found that simple random sampling of all storms or of medium and large storms had the lowest standard error and the least bias in estimating concentrations, although these designs yielded no advantage in loading estimation.

While randomization can be appropriate in many situations, the program may be best served by having some limiting criteria. For example, it would not be a good use of resources to mobilize for sampling every storm, when some would not have enough precipitation to produce runoff, and some would come with a very short dry period since the preceding runoff. The best course could be to deviate from strict randomization to attain an emphasis that best serves objectives. This strategy would be a stratified random design. For the example of emphasizing periods of highest and lowest runoff, the stratification would allocate more samples to each of these intervals than to times of intermediate flows. The storms within each period would then be selected randomly.

In general, then, it is a practical necessity to have weather forecasts to target the most productive sampling times for the given objectives and anticipate their start. Weather service or university websites, an independent weather forecasting consultant, or some combination of these can be used for this purpose. Since it is very difficult to predict accurately the depth, intensity, and duration of rainfall, it is recommended that the monitoring team be prepared to work during any storm that has a high probability of generating the amount and pattern of rainfall designated for sampling.

Criteria for storms that will be targeted for sampling are commonly established according to the following factors: minimums in rainfall quantity anticipated to be necessary to produce enough runoff to sample, storm duration, and antecedent period without measurable rain. How these criteria should be set numerically depends on local experience, hydrologic modeling, or both. Somewhat typical criteria, as minimums, are 0.15 to 0.25 inch (4 to 6 mm) of expected rain, 1 to 3 hours in duration, and 48 to 72 preceding hours without measurable rain.

A related issue is how long to continue monitoring. An outer limit for many programs relying on sample composites over time is 24 hours, because maximum holding times prescribed for some constituents in water will be exceeded if processing does not start shortly beyond that point. If the quantities of interest are relatively stabile, though, the time can be extended to get more complete coverage of a hydrograph. The minimum sampling time is best judged in terms of collecting a minimum number of aliquots in a composite and covering a designated minimum proportion of a total hydrograph before flow returns to the pre-existing condition. Typical criteria are at least 8 to 12 aliquots and 75 to 90 percent of the hydrograph.

How Many Samples to Take

Another issue that affects representativeness in sampling is the question on how many different occasions to take samples at each site. This decision is highly influenced by the available budget as well as by the objectives. Making the decision is a classic use for the statistically based methods for choosing sample numbers presented in Chapter 5.

The main difficulty in using these statistical methods concerns the pilot data set, which must be available to define variability. Site-specific data are often not available, and even reliable data from a similar setting may be lacking. Therefore, the temptation to use more arbitrary rules to select sample numbers is strong. Professional judgment based on extensive local experience is the best fallback if there are no suitable pilot data. Site- and case-specific factors have a strong bearing on the sample size necessary to meet the program's objectives.

Thompson et al. (1997) randomly selected from historical highway runoff records to create alternative test sequences of runoff events. They calculated mean concentrations of total suspended solids, total dissolved solids, total organic carbon, and zinc, as well as their 95 percent confidence intervals. The researchers compared results from these test monitoring programs with the actual sequences to see how quickly the sample mean approached the mean of the population established historically. They found that estimates of the means became approximately constant after 20 samples and that variances also stabilized. While Thompson et al. (1997) concluded that approximately 15 to 20 samples are required to provide reasonable mean concentration estimates of these water quality variables, they also cautioned about the possible influence of factors like seasonality on the numbers and allocation of samples.

How to Sample

General Considerations

Water can be collected manually or with automatic samplers in several ways, each with advantages and disadvantages:

- Grab samples – collected once per sampling occasion at a location, usually manually;
- Discrete samples – collected at a series of specific points in time at a location;

Figure 6-4: Multiple-Variable Probe and Data Logger

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- Composite samples – made up by combining a number of samples taken at different times or locations (vertically or horizontally distributed) to represent a time interval or a spatial area; and
- Continuous samples – made up by collecting a fraction of all passing flow to produce an uninterrupted composite sample.

Grab sampling is low-cost but tends not to be very representative because of the temporal and spatial variability usually associated with urban water resources and stormwater runoff. Thorough coverage with discrete samples provides the most complete picture of water quality but creates a large and often unaffordable analytical burden.

Continuous sampling is entirely feasible, using single- or multiple-purpose electronic probes and associated recording instrumentation (termed datasondes), for a number of variables: dissolved oxygen, temperature, pH, conductivity, salinity, total dissolved solids, and oxidation-reduction potential (Figure 6-4). For other variables, continuous sampling is problematic because many substances are impossible to detect electronically at environmental concentrations. Samples must usually be accumulated in an automatic sampler, retrieved, and delivered to a laboratory for analysis. This rather labor-intensive process might be manageable for variables with relatively long allowed holding times, but it is usually impracticable for those that must be filtered, analyzed, or both while very fresh. In these cases, which represent much of the monitoring of urban water resources and runoff, the best strategy is often to take a composite sample over a period of hours and repeat it on other occasions to form a representative database that meets objectives.

With composite sampling being a mainstay in urban water monitoring, the basis for compositing is an important question. Given the temporal variability that usually occurs, collecting aliquots for the composite at equal time intervals is generally not representative. A much better basis is compositing in proportion to flow; i.e., weighting individual samples in the composite in direct proportion to how much of the total flow they represent. For this purpose, usually a flow meter is used in conjunction with an automatic sampler, which triggers sample collection according to flow registration. The section *Automatic Samplers: General Considerations* below further discusses flow-proportional compositing. If there is a flow record, it is possible to produce flow-proportional composites manually. Then flow-proportional subsample volumes are extracted from samples previously taken at equal time intervals and combined.

Whether water samples are taken with manual or automatic samplers, care should be taken to select equipment that does not change the characteristics of the sample through contact with parts made of contaminating materials. Modern automatic water samplers use Teflon, stainless steel, and non-reactive plastics in tubing and other parts that come in contact with sample water.

An important consideration in sampling is to obtain sufficient quantity for the anticipated analyses. In manual sampling, it is normally easy to collect more than enough volume. Quantity becomes more of an issue in setting up automatic samplers, both to represent the event well and collect sufficient volume. The section *Automatic Samplers: Programming Considerations* below covers how to resolve this issue. Appendix B gives amounts required for common analyses. It is always a good idea to collect excess sample volume, if possible, to allow for rinsing instrument sensors with the sample itself, replicating analyses for QA/QC purposes, and re-analyzing if QA/QC criteria are violated. The best rule is to collect 2.5 times the total recommended volume for all anticipated analyses, but actually obtaining that quantity may not be possible in automatic sampling.

In addition to the duties associated with collecting samples, personnel visiting sampling stations should always take copious field notes. These observations are often invaluable later in understanding and interpreting results. The records should include, as appropriate:

- Date;
- Time of sample collection or visit;
- Name(s) of sampling personnel;
- Weather and flow conditions preceding and during visit;
- Number and type of samples collected;
- Calibration results for field instrumentation;
- Field measurements;
- Log of photographs taken;
- Comments on the working condition of the sampling equipment;
- Deviations from sampling procedures; and
- Unusual conditions (e.g., water color or turbidity, presence of oil sheen, odors, and land disturbances).

Manual Sampling

While automatic samplers are now more used in urban water work, they are much more expensive than manual samplers, still require substantial operator attention, and cannot be used in all situations. Manual samplers are still the necessary or better choice for sampling lakes and rivers that are relatively wide, deep, or both. They also give the flexibility to cover more locations than automatic samplers, which require a considerable installation effort.

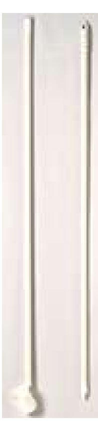
Most manual samplers are one of two types: simple dipper pails (Figure 6-5) or cylinders with open ends that are shut with a remotely operated messenger tripping closure lids. Several designs of the second type have long been on the market, two common ones being the Van Dorn (Figure 6-5) and the Kemmerer bottles. Samplers of this type are typically cylinders lowered into the water with both end closures held open. When the sampler reaches the desired depth (determined from a marked line attached to it), a messenger is dropped down the line to trip the closure mechanism. The sampler is then drained through a spigot into sample bottles. Sample bottle rinsing can be accomplished by overflowing two or three bottle volumes. In recent years, these standbys have been redesigned, using materials that will not contaminate samples intended for analysis of metals and organic chemicals. Manual pump samplers are also available, generally for use in fast currents from a bridge.

In the case of streams, rivers, ditches, and other channels, it usually must be assumed for practicality that relatively homogeneous conditions prevail over the width and depth dimensions of the water. Small systems are generally more homogeneous than large ones. As a result of this assumption, samples for water quality in flowing channels are usually collected in midstream and at one depth. In the absence of any special considerations, collection at half of full depth is recommended. Environmental conditions in channels can differ longitudinally and with changing flows. These conditions especially affect particle transport (see 'Special Considerations for Sampling Solids' below). Therefore, sampling programs often require multiple stations and sample collection in a range of flow conditions in dry and wet weather.

Most channel sampling is conducted on foot in shallow flows. When wading, the individual collecting samples should face upstream. This orientation minimizes contamination of the sampled water that would be caused by the sampler's presence. The container should enter the water with the opening down to minimize collection of material from the surface layer. Unless a preservative has been added to the sample bottle before collection, it should be rinsed with two or three volumes of water by filling and totally emptying the sample bottle several times before capping. Several dip samples should be collected and composited. If the protocol for for the intended analyte(s) calls for adding a preservative to the sample bottle, it should be tilted down just slightly from the horizontal and should not be rinsed.

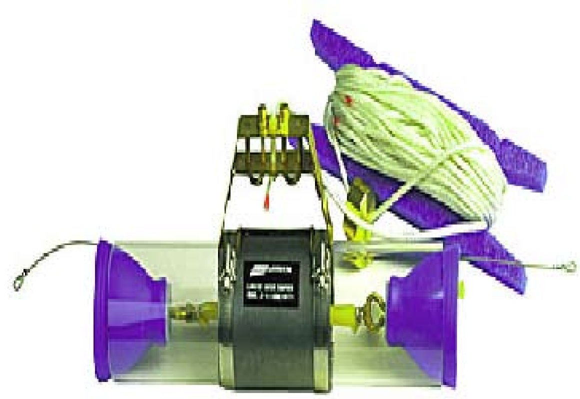
When sampling on foot is impossible or unsafe, some device must be lowered from overhead to reach the water. If a bridge spans a large channel, a Van Dorn

Figure 6-5: Polyethylene Dip Sampler, Swing Sampler, and Van Dorn Bottle



Polyethylene Dip Sampler

Swing Sampler



Van Dorn Bottle

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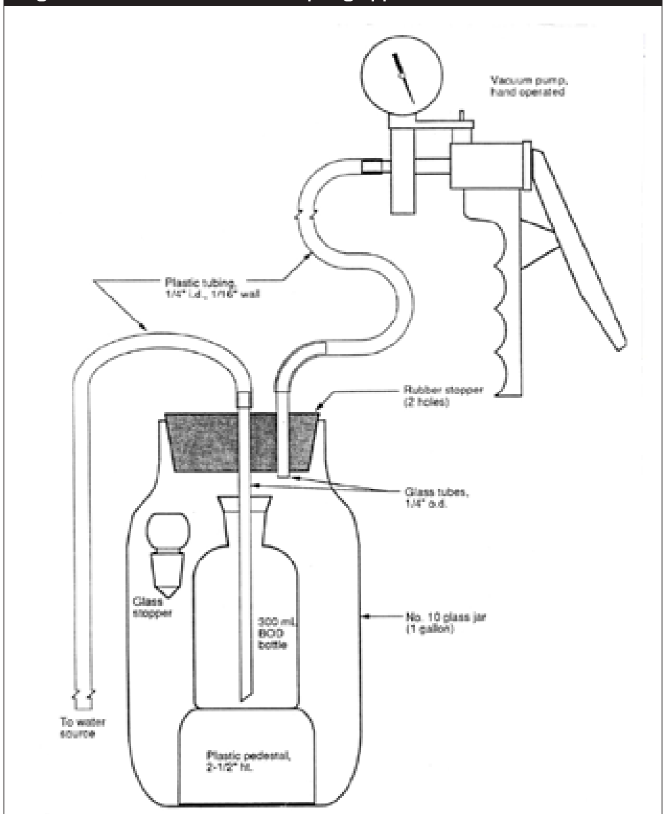
or similar sampler can be used if the current is not too great. In higher currents, a weighted, stainless steel bucket can be lowered on a line to a depth of 30 cm (1 ft) below the surface and then raised.

Other considerations prevail when sampling standing or tidal waters. When sampling a freshwater reservoir (e.g., a lake or wastewater lagoon) or relatively static estuary deeper than about 2 meters (6 ft), it is necessary to take into account the possibility of thermal stratification and the consequent variation in environmental conditions with depth. Depending on the program's objectives, samples might be drawn from the relatively uniform surface (epilimnion) or deep (hypolimnion) zones, in the transition area between them, or some combination of these. Sample locations will generally be accessed by boat, from which the sample can be taken with a Van Dorn or similar sampler.

Sampling in wetlands, shallow channels, and sheet flows is often complicated by shallow depth and patchy physical and vegetation structure. Sampling for dissolved oxygen analysis in shallow water without entraining atmospheric oxygen is especially difficult. The problem can be overcome by placing the sample bottle inside a larger stoppered bottle that is evacuated with a hand pump and then drawing the sample through a tube from the source to the sample bottle (Figure 6-6). In extremely shallow locations, it is best to use equipment like this for all water column sampling in order to avoid collecting sediments and organic debris from the bottom. Whether just one sample or several spatially distributed ones are collected, depends on the program's objectives and the water body's structure.

Any time the variable to be measured can easily exchange with the atmosphere, the sample bottle must be filled to overflowing, capped without trapping any air, and double-checked visually to be sure there are no air bubbles. The leading examples are dissolved oxygen and volatile organic compounds. The best practice, which should be followed whenever possible, is to cap the sample bottle under water. When this is impossible because of accessibility, it is acceptable to drain from a Van Dorn or similar sampler into the bottle that will be used for analysis, overflow it, and cap while water flows. It is never acceptable to pour from a sampler into a sample bottle for these analyses. In inaccessible locations where a messenger-activated sampler will not work, a special grab sampler allowing bottle opening and closure under water must be used.

Figure 6-6: Shallow Water Sampling Apparatus



Automatic Samplers: General Description

Automatic samplers offer a number of advantages over manual methods, as well as drawbacks associated with the expense of purchasing and installing them and the occasional unreliability of any electromechanical equipment. In any event, they are a necessity for many urban water monitoring programs. It is usually infeasible for human sampling personnel to function over extended flow events and produce a representative, flow-related composite sample or series of discrete samples. Attempting to do so still requires a flow meter, as well as expenditure of a significant portion of the capital and effort needed for a coordinated flow meter and automatic sampler setup.

While automatic samplers are indispensable in modern urban water monitoring programs, they must be installed and operated with care to produce data reliably. In most settings, they should be in secure housing to minimize the risk of damage and vandalism. Success usually requires experienced personnel for installation and maintenance when needed. Less experienced staff can be trained to program and attend the sampler and flow meter routinely. These operators must be at the sampling site frequently to check equipment operating condition, prepare for events, reprogram settings depending on expected flow conditions, change containers, and, of course, remove and handle samples.

Several manufacturers produce automatic samplers and compatible flow meters (Figure 6-7). While there is some variability in features, the most commonly used samplers have similar standard elements and capabilities. As mentioned above, modern samplers use materials for water-contacting parts that will not contaminate samples with metals or organics that may be subject to analysis. They can generally either deposit samples in a composite chamber or in multiple bottles arranged circularly in a base. Sampler base designs typically allow preserving samples by placing ice around them, and some manufacturers offer refrigerated units as an option. Samplers usually offer the hardware and software for flexible programming capability, allowing monitoring over a fairly wide range of conditions as well as data logging. They can be powered by line current with voltage step-down, 12-volt batteries, or, in many cases, solar panels. Many can utilize telemetry to reduce the need for human intervention and conveniently download data.

Automatic samplers employ vacuum suction to pull samples and have limitations on both the horizontal and vertical distances from which they can draw. While there is some difference among models, these limits are approximately 30 meters (100 ft) and 7 meters (23 ft), respectively. Beyond these distances, a submerged pump can be used to discharge samples into a container from which the automatic sampler draws.

There are a number of precautions in placing sampler intakes to avoid problems and unrepresentative samples. Both these risks can be reduced by situating intake so as to avoid sediments being scoured from the bottom and drawn into the line. The end of a flume for flow measurement, which will often be present, is an excellent spot for the sampler intake. If there is no flume, a good way to avoid sediment entrainment is by locating intakes on top of a small anchored piece of concrete. Water velocity should be above 100 cm/s (0.3 ft/s) to avoid the accumulation of particulates and ensure that all sediments are sampled (Burton and Pitt, 2002). Sampler intakes should not be located downstream of any treated wood structures whenever heavy metals or organic compounds are subject to analysis. Intakes in pipes should generally be elevated and in a turbulent zone to reduce blockage by debris. Pipe flow is often vertically stratified, and elevation tends to produce a more representative sample (Burton and Pitt, 2002).

A water level actuator, depth sensor, or rain gage can initiate sampling. Most commonly used is the flow meter's depth sensor, generally located just upstream of the sampling point in a control section (e.g., in a flume or behind a weir). The calibrated stage-discharge relationship provides flow measurements that determine, in conjunction with the programming, when collection should take place to provide a representative sample.

Automatic compositing can be done either on a time- or a flow-proportional basis. Time-based samples are drawn in equal volumes at equally spaced intervals, regardless of flow conditions. They do not closely rep-

Figure 6-7: Examples of Automatic Samplers

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resent the flow hydrograph and pollutant mass loading delivery, unless the time interval is relatively short. In its most common configuration, flow-proportional sampling weights the volume of a sample taken in proportion to the flow volume that has passed since the preceding sample. This method is the most convenient way of obtaining a sample representative of the overall event. In operation, the flow meter sends a signal to the sampler after a predetermined increment of flow to draw a predetermined volume of water.

Automatic samplers are now sometimes being used in conjunction with datasondes for continuous measurement of basic water quality variables (see listing above). For examples of using this combination of instruments, refer to Wilcock et al. (1995), St. Croix Watershed Research Station (1999), Baxter (2003,) Hall et al. (2004), and Buchanan (2006). Portable autoanalyzers are now coming on the market for potential use in association with automatic samplers for measuring other variables in the field. These instruments have long measured nutrients in the laboratory and are now coming into field service, especially in seawater monitoring applications. A portable incubator and microbial autoanalyzer is also available and could help in getting more representative bacteria data, given the problems in avoiding contamination with composite sampling and the consequent need to rely on grab samples.

Automatic Samplers: Programming Considerations

Thoughtful attention to programming will yield better data from automatic samplers. There will be less wasted effort in taking, and possibly expensively analyzing, samples that are not representative in terms of the monitoring program's objectives. The sampler must be programmed with: (1) the volume to collect each time an individual sample is drawn, and (2) the total number of samples to collect. The flow meter must be programmed with the flow pace, or flow quantity increment, i.e., the additional flow volume registered by the meter that will signal the samplers to draw water. The volume, number of samples, and flow pace must be balanced according to three considerations: (1) obtaining adequate quantity for laboratory analyses; (2) avoiding overfilling the sample containers, which results in unusable samples; and (3) sampling at points spaced sufficiently close to represent the runoff hydrograph relatively well. With faulty programming, small flow events can fail to produce enough quantity, and the hydrograph of large ones may not be well represented.

The more closely spaced in time the sampling points are, the more representative the sample will be. However, this consideration must be balanced with overfilling risk and, in case of battery power supply, the charge life relative to anticipated sampling duration. There is no certain way of specifying volume, number of samples, and flow pace at the outset of a new monitoring effort, but programming can be improved as experience with the site accrues. The best strategy, at least initially, is to use the smallest sample volume found to work well and, in case of compositing, the largest available container to add assurance against overfilling. Specifying flow pace and number of samples is usually hindered at the beginning by lack of information on flow volumes produced at the site by typical runoff events.

Common recommendation for the individual sample volume is in the vicinity of 100 mL. This quantity might be decreased to as little as perhaps 50 mL, if the list of analyses is relatively short, if there seems to be substantial overfilling risk, if there is a desire to sample at quite closely spaced points, or in case of some combination of these circumstances. It might be increased to perhaps 150 mL or even higher if there is concern about obtaining sufficient sample volume for analyses, and overfilling is not risky, so long as enough samples can be taken to represent the hydrograph well.

Deciding on the number of samples is best done with some analysis of expected flow patterns and the resulting hydrographs. The first thing to consider is how many samples the container can hold. For example, the commonly used 20-liter (5-gallon) carboy can hold 400 50-mL or 200 100-mL aliquots, which is not much of a limitation. Deep-cycle marine batteries are capable of drawing at least 300 samples before they need a recharge, which also does not pose much limitation in most cases. If the sampler were programmed to collect 200 samples for compositing over an anticipated 24-hour flow period, a sample could be collected every 7.2 minutes, offering excellent hydrograph coverage. However, a somewhat longer interval between samples would still give good coverage and at the same time leave carboy volume available for more insurance against overfilling. With experience, the setting can always be reconsidered for possible adjustment.

Flow pace is the total flow volume expected over a sampling period, divided by the number of samples. Obviously, volume will differ with conditions. Performing this programming step is easier if there is an existing flow record. The next best foundation is a good hydrologic model to forecast potential volume. In the absence of either of these assets, even a simple hydrologic model like the Rational Method can provide some basis. For example, the total runoff volume from a rainstorm can be approximated by multiplying the expected precipitation quantity, the area of the contributing catchment, and a runoff coefficient representing the catchment, along with appropriate conversion factors.

While good, strategic programming is the best way to balance the various considerations, there are some options if the circumstances create problems that make a balance impossible, or if the programming fails because of an unexpected occurrence. If somebody is at the sampling site or in touch by telemetry or anticipates a problem (e.g., overfilling), that person can either reprogram or change bottles or both. If the conflict between small flow event sample volume and large event hydrograph coverage cannot be resolved by programming alone, a possibility raised by Burton and Pitt (2002) is to substitute an enlarged container such as a Teflon-lined or stainless steel drum for the standard sample base. A smaller glass jar can then be suspended inside the large container to collect samples during relatively small events, while the overflow during bigger ones is collected in the large container.

Special Considerations for Sampling Solids

Solid particles are constituents of urban waters that are of interest both in their own right and as transport media for other pollutants. In aquatic monitoring programs, they are almost always represented by total suspended solids (TSS) analysis on a bulk sample intended for analyzing all constituents, dissolved or particulate. Dissolved substances are uniformly distributed in flowing and standing waters, but particles are often stratified vertically, horizontally, or both. Ideally, sampling would represent the actual distribution of particles. However, achieving this is likely to be complex and labor-intensive, making true representation infeasible in many routine monitoring programs. Still, with recognition of the issue, those designing and performing monitoring programs may often be able to avoid patently unrepresentative solids sampling and institute improvements whenever possible.

A number of factors are responsible for spatial stratification of particles (URS Greiner Woodward-Clyde, 1999). To start with, particles in water vary substantially in shape, size, and mass. These differences stem from watershed characteristics such as soils and topography, as well as from meteorological and hydraulic conditions like antecedent dry period, rainfall intensity, and flow rate and velocity. The conveyance (e.g., stream, pipe, ditch) represents another class of influences. The point in time of measurement (early in the flow event, during first flush, if there is one, versus later) also has a bearing.

Sediments larger than 60 μm in diameter are particularly susceptible to gravity and are not distributed evenly throughout the water column. In sinuous streams, particles can be distributed differently horizontally based on size, because of the differential velocity at the outside versus the inside of meanders. However, lighter particles, such as clays and silts, tend to be distributed more uniformly than larger ones (Burton and Pitt, 2002). Solids in these size fractions also generally have more environmental significance than the heavier particles, since they travel farther, affect more aquatic organisms, and collectively have much greater surface area for transporting other pollutants. Therefore, they are the implicit emphasis of many urban water monitoring programs, and the frequent stratification of larger particles is not such a large issue.

Even so, there are some sampling strategies that can reduce what stratification impact there may be on the program's representativeness. The easiest one is to orient the automatic sampler intake in the downstream direction. Neither manual nor automatic samples should be taken in conditions promoting solids stratification (e.g., at a tight bend in a stream or where there is a strong velocity differential with depth).

Several more burdensome strategies exist for consideration when either stratifying conditions cannot be avoided or the program's objectives depend on representing the relatively large solids well. One is to take multiple samples spatially distributed to cover the anticipated variability (vertical, horizontal, or both) and then composite them. This strategy is effectively limited to manual sampling, and it is difficult to get enough information on flow at multiple points to composite the samples proportionately. Isokinetic samplers (Figure 6-8) overcome the fundamental problem causing poor representation of large solids (namely, that sample intake is usually at lower velocity than the flow from which the sample is drawn) by pumping the sample at the same velocity as the flow. The higher the flow velocity, the more representative an isokinetic sample is likely to be. If the stratification is vertical, manual depth-integrating samplers (Figure 6-8) exist to represent the distribution. These samplers point directly into the flow; thus, the sample enters approximately isokinetically. Raising the sampler from the invert to the surface produces an integrated sample.

Water Sample Handling

The principal problem in any monitoring program is obtaining a sample that represents the conditions being investigated. When samples are taken from the field, they are removed from their original context and can undergo significant changes. Some of the key variables of common interest are particularly prone to alteration, including dissolved oxygen, metals and nutrient speciation and solubility, pathogens, and volatile organic compounds. The key to avoiding modification following sampling is to practice proper sample handling procedures. These procedures involve careful sample labeling and tracking, preserving samples as recommended, and beginning analytical processing within maximum holding times established for the respective water quality variables.

To avoid mistakes, label a sample bottle with an indelible marker before going into the field. The label should include:

- Sampling station;
- Date of collection (day/month/year);
- Time of collection (24-hour format, added in field);
- Name of person(s) performing sampling;
- Preservative added (if any); and
- Analyses to be performed.

Once the sample is collected, it should be capped with a lid also labeled with the station and date, as well as with the time if samples are taken over time. The bottle should then be sealed and the seal also marked with matching sample information. The seal should remain unbroken until the sample is ready to be opened for laboratory processing.

A tracking record for each sample registers possession as the sample travels from collection through analysis, making misplaced samples easier to find. Samples involved in litigation, especially, require formal chain-of-custody records. Analytical laboratories typically develop these forms appropriate to their services and supply them to customers.

Appendix B gives recommended preservation techniques and maximum holding times for common analytes. The specified maximum intervals between the time a sample is taken and the time it is analyzed vary widely for the different variables and constituents. Some key parameters must be determined in the field immediately after collection, including temperature, pH, dissolved oxygen, and chlorine residual. Other field determinations are, however, often less precise than laboratory analysis. Properly preserved samples can be carried back to the laboratory for replication of field analyses within 24 hours to add reliability. Once a sample reaches the laboratory, filtering for those analyses requiring it should be done within 24 hours. Samples should then be preserved at a temperature of 4°C, unless they are preserved with nitric acid for

Figure 6-8: Isokinetic Sampler and Depth-Integrating Sampler

Isokinetic Sampler

Depth-Integrating Sampler

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metals analysis. The principal references for more detail on sample handling as well as analysis are the American Public Health Association's (1998) *Standard Methods* (available by subscription on-line as of 2004) and the U.S. Environmental Protection Agency's (1983) methods manual.

Sample Analysis

Appendix C classifies into physical and chemical groups the many substances and characteristics that can be measured in water and sediments. The list represents hundreds of quantities, most among the organic chemicals, that could be analyzed. Clearly, judicious choice among all of these possibilities demands reliance on carefully developed objectives representing what one is trying to accomplish with the monitoring program. Among the information in Appendix B are the recommended analytical techniques for common analytes from the larger list. Excepting those few analyses that must be performed immediately in the field, most are laboratory procedures.

A leading issue in sample analysis is the ability to detect and numerically measure within defined bounds of certainty the quantities of interest. Relative to waste streams like industrial and municipal effluents, constituents of natural waters and even urban runoff are often present in low concentrations. These amounts could still be biologically significant, though, and thus the focus of monitoring program objectives.

There are a number of ways in which detection levels can be specified. But however the laboratory itself quantifies detectability, it should be capable of reliably giving results down to the reporting limits (RLs), the lowest concentration of a variable that can be reliably quantified within specified limits of precision and accuracy. Another approach to estimating limits is to use method detection limits (MDLs), the minimum that can be measured with 99 percent confidence that the concentration is above zero. Reliable RLs are often higher than MDLs. Laboratories frequently report values between MDL and RL but flag them to indicate uncertainty in the quantification. Appendix B gives RLs for common analytes as well as methods typically used for their measurement.

A key to achieving detectability in line with program objectives and overall good service is to select a laboratory carefully. Check a candidate laboratory's accreditation, if an accrediting program exists, and seek recommendations from past customers. Most importantly, write detailed specifications on sample handling procedures, methods, detection limits, and QA/QC requirements, using this chapter's recommendations. It is best to establish a contract with the laboratory spelling out all terms.

Simple field test kits have been marketed for years to perform many of the routine analyses. While these kits are easy to use, they have disadvantages for most urban water monitoring. For most analyses, their detection sensitivity is too low relative to concentrations usually encountered in natural waters and urban runoff, and they are subject to interferences that can be removed in more sophisticated laboratory analyses. They also pose the problem of distracting field workers with the time needed to perform analyses and properly handling reagents that may be toxic. Miniaturized laboratory instruments and multi-parameter testers, such as spectrophotometers and titration kits, represent improvements at least in detectability; but these instruments are quite expensive.

Quality Assurance/Quality Control

General Considerations

The effectiveness of any monitoring effort depends on a QA/QC program. This program provides quantitative measurements of the "goodness" of the data. For some variables, QA/QC involves calibration of instruments with known standards. To obtain measures of accuracy and precision, QA/QC may further involve analyses of blanks, replicate samples, control samples, and spiked samples. QA/QC also embraces cleaning and handling, as well as assessment measures taken with sampling equipment and containers to avoid contamination and validate the success of that endeavor.

Two of the most basic considerations in QA/QC were defined in Chapter 5: accuracy – agreement between the measurement of a variable in a sample and the true value; and precision – agreement among replicate analyses of a sample or among analyses performed on replicate samples. Other terms commonly used in both field and laboratory QA/QC programs are:

- Calibration samples – Samples prepared from distilled-deionized water that contain a known concentration of a specific substance or will produce a known instrument response; used for all instrumental analyses. Calibration samples are typically run at the beginning of an analytical series to set up the instrument and, often, during the course of the series, when they are often referred to as control samples or check standards.

- Blanks – Samples prepared from distilled water, perhaps with reagents added, to represent zero concentration of a specific substance, or to produce an instrument response that indicates zero concentration; used for nutrients, metals, and organics to check contamination.

Blank samples are taken from distilled, reagent grade, analyte-free, deionized water that is used to rinse sample bottles and sampler apparatus. There are a number of types of blanks: (1) instrument blanks – passed through measurement instruments; (2) calibration blanks – tested to discover contamination in auxiliary chemicals used to prepare calibration samples; (3) reagent or method blanks – checked with all analytical reagents added to detect contamination from these sources; (4) transport or field blanks – transported to the sampling location and treated like a sample thereafter to check for contamination introduced in the field; and (5) equipment blanks – pumped through sampling equipment.

QA/QC involves steps at various times in the process: prior to sampling, in the field, during laboratory analyses, and following up to evaluate and properly report results. The next several sections give general QA/QC guidelines for these steps. More detailed information is available in U.S. Environmental Protection Agency (1979).

Preliminary and Field QA/QC

Avoiding sample contamination requires careful cleaning of samplers, sample bottles, and laboratory equipment. Some general guidelines for cleaning are presented here. Analytical procedures for certain variables specify additional requirements. The recommended procedures should be applied to samplers, sample containers, and all laboratory glassware and implements that will come into direct contact with samples during collection, storage, or analysis.

To avoid the contamination of samples by residues or materials commonly found in sampling equipment, sampler apparatus and containers, including automatic sampler tubing and strainer, should be washed first with phosphorus-free detergent, followed by a tap water rinse. Tubing and containers should be treated with 10 percent hydrochloric acid, ultra-pure deionized water, and methanol rinses (omit acid for strainer). Follow cleaning with air drying. After the decontamination procedures have been completed, cap containers and seal other apparatus with aluminum foil. Keep all equipment in a clean, protected area.

Laboratory equipment should always be washed with detergent (generally phosphorus-free), rinsed with tap water, and rinsed an additional three times with ultra-pure deionized water. An ultrasonic cleaner can minimize the need for hand scrubbing. Following the water rinses, perform acid washing with high-purity acids as appropriate (sulfuric acid for nutrient analyses or nitric acid for metals testing). After acid washing, rinse equipment completely at least six times with ultra-pure deionized water and air dry.

If QA/QC criteria given later are not met, thoroughly review the cleaning operation to determine if inadequate cleaning procedures could be causing contamination.

The field QA/QC program consists of instrument calibration, field sample replication, and transport and equipment blanks. It is important to calibrate field instruments like pH and dissolved oxygen meters exactly as specified by the manufacturer. In many cases, the calibration should occur either once with each batch (up to 20) of samples, every few hours when the meter is stationary and continuously powered, each time the meter is turned on or the range is changed, or each time it is moved from one place to another. It is recommended that pH meters be calibrated with two buffers (e.g., pH 4.0 and 7.0) and checked with a third.

Field replicates are repeated samples, collected simultaneously or nearly so at the same location to provide an evaluation of total sample variability (i.e., field plus analytical variability). Generally, duplicates are sufficient for field replication, requiring one extra set of sample containers. As a rule, 5 to 10 percent of sample collections should be duplicated, allocated randomly. This frequency is not burdensome in manual sampling but can be when using automatic samplers. The most careful programs rotate a separate sampler and associated tubing among sites to obtain the requisite number of field duplicates. In addition, equipment blank samples should be collected on 5 percent of occasions and transport blanks carried along on at least 10 percent of site visits, randomly allocated in both cases.

Laboratory QA/QC

Laboratory QA/QC begins with properly registering sample receipt, using a tracking form or custody-transfer record appropriate for the monitoring program. While the layout of these forms varies considerably, they generally include:

- Sampling and laboratory personnel (delivering and receiving sample and perhaps others);
- Delivery date and time;
- Sample identifiers;
- Sampling date and time;
- Sample matrix (e.g., water, sediment);
- Sample container type;
- Sample condition (e.g., preserved, on ice, warm, etc.); and
- Requested analyses.

Performing the remaining laboratory QA/QC procedures is generally the responsibility of the laboratory staff and not directly the province of urban water monitoring personnel. However, these personnel must specify and contract for sufficient QA/QC to meet the monitoring program's objectives as well as assess if it is carried out properly. Therefore, this section outlines typical laboratory QA/QC procedures. A monitoring program may not use all of the procedures mentioned, but all should be considered at the outset and dispensed with only if there is a good reason for doing so.

The next consideration is proper calibration of laboratory instruments. Specialized instruments (e.g., conductivity meters, turbidimeters) should be calibrated at least once with each batch of samples and whenever the instrument range is changed. A batch is considered to be up to 20 samples. Other constituents are analyzed on multiple-purpose instruments like spectrophotometers (nutrients), inductively coupled plasma-mass spectrometers (metals), and gas chromatograph-mass spectrometers (organics). These instruments should be calibrated at the outset of analyses with a calibration blank and a range of at least three concentrations spanning the complete anticipated range in the samples (e.g., 20, 50, and 100 percent of expected upper limit). Control samples should be run at two concentrations (e.g., 20 and 90 percent of the upper limit) with every sample batch.

Following is a list of other laboratory QA/QC procedures. In general, every monitoring program should strongly consider specifying at least the first four, as appropriate for the analyses to be performed. The latter two are often routine with laboratories. Those operating urban water monitoring programs should request the results if available, if they do not choose to write them into the specifications.

- Laboratory replicates – Replicate analyses should be performed on randomly selected sample bottles, generally at the rate of 5 to 10 percent for each analyte, to assess analytical precision. Usually, duplicate analyses are sufficient for procedures that are well proven in the laboratory.
- Method blanks – A method blank should be run with each batch of samples for each analyte requiring reagent addition, passage through an instrument, or both. A result exceeding the reporting limit is an indication of possible contamination. An investigation of contamination might include running an instrument blank, which can distinguish contamination originating in the instrument from a source in the reagents.
- Spiked samples – Spiked samples are prepared by adding known concentrations of a specific substance to an environmental sample. One set of spiked sample analyses should be performed on each batch for analytes subject to interferences from other substances in the water, often termed matrix interferences. Generally, nutrients, metals, and organics are candidates for this procedure. To perform it, a sample is first split into three portions. One part is analyzed for the constituent of interest as usual. The others are spiked with this constituent at a particular concentration, producing a pair of matrix spikes (MSs), together known as matrix-spike duplicates (MSDs). MSD analysis provides measures of both spike recovery and replicability (see *Data Quality Assessment* below).
- Surrogate samples – A surrogate is a type of spiked sample for checking extraction efficiency applied to samples from which organics (e.g., total petroleum hydrocarbons, volatile organic chemicals, pesticides) are extracted and analyzed. Surrogate standards are "non-target" compounds that behave similarly to the constituents of interest when analyzed. A thorough program applies the procedure to every sample, including calibration samples, blanks, and spiked samples, and evaluates recovery.

- External samples (also known as standard reference materials, SRMs) – External samples are prepared by a source outside of the monitoring program to known concentrations of the analytes of interest. To evaluate accuracy, the laboratory should have a set schedule to submit these reference samples to analysts without divulging the concentrations.
- Split samples – Splits are samples divided for independent analysis between two or more parties as a measure of precision. Laboratories should establish arrangements with each other for periodic sample trading.

Data Quality Assessment

Data quality assessment involves steps at the laboratory level and, after receipt of the data, by monitoring program personnel. The laboratory should assess compliance with specific data quality objectives related to the various QA/QC procedures and undertake designated corrective actions if necessary and if at all possible. Monitoring program personnel should review the laboratory's performance in this regard and evaluate field QA/QC results and the completeness and representativeness of the overall program.

The various QA/QC results are assessed quantitatively as follows. Table 6-1 gives suggested data quality objectives as criteria to judge results, as well as actions to take if they are not met.

- Laboratory duplicates – Express precision as the relative percent difference (RPD):
  $RPD = 100(C_1 – C_2)/(C_{avg})$
  where: C_1 = Larger of two values;
  C_2 = Smaller of two values; and
  C_{avg} = Average of two values = $C_1 + C_2/2$
- Method blanks – Compare to RLs, MDLs, and measured sample values. Elevated readings signal probable contamination and reduce ability

Table 6-1: Suggested Data Quality Objectives and Actions to Take

QA/QC Procedure	Data Quality Objective[a]	Action[b]
Laboratory duplicates	Total suspended solids – RPD ≤ 20 Particle size distribution – RPD ≤ 30 Nutrients – RPD ≤ 20 Total hardness – RPD ≤ 10 Metals – RPD ≤ 20	
Pesticides – RPD ≤ 20	Reject batch results if RPD > 2 times objective. Flag batch results as estimates if RPD = 1-2 times objective.	
Method blanks	Maximum detected blank value ≤ 2 times RL	Reject batch results if blank value > 2 times RL. Flag as an estimate if sample measurement < 5 times a detected blank value. Investigate possible contamination sources in the field and laboratory. As needed, make use of equipment and instrument blanks and review all cleaning procedures.
Spiked samples	%R = 75-125	Reject batch results if %R < 50 or > 150. Flag batch results as estimates if %R = 50-75 or 125-150 (except do not flag if measurement < RL and %R = 125-150).
Surrogate samples	%R = 50-150	Reject batch results if %R < 50 or > 150.
External samples	%R within control limits established by the laboratory based on historical performance	Review and correct all relevant procedures if %R outside control limits and reanalyze control samples until objective met.

Notes: [a] RPD – Relative percent difference; RL – Reporting limit; %R – Percent recovery
[b] When data quality objectives are not achieved, reanalyze the affected batch, beginning with preliminary processing steps like filtering, if possible.

to quantify concentrations with confidence, particularly low values.

- Spiked and surrogate samples – Express percent recovery (percent R) as:

  percent R = 100(S – U)/C

  where: S = Measured concentration in spiked or surrogate sample;

  U = Measured concentration in untreated sample (zero if not detected); and

  C = Actual concentration of spike or surrogate added.

The precision of spiked or surrogate sample analysis can also be assessed as RPD using the two MSD results.

- External samples – Express accuracy as percent recovery (percent R) using the formula:

  percent R = 100(M – T)/T

  where: M = Measured value; and

  T = True value of external standard.

In addition to tracking the laboratory's QA/QC performance and treating the data accordingly, monitoring program personnel should make assessments of:

- Field replicates – Poor replication could result from problems in the field or in the laboratory. If laboratory QA/QC shows that the laboratory is not the source, field notes and field procedures should be reviewed and corrected as necessary. Data from the sampling occasion are suspect, and judgment must be rendered on whether or not those data should be used or, if they are, flagged as estimates. Use the same RPD criteria as applied to laboratory duplicates to assist judgment.
- Sample holding times – Exceedence of designated holding times is cause for evaluating the acceptability of results, with judgment and overall QA/QC performance determining if and how the data will be used.
- Completeness – There should be evaluation and reporting of how completely the monitoring program fulfilled the coverage anticipated by the original objectives. One rather arbitrary criterion is that the program should produce as valid samples at least 95 percent of the targeted numbers; i.e., that analyzed data be reported for a minimum of 95 percent of the collected samples.
- Representativeness – It should be evaluated and reported how well the monitoring program reflected the scope of coverage anticipated by the original objectives. Criteria set during the monitoring program design should be brought into this evaluation (e.g., hydrograph coverage, rainfall quantities, antecedent dry period length, etc.).

How to Analyze Data

Handling Data Below Reporting Limits

Data sets from urban water monitoring programs frequently exhibit values below reporting limits, termed non-detected (ND) or censored data. This situation is particularly prevalent with dissolved metals and organic chemicals. The initial task in data analysis is to decide how to handle such values quantitatively in conjunction with higher numbers. Statisticians and others have debated numerous philosophies and techniques for handling censored data. The choice of method can substantially affect calculations performed on the data, particularly when ND values are proportionately numerous in the data set.

Kayhanian, Singh, and Meyer (2002) outlined and applied five methods to calculate the multi-event averages of event mean concentrations (EMCs) for 16 pollutants in highway runoff and used the averages to estimate mass loadings. Non-detected values among the pollutants ranged from 2 to 88 percent of the total measurements. Mass loading estimates produced by the five methods differed by less than 1 to more than 70 percent. The lower the reporting limit and the percent not detected, the lower the disagreement was. There was little consistency in the tendency of any method to give relatively high or low estimates. The authors did not recommend any one method. It appears from their results that a simple conventional technique like assigning a value of half the reporting limit to non-detected data gives results very similar to more sophisticated methods if the percentage not detected is relatively small (< 5 to 10 percent). With a larger proportion of censored data, a statistician's assistance should be enlisted to apply one of the other methods.

Graphing

The variety of possible objectives for urban water monitoring programs and the many different statistical and numerical data analysis methods available mean that any extensive treatment of these topics is beyond the scope of this book. The reader interested in references relevant to urban water resources should consult a statistical text like Zar (1998) or, for exploring the relationships among variables, a multivariate data analysis reference like Everitt and Dunn (2001). It can be said that, before any of these methods are applied to most cases, the data should be explored graphically.

Graphing too can take many different forms. The most basic is probably the scatter plot, in which a water quality variable is graphed against an independent variable like time, distance, or some condition thought to have a possible influence on it (e.g., a land use characteristic or another water quality variable). The scatter plot can be studied for the existence of a trend, which might suggest a follow-up statistical or numerical analysis (e.g., linear regression with an apparent linear trend).

Another very useful graph is the box plot (Figure 6-9). The box embraces relatively high and low values (e.g., upper quartile or decile [75th or 90th percentile] and lower quartile or decile [25th or 10th percentile]), with a bar to signify the median value (50th percentile). "Whiskers" extend from the box to indicate maximum and minimum values that are not outliers. Outliers are defined in various ways (e.g., more than 1.5 box lengths or 3 standard deviations above or below the box) and are shown individually.

Probability plots also find considerable use in analyzing urban water resources data. A probability plot graphs the cumulative probability of measurements falling above or below given values. A special form much used in this work is the log-probability plot to investigate if the data have a log-normal distribution (Figure 6-10).

Figure 6-9: Box Plot Structure

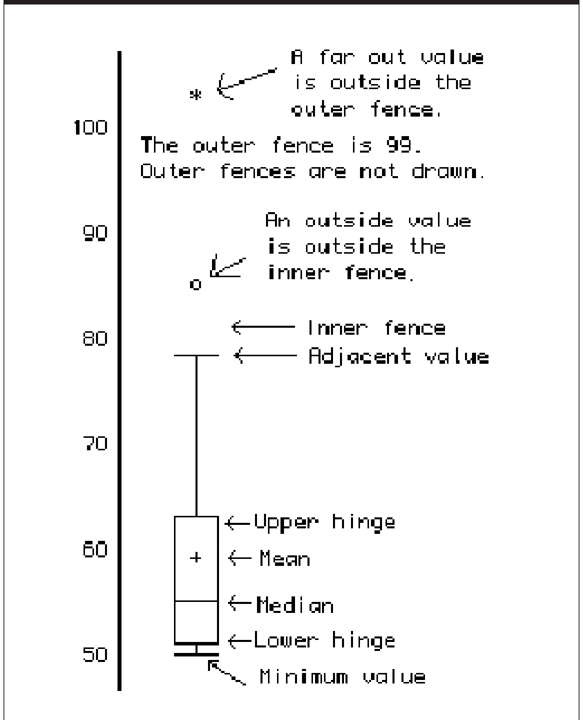
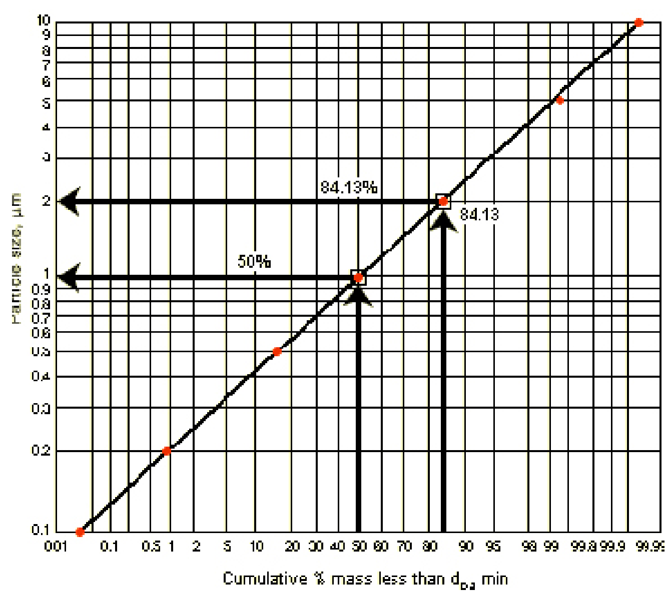


Figure 6-10: Lognormal Probability Plot Example



If the probability plot of logarithmically transformed data is linear or nearly so, the log-normal distribution can be safely concluded. Water quality data are often found to have this distribution, and it is therefore frequently assumed without testing; but performing the simple graphical exercise even without formal statistical assessment adds assurance to data analysis.

Pollutant Mass Loading Estimation

Mass loading is the product of water volume times pollutant concentration. It is usually computed over a period of time, commonly a year. It is ideal and feasible to have a continuous flow record over the entire period to establish volume. However, there is almost never anywhere near complete coverage of concentration for a period of any length. Therefore, mass loading estimates must be made with only a partial record of events, usually made up of a series of EMCs from sampled events. To obtain a good loading estimate, it is obligatory that the sampled events be representative of the period of interest. As discussed earlier, representativeness is an important aspect of monitoring program design at the outset, and of data quality assessment at the conclusion of monitoring. Charbeneau and Barrett (1998) made the case that averaging EMCs is appropriate for estimating long-term mass loadings, which are affected more by volume than concentration. This advice reinforces the advisability of obtaining a continuous and complete flow record.

If the EMCs have a log-normal probability distribution, as they usually do in water quality data sets, they cannot be simply averaged. In that case, the appropriate expression for the mean EMC (a) is (Marsalek, 1990):

$$a = \mathrm{Exp}(\mu + s^2/2)$$

where: Exp signifies exponentiation on the base of the natural logarithms, e;

μ = Mean of natural logarithms of EMCs; and

s^2 = Variance of natural logarithms of EMCs.

The confidence interval (CI) surrounding the estimate can also be calculated by (Marsalek, 1990):

$$CI = a\ \mathrm{Exp}\{\pm\ \varphi\ [s^2/n + 2\ (s^2)3/(n-1)]^{0.5}\}$$

where: + is used for the upper confidence limit;

- is used for the lower confidence limit;

φ = 1.96 for 95 percent confidence interval or 1.69 for 90 percent confidence interval; and

n = Number of EMC values used to estimate mean.

Assessment of BMP Performance

BMP performance is a frequent subject of urban water monitoring programs. Most often, performance has been expressed in terms of efficiency as a percentage of entering pollutants captured by the device. Efficiency can be computed according to pollutant concentrations or loadings and in several ways. A cooperative project of the U.S. Environmental Protection Agency and the American Society of Civil Engineers established a National Stormwater Pollutant Database and prescribed methods for analyzing efficiency, pointing out six different ways to compute it (URS Greiner Woodward-Clyde et al., 1999). In a comparative example calculation, efficiencies ranged widely, depending on the computation method, in some cases by more than an order of magnitude.

Reliance on efficiency as the chief performance measure stems from wastewater practices that preceded the development of the stormwater management field. This measure generally provides a reasonably good picture of effectiveness in treating municipal and industrial effluents. However, stormwater and its management differ from these effluents in important respects: (1) flow is intermittent instead of, usually, continuous; (2) both flow rates and pollutant concentrations are generally more variable in stormwater; (3) long storage periods in some stormwater BMPs separate influent and effluent hydrologic characteristics widely in time; and (4) extended exposure to the soil and atmosphere in many stormwater BMPs subtracts water through infiltration and evapotranspiration. Therefore, calculating efficiency from point-in-time inlet and outlet concentration measurements, which is a common practice in wastewater work, is generally not valid in stormwater BMP monitoring.

The best way of setting up the efficiency calculation to recognize these realities of stormwater dynamics is as a comparison of the summation of inlet and outlet mass loadings:

Efficiency = (Sum of inlet loadings – Sum of outlet loadings)/ Sum of inlet loadings

Obtaining a good estimate of efficiency depends on having continuous flow records at the inlet and outlet and sufficient event mean concentration measurements to estimate loading within acceptable error bounds. If these data are adequate, estimating efficiency in this way will account for the effects of intermittent flow, variability, lag between inflow and outflow, and water loss. The resulting efficiency will reflect interdiction of pollutant transport through physical, chemical, and biological treatment mechanisms, as well as from flow quantity reduction through water loss. As URS Greiner Woodward-Clyde et al. (1999) pointed out, though, this and other similar methods they reviewed still will not tell if pollutant removal from inlet to outlet is statistically significant. To make that judgment, they put forth the method they called log-normal statistical efficiency, which uses an analysis of variance procedure. The interested reader should consult the original source for details.

Beyond the issue of how to calculate efficiency lies the matter of its adequacy as the only performance measure for stormwater BMPs. The relative variability of pollutant concentrations that can be found in stormwater affects efficiency. It has frequently been noted that efficiency of a given BMP tends to drop as the influent pollutant concentrations decline. The reason for this phenomenon is probably that it is relatively easy to achieve a large initial reduction by capturing the most treatable flow components (e.g., the largest solids making up the TSS) but increasingly harder to gain additional efficiency operating on the less treatable components (e.g., the smaller solids). While a relatively "clean" influent is often associated with low efficiencies, though, the effluent concentration often tends to be similar to that discharged by the same BMP treating a "dirtier" influent at a higher efficiency. Therefore, BMP performance should be judged by both efficiency of mass loading reduction and effluent quality.

The California Department of Transportation (2004) analyzed by linear regression effluent concentrations as functions of influent concentrations for a range of pollutants and various types of ponds, biofilters, and media filters. In some cases, the regressions were not statistically significant and the effluent concentrations were fairly uniform regardless of influent quality, whereas in others, significant regression equations were derived to forecast effluent (C_{eff}) in relation to influent (C_{inf}) concentrations:

$$C_{eff} = m\, C_{inf} + b$$

where: m = Slope of the regression line; and

b = Vertical-axis intercept of regression line.

The intercept b represents the irreducible minimum C_{eff}, the best quality effluent the BMP is capable of.

Sediment Monitoring

Introduction

Urban runoff and other diffuse sources of pollution are highly variable from place to place, and even in one place over time, in effluent quality and environmental effects. Therefore, these sources and their effects are difficult to characterize. They are often more dilute in contamination than industrial and municipal wastewater sources, and their negative impacts may be more the result of cumulative, chronic effects than of short-term, acute ones. Sediment monitoring offers the opportunity both to perform measurements on a component of the environment that does not vary so rapidly and to assess the potential for cumulative effects as well. Because of these advantages, sediment monitoring deserves more attention in urban water resource monitoring programs than it currently receives.

Sediments influence the environmental fate of many toxic and bioaccumulative substances in aquatic ecosystems. Specifically, sediment quality is important because many toxic contaminants found only in trace amounts in water can accumulate to elevated levels in sediments. As such, sediments serve both as reservoirs and as potential contaminant sources to the water column. Sediments tend to integrate contaminant concentrations over time and can represent long-term sources of contamination. Sediment-associated contaminants can also directly affect benthic full-time residents and other organisms that utilize bottom habitats for essential biological processes (e.g., spawning, incubation, rearing). Sediments, therefore, provide an essential link between chemical and biological processes. By understanding this link, environmental scientists can develop assessment tools and conduct monitoring programs to evaluate the health of aquatic systems more accurately.

Sediment monitoring has a great deal in common with water quality monitoring, particularly in the areas of objectives, the determination process for sample numbers, sample handling, commonly performed laboratory analyses, QA/QC, and data quality assessment. In these areas, provisions similar to those for water quality apply, except where supplemented or modified by special considerations given here. This section emphasizes subjects where the two types of monitoring differ, particularly locations, timing, and collection of samples and data analysis and interpretation. On the latter subject, an important issue in using sediment data is whether contaminants, especially metals, found in sediments are natural or from human activity. Research in Florida pioneered interpretive techniques to assist in making this judgment and design watershed management strategies accordingly. This section will highlight these methods after covering the basics of sediment monitoring.

Sample Collection

What and Where to Sample

As with all monitoring, sediment monitoring programs should be designed with respect to clear, comprehensive, specific objectives. One common general objective for sediment monitoring programs is to determine the level of sediment contamination existing, perhaps for comparison with quality criteria, dredging, or targeting sediment capping. In this case, it will probably be appropriate to composite a number of samples from the area of interest. Other common purposes for sediment monitoring are to determine the spatial variability of contamination, or to compare two or more areas or different situations. The best sampling design in this case would be to collect replicate samples from each area for separate analyses, with a composite from each as the fallback strategy if budget is limited. In all of these situations, the number depends on the areas' sizes, pollutant variability, acceptable uncertainty, and the cost of sampling. Three samples from an area, analyzed separately or composited, are a minimum for statistical purposes.

Another consideration in locating sampling stations concerns the variability of contaminant levels as a function of sediment grain sizes. The finer solids tend to concentrate pollutants more than the larger particles because of their greater surface area per unit volume available for surface processes of attachment (e.g., adsorption). The relative distribution of particles by size depends on hydrodynamic conditions, with the finer ones tending to deposit in slower flowing areas and the larger ones in faster moving locations. How to decide what flow regimes to sample depends, once again, on objectives and should be carefully considered.

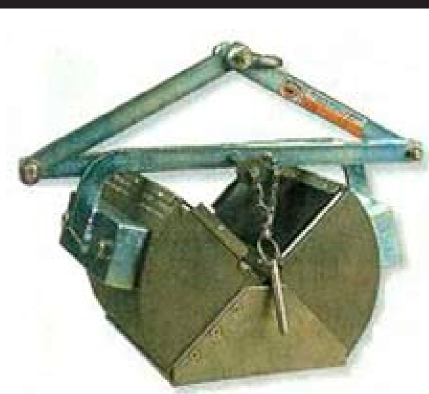
Concerning sediment depth to sample, most monitoring is intended to document recent contamination and relatively short-term trends; hence, samples are most frequently collected from the surficial sediments (typically, the top 5 to 15 cm). Deeper sampling should probably be considered only when the objectives are directed at longer-term or historical trends.

When to Sample

Sediment monitoring has the considerable advantages over water column monitoring of integrating pollutants over time and being much less transient. Still, currents do redistribute sediments, and this phenomenon must be considered in monitoring program design. With reference to the program's objectives, the designer can decide how important this factor is and, if it matters, how to structure sampling seasonally. Obviously, if very short-term trends in a stream are the focus of objec-

Figure 6-11: Ekman Dredge and Ponar Dredge

Ekman Dredge



Ponar Dredge

Presented as examples only and does not constitute endorsement of the products by the U.S. Environmental Protection Agency or the North American Lake Management Society

tives, sediment monitoring must occur when sediments are being deposited during the storm season. If the overall potential effect of a winter's deposition on fish spawning in the early fall is the issue, monitoring must occur after the cessation of winter runoff and before spawning starts. If comparisons are to be made based on samples taken once a year for several years, monitoring should take place at the same time each year. If sediment monitoring is to occur in a relatively stable lake or wetland system, seasonal timing probably matters less. A clear statement of objectives should make these decisions quite easy to make.

How to Sample

Samplers available for sediment monitoring include scoops, corers, and dredges. The basic type to choose and the particular selection among the alternatives depend on the water body, the conditions, and the program objectives. Scoops can be ordinary shovels or fashioned from common materials for use in shallow water while wading. Some corers are meant for hand use, while others are for deep water. They are usually not appropriate for sediment sampling in streams unless they are being used for frozen core samples (Burton and Pitt, 2002). Dredges are usually most appropriate for dropping from a boat or bridge.

Samplers should be cleansed in the same way as containers used for water samples for analyses of nutrients, bacteria, metals, and organic chemicals. When the objective is to analyze certain quantities without contamination, the collector must be made from special materials. Again, the same guidelines prevail as presented for analyses of the same quantities in water; e.g., sediments to be subjected to volatile organic compound analysis must contact only Teflon.

- One of the most commonly used sediment samplers is the Ekman dredge (Figure 6-11). It is small and light and the easiest among the options to set. Because of its light weight, the Ekman dredge can collect samples only from soft mud, silt, or sand. However, for the same reason, it is the best choice where fine particles are likely to be disturbed by the force produced by objects moving through water. Depending on the required amount of sediment that must be collected, its relatively small size can necessitate collecting replicates to obtain sufficient quantity. Setting the dredge with a pole in shallow water increases the depth and success of sample collection. The Ekman dredge must be tripped with a messenger, similar to a Van Dorn water sampler. If the depth is not known, it is a good idea to touch the bottom with the dredge, pick it up a few feet, and move over a few feet before slowly lowering it back down to the bottom. If the sediment is compacted, or if there is a lot of gravel, rocks, or large debris, the heavier Peterson or Ponar (Figure 6-11) dredges must be used for sampling.
- Both the Peterson and Ponar dredges are large enough (14 to 32 kg, 30 to 70 lbs) to usually require winches for raising and lowering, although models small enough to be raised and lowered by hand are available. Small sticks that can prevent the Ekman dredge from closing will be crushed by these two dredges. Because of their larger sizes, much more water is displaced, and thus fine sediment is easily swept away before the jaws close. Gently lowering these dredges the last few feet can reduce the problem. These larger dredges can collect a larger surface area, but still only sample the top few centimeters of sediment. Attaching weights to them is one way to increase the depth of their bite. The Peterson and Ponar dredges are held open by their own weight and tripped by letting the line go slack.
- Whatever the type of dredge, a smooth retrieval is desirable to avoid losing some of the sample. It is a good idea to place a bucket under the dredge and haul it out of the water within the bucket to avoid letting some of the sample escape with the dripping water.

Less common in general urban water resources work are bedload samplers, which collect the sediments that travel along the stream bed. These samplers are box or basket traps located on the bed with open ends facing into the current. Some bedload samplers are embedded in the stream bottom with a slot opening even with the sediment surface.

Special Considerations for Sample Handling

Sediment samples should generally be passed through a 2-mm sieve to remove twigs, leaves, and other debris larger than any of the sediment particles. For some objectives, it is appropriate to separate the sample further by particle size. If the analysts wish to distinguish contaminant levels in fine sediments versus larger ones or versus an overall bulk sample, a subsample of fines should be separated out by sieving through a sieve of appropriate opening size (e.g., 63 mm).

Special Considerations for Sample Analysis

As with water samples, the variables to specify for laboratory analysis depend on the program's objectives, required certainty, costs, and available budget. With sediments, in comparison to water, there is a much higher potential to detect trace substances like the less prevalent metals, pesticides, polynuclear aromatic hydrocarbons, PCBs, etc.; and these measurements are often the principal subjects for analysis.

In addition to contaminants, sediment samples should generally be analyzed for:

- Grain size distribution;
- Moisture content;
- PH; and
- Organic content (as loss on ignition, also termed volatile organic solids, or total organic carbon).

Grain size distribution is important information in interpreting the relative ability of the sample to concentrate contaminants depending on the relative surface area of its makeup. Knowing the moisture content permits expressing results in terms of dry weight of bulk sample, which is superior to expression in terms of wet weight, which is variable. The pH is a key factor in the relative solubility of metals. A relatively acidic pH can mean that metals have dissolved in the water instead of adsorbed to solids; it does not necessarily indicate that they are low in the overall environment. Physicochemical processes by which contaminants associate with solids are related to the amount of organics present. For example, organics provide small pores for adsorption of synthetic organic chemicals.

Special Considerations on How to Analyze Data

General Guidelines

Sediment contaminant concentrations should be expressed in mass of pollutant per unit dry weight of sediment (e.g., mg zinc/dry kg of sediment, which is equivalent to parts per million). Quantities that occur in smaller amounts can be expressed in μg/kg, which is equivalent to parts per billion.

A useful way of expressing sediment contamination is the enrichment ratio, which is the ratio of the pollutant concentration in a sample to the concentration in a reference sample. The reference should be a sample that is equivalent in every way possible but stems from a location considered to be unaffected or minimally affected by the contamination sources influencing the sample being quantified.

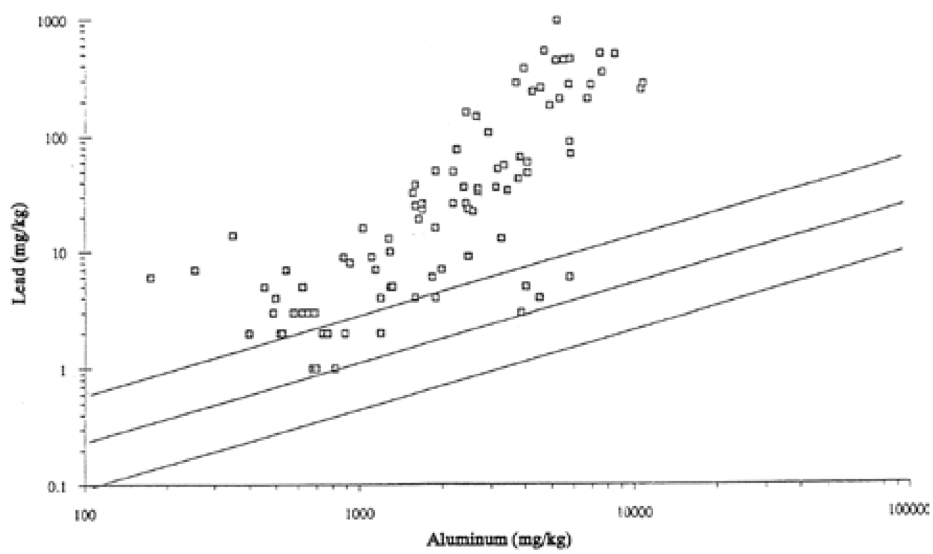
Assessing Sediment Contamination Source

In the past, determining whether aquatic sediments were anthropogenically enriched with metals was a difficult process requiring comprehensive site-specific assessments. In recent years, Florida researchers have developed a practical approach for judging the likelihood of human versus natural sources, relying on normalization of metal concentrations to a reference element (Florida Department of Environmental Protection 1988). In Florida, normalization of metal concentrations to aluminum concentrations in estuarine sediments proved the most promising method of comparing metal levels regionally. Further research in Florida, Canada, and Washington State indicated that other crustal metals little influenced by anthropogenic sources (e.g., lithium) can also be appropriate reference elements for assessing sediments. Lithium is sometimes a better basis in areas whose geology is strongly influenced by glacial erosion. The Washington work extended application of the technique from estuarine to freshwater wetland sediments.

To understand this assessment tool, it is helpful to know the geochemical processes that govern the behavior and fate of metals in water. Natural sediments are predominantly composed of debris from weathering of rocks. Acids formed in the atmosphere or from the breakdown of organic matter (e.g., carbonic, humic, fulvic acids) mix with water and form leaching solutions. These leaching solutions break down rocks and carry away the products in solution or as solid debris. This debris is chiefly composed of chemically resistant minerals, such as quartz and clay minerals, which are the alteration products of other aluminosilicate minerals. Naturally occurring metals can substitute for aluminum in the aluminosilicate structure, where they are tightly bound and not prone to being released in water. In contrast, dissolved metals from natural and anthropogenic sources adsorb to particulate matter, a more loosely bound configuration. These metals are generally more subject to release back into the dissolved form by physical or chemical changes in the water.

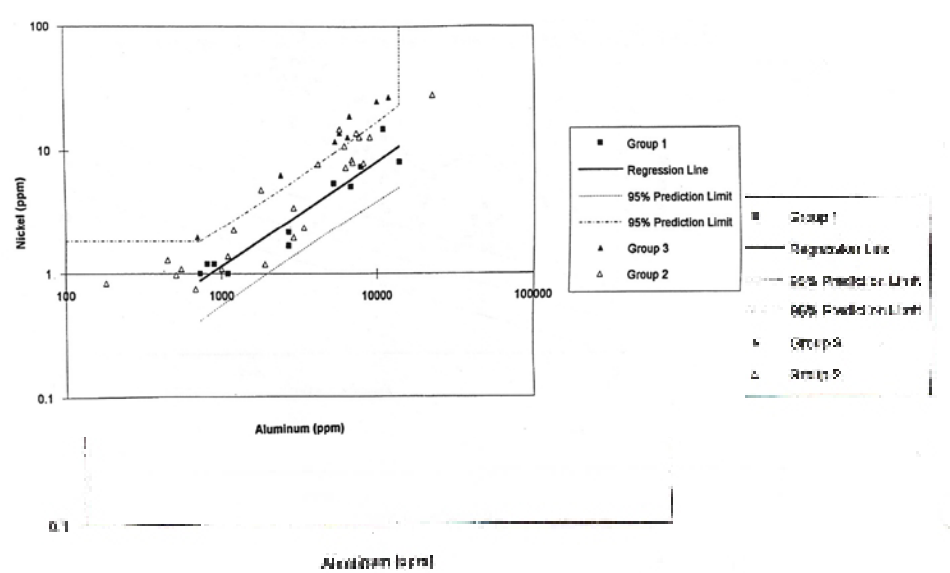
The tool for interpreting metal concentrations in sediments is based on demonstrated, naturally occurring relationships between metals and aluminum (or an alternative like lithium). These natural relationships were used to develop guidelines to distinguish natural sediment deposits from anthropogenically contaminated sediments. This tool is based on a statistical linear regression analysis with aluminum (or lithium) as the independent variable and another metal of interest as the dependent variable. A plot of the regression line

Figure 6-12: Lead Versus Aluminum in Biscayne Bay Sediments



Source: Adapted from Florida Department of Environmental Protection, 1993

Figure 6-13: Nickel Versus Aluminum in Puget Sound Area Freshwater Wetland Sediments



Source: Adapted from Valentine 1994, Azous and Horner 2001 and 95 percent confidence lines on either side brackets the region expected to contain sediments with the metal primarily originating from natural sources. Figure 6-12 shows the relationship for lead versus aluminum in Biscayne Bay (Florida) sediments. All points lying above the upper confidence limit are regarded as anthropogenically contaminated.

For Puget Sound area (Washington) freshwater wetlands, the regression was first performed using data from a set of wetlands whose watersheds had relatively little urbanization (Group 1 in Figure 6-13 for nickel versus aluminum; Valentine, 1994; Azous and Horner, 2001). Then data were plotted from moderately and heavily urbanized sets (Groups 2 and 3, respectively, in Figure 6-13). The figure shows that, for nickel, all of the Group 3 samples fall above the 95 percent confidence line, as do a majority of the Group 2 samples. While aluminum as the reference yielded the better regression in the example, for other metals, lithium as the reference resulted in a superior regression for this glaciated region.

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Measurement Procedure Using a Current Meter

1. Extend a measuring tape at right angles to the direction of flow and measure the width of the cross section. Record measurements on a data sheet. Leave the tape strung across the stream.
2. Divide the width into segments using at least 20 points of measurement. If previous flow measurements have shown uniform depth and velocity, fewer points may be used; smaller streams may also require fewer points. Measuring points should be closer together where depths or velocities are more variable. Cross sections with uniform depth and velocity can have equal spacing.
3. Record the distance from the initial starting bank and the depth.
4. Record the current velocity at each measuring point. Horizontal (from left to right bank) and vertical (top to bottom) variation of stream velocity may influence streamflow measurements. To correct for vertical differences, measuring at certain depths can yield acceptable estimates of the mean velocity over a vertical profile. If the depth exceeds 0.8 m (2.5 ft), velocities should be measured at 20 percent and 80 percent of full depth and averaged to estimate mean velocity. In the depth range 0.1 to 0.8 m (0.3 to 2.5 ft), take the velocity at 60 percent of the full depth (measured from the surface) as an estimate of the mean over the profile. Measuring velocity in water shallower than 0.1 m (0.3 ft) is difficult with conventional current meters. If much of the reach of interest is very shallow, or flow is too slow for current meter measurement, consider installing a control section and V-notch weir.
5. Calculate flow as a summation of flows in partial areas (Figure 6-A-1) using the following equation:

$$q_n = v_n d_n (b_{n+1} - b_{n-1})/2$$

where: b_{n-l} = Distance from initial point n to the preceding point n-1 (m [ft]);

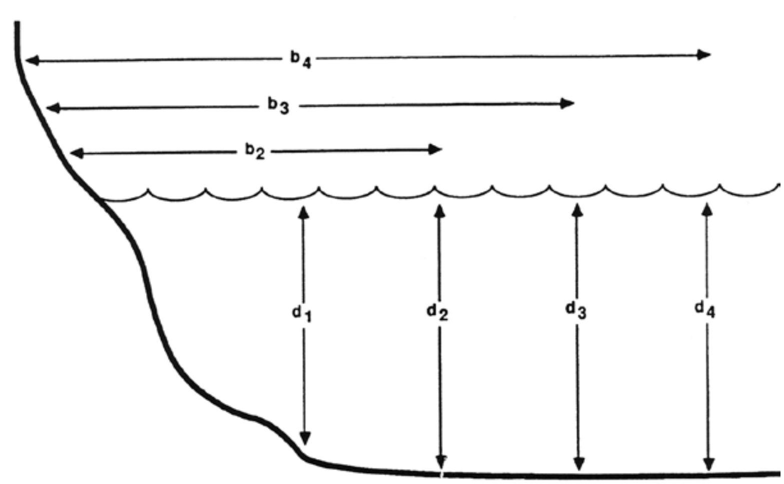
b_{n+l} = Distance from initial point n to the following point n+1 (m [ft]);

d = Mean depth of partial area n (m [ft]);

v = Average current velocity in partial area n (m/sec [ft/sec]); and

q = Discharge in partial area n (m^3/sec [ft^3/sec]).

Figure 6-A-1: Variables Used to Calculate Stream Discharge Using the Current Meter Method



Recommended Sampling and Analysis Procedures for Water Quality Variables

Variable	Container[a]	Preservation[b]	Maximum Holding Time	Analytical Methods[c]	Reporting Limit	Unit
Miscellaneous						
pH	P, G	None (field)	None	EPA 150.1; SM 4500-H^+	0.1	pH
Dissolved oxygen	Gd[d]	None (field)[e]	None[e]	EPA 360.1, 360.2; SM 4500-O	0.1	mg/L
Conductivity	P, G		28 days	EPA 120.1; SM 2510	1	μS
Total hardness	P, G	HNO_3 to pH < 2	6 months	EPA 130.1, 130.2; SM 2340B	0.5	mg/L
Alkalinity	P, G		24 hours	EPA 310.1, 310.2; SM 2320	0.1	mg/L
Biochemical oxygen demand	Gd[d]		24 hours (6 preferred)	EPA 405.1; SM 5210	3	mg/L
Chemical oxygen demand	P, G	H_2SO_4 to pH < 2	28 days	EPA 410.1; SM 5220	10	mg/L
Residual chlorine	P, G	None (field)	None	EPA 330.5; SM 4500-Cl	0.1	mg/L
Cyanide	P, G	NaOH to pH > 12	14 days	EPA 335.2; SM 4500-CN^-	3	μg/L
Solids						
Total suspended solids	P, G		7 days	EPA 160.2; SM 2540D	1	mg/L
Total dissolved solids			7 days	EPA 160.1; SM 2540C	1	mg/L
Turbidity			48 hours	EPA 180.1; SM 2130	0.05	NTU
Particle size distribution				SM 2560	1	μL/L
Nutrients						
Total phosphorus	P, G	H_2SO_4 to pH < 2	28 days	EPA 365.1; SM 4500-P F	5	μg/L
Soluble reactive phosphorus			48 hours	EPA 365.1; SM 4500-P F	2	μg/L
Ammonia-nitrogen			28 days	EPA 350.1; SM 4500-NH_3	10	μg/L
Nitrate + nitrite-nitrogen			28 days	EPA 353.1; SM 4500-NO_2, NO_3	10	μg/L
Total nitrogen			28 days	SM 4500-N	10	μg/L
Total Kjeldahl nitrogen			28 days	EPA 351.1; SM 4500-N_{org}	100	μg/L

Recommended Sampling and Analysis Procedures for Water Quality Variables *continued*						
Variable	Container[a]	Preservation[b]	Maximum Holding Time	Analytical Methods[c]	Reporting Limit	Unit
Metals						
Silver	P, Teflon, or borosilicate glass	HNO_3 to pH < 2	48 hours to filter for dissolved, 6 months to analyze	EPA 200.8; SM 3125	0.2	μg/L
Aluminum					25	μg/L
Arsenic					0.5	μg/L
Cadmium					0.2	μg/L
Chromium (total)					1	μg/L
Copper					1	μg/L
Nickel					2	μg/L
Lead					1	μg/L
Zinc					1	μg/L
Chromium (Cr^{+6})				EPA 218.4; SM 3500-Cr	50	μg/L
Selenium				EPA 270.2, 270.3; SM 3500-Se	2	μg/L
Mercury	G or Teflon	5 mL/L of 12 N HCl or BrCl[f]	48 hours to filter for dissolved, 28 days to analyze	EPA 245.2; SM 3112	0.5	μg/L
Pathogens						
Fecal coliforms	Sterile P, G	None[g]	8 hours	SM 9221, 9222	1	cfu/100 mL or MPN/100 mL
Escherichia coli				SM 9221, 9222	1	cfu/100 mL or MPN/100 mL
Total coliforms				SM 9221, 9222	1	cfu/100 mL or MPN/100 mL
Enterococci				SM 9230	1	col/100 mL
Total petroleum hydrocarbons						
TPH-gasoline	G		14 days	EPA SW 8015	50	μg/L
TPH-Diesel			7 days to extract, 40 days to analyze	EPA SW 8015	50	μg/L
TPH-motor oil			7 days to extract, 40 days to analyze	EPA SW 8015	50	μg/L
Oil and grease		HCl or H_2SO_4 to pH < 2	28 days	SM 5520	5	μg/L

Recommended Sampling and Analysis Procedures for Water Quality Variables *continued*						
Variable	Container[a]	Preservation[b]	Maximum Holding Time	Analytical Methods[c]	Reporting Limit	Unit
Pesticides						
Organochlorines	Amber glass		7 days to extract, 40 days to analyze	EPA SW 8081. 8085; SM 6630	0.01-0.1	μg/L
Organophosphorus				EPA SW 8085	0.01-0.1	μg/L
Nitrogen				EPA SW 8085	0.01-0.1	μg/L
Carbamates				EPA SW 8321	0.07-3.5	μg/L
Herbicides				EPA SW 8085; SM 6640	0.1-1.0	μg/L
Miscellaneous organics						
Polynuclear aromatic hydrocarbons	Amber glass	None[g]	7 days to extract, 40 days to analyze	EPA SW 8270, 8310; SM 6440	0.05	μg/L

[a] P – plastic (polyethylene); G – glass.
[b] Hold all samples on ice in the field and at 4°C in the laboratory, in addition to any preservation listed. HNO_3 – nitric acid; H_2SO_4 – sulfuric acid; HCl – hydrochloric acid.
[c] EPA – from U.S. Environmental Protection Agency (1983); SM – from American Public Health Association (1998); EPA SW – from U.S. Environmental Protection Agency (1986).
[d] Biochemical oxygen demand bottle.
[e] Can be chemically fixed in the field and titrated in the laboratory.
[f] Filter for dissolved sample analysis before preservation.
[g] Normally none except holding at 4°C but add sodium thiosulfate in the presence of chlorine.

Types of Water and Sediment Quality Variables

Note: Abbreviations and customary units of measurement are in parentheses; mL – milliliters; L – liters; mg – milligrams; µg – micrograms; µL – microliters.

- Measures of solids – Impacts include light and visibility reduction, abrasion of sensitive aquatic animal tissues, transport of other pollutants, and sediment deposition.
    - Settleable solids (mL/L).
    - Total suspended solids (TSS, mg/L) – Trapped by 0.45-micrometer filter.
    - Total dissolved solids (TDS, mg/L) – Passed through 0.45 micrometer filter and measured gravimetrically after sample evaporation.
    - Turbidity (Nephelometric Turbidity Units, NTUs) – Represents light-scattering ability of suspended particles.
    - Particle size distribution (PSD, % by volume larger than or smaller than given sizes; diameters at which selected % occur [e.g., d_{10}, d_{50}]; µL particle volume/L water volume) – Determined by an electronic particle counter.
- Nutrients – Increases cause eutrophication, excessive nuisance algal growth accompanied by change in algae types (tendency toward filamentous); oxygen depletion upon death and decay.
    - Phosphorus (µg/L in natural waters, sometimes mg/L in effluents) – Most often responsible for eutrophication in fresh waters.
        - Total phosphorus (TP).
        - Soluble reactive phosphorus (SRP), sometimes orthophosphate-phosphorus, which makes up most of SRP.
    - Nitrogen (µg/L in natural waters, sometimes mg/L in effluents) – Most often responsible for eutrophication in salt waters.
        - Ammonia nitrogen (NH_3-N or NH_4^+-N) – Also toxic in high concentrations.
        - Nitrate- (NO_3^-), nitrite- (NO_2^-), and nitrate+nitrite-nitrogen.
        - Total Kjeldahl nitrogen (TKN) – Ammonia plus organic nitrogen.
        - Total nitrogen (TN).
- Metals (µg/L in natural waters, sometimes mg/L in effluents) – Many are toxic to aquatic life, and some bioaccumulate and biomagnify; the first three are most often detected in stormwater runoff and natural waters.

    Copper (Cu)
    Lead (Pb)
    Zinc (Zn)
    Antimony (Sb)
    Arsenic (As)
    Beryllium (Be)
    Cadmium (Cd)
    Chromium (Cr), +3 and +6 valences, total
    Mercury (Hg)
    Nickel (Ni)
    Selenium (Se)
    Silver (Ag)
    Thallium (Th)

Metals can be measured as dissolved, "total recoverable," or both. Dissolved metals have the most immediate toxic effects, but those in solid state can dissolve and also accumulate in sediments and affect life there.

Calcium (Ca) and magnesium (Mg) are non-toxic metals that reduce solubility and therefore harmful effects of other metals and together produce what we call "water hardness." Water quality criteria are based on hardness. Whenever the objective is to determine if natural water metals criteria are met, hardness should be determined and expressed as "mg/L calcium carbonate, $CaCO_3$."

- Pathogens (no. colonies/100 mL, with no. colonies often expressed as most probable no. [MPN]) – Criteria and limits are in terms of “indicators” that may not be disease-causing themselves but are intended to indicate the presence of fellow-traveling direct pathogens. Analysis of specific pathogens is almost never done in routine environmental work, indicators are extremely variable, and pathogen methods are arguably the least satisfactory in aquatic science.
    - Fecal coliforms – Present in the bodies of all warm-blooded animals.
    - Total coliforms – Some have natural sources, especially soils.
    - Enterococci – Closer indicator of human disease potential than fecal coliforms but not much advantage in environmental variability.
- General measures of organics:
    - Biochemical oxygen demand (BOD, mg/L) – Commonly used to monitor sewage and other effluents high in rapidly decomposable organics.
    - Total organic carbon (TOC, mg/L).
    - Chemical oxygen demand (COD, mg/L).
- Petroleum and its products:
    - Oil and grease (mg/L).
    - Total petroleum hydrocarbons (TPH, mg/L) – Often divided into fractions such as Diesel and gasoline.
- Specific organic chemicals (μg/L) – Many are toxic to aquatic life, and some bioaccumulate and biomagnify.
    - Only pentachlorophenol, certain pesticides, and polychlorinated biphenyls (PCBs) have water criteria for natural waters.
    - Other groups may be represented in effluent limitations and are sometimes detected in natural waters; examples:
        - Volatile organic chemicals (VOCs) – Components of solvents and fuels, most but not all containing chlorine or bromine (benzene and its relatives are exceptions); many are carcinogenic; easily lost from samples to atmosphere and very reactive with other substances.
        - Organophosphorus pesticides (e.g., the commonly used diazanon).
        - Polynuclear aromatic hydrocarbons (PAHs) – Combustion by-products often found in stormwater runoff and sometimes in natural waters.
        - Numerous other industrial and commercial chemicals with various formulations.
- Miscellaneous quantities.
    - Temperature (^{o} Celsius).
    - pH – On 0-14 scale, with 0-6.99 signifying acidic, 7.00 neutral, and 7.01-14 alkaline.
    - Dissolved oxygen (DO, mg/L).
    - Conductivity (microsiemens/centimeter, μS/cm) – Measures ability of water to conduct an electric current because of presence of all dissolved substances, most of which are of natural mineral origin and are not pollutants.
    - Total residual chlorine (mg/L) – Toxic to aquatic life.
    - Cyanide (μg/L) – Toxic to aquatic life.

CHAPTER 7

Biological Monitoring and Assessment

CHAPTER 7

Biological Monitoring and Assessment

Introduction

Scope of Biological Monitoring and Assessment

Traditionally, monitoring surface waters to assess their ecological health and the effects of pollution discharges has relied upon physical and chemical measurements of samples from the water column. This approach originated when the emphasis was on the effects of discharges such as wastewater treatment plants and industrial effluents. These discharges are continuous and generally have a lower degree of variability in flow and water quality than intermittent sources like urban stormwater runoff. Judgment of the effects of relatively uniform discharges on aquatic organisms is based to a large extent on bioassays exposing test species to concentration ranges of pollutants. While these standard procedures deviate from the reality of stresses in natural systems in many respects, they do reflect uniform effluents better than intermittent, more variable ones. When attention turned to these latter discharges, the need for a more direct means of assessing actual ecological effects became apparent. This chapter outlines techniques of biological community assessment to detect the effect of diffuse sources of pollution on aquatic life.

Intermittent discharges create shock loadings to a water body, and the ecological effects depend on many variables and complex interactions. Moreover, many runoff pollutants become attached to sediment particles and settle quickly, exerting detrimental effects over a long period. Furthermore, the high peak flow rates and volumes of urban runoff degrade habitat (e.g., channel and bank erosion) and elevate sediment deposition, the effects of which are not detected by water quality monitoring.

Monitoring biological communities is the most integrated approach to surface water quality assessment and management. While water quality data reflect short-term conditions that exist when a particular sample is collected, biological communities accurately indicate overall environmental health because they continuously inhabit receiving waters and react to various long-term physical and chemical influences. Aquatic organisms also integrate a variety of environmental influences, hydrologic and other physical aspects, chemical effects, and interactions among the biota themselves.

Biological assessment involves integrated analyses of structural and functional components of the aquatic communities. Bioassessments are best used to detect aquatic life impairments and assess their relative severity. Once impairment is detected, additional chemical and biological toxicity testing can identify the causative agent(s) and the source. Both biological and chemical methods are critical in successful pollution control and environmental management programs. They are complementary, not mutually exclusive, ways to enhance overall program effectiveness.

In summary, key advantages of bioassessments are (after Barbour et al., 1999):

- Biological communities reflect the overall ecological integrity of all elements of complex systems.
- Over time, biological communities integrate the effects of various stressors operating at different levels, providing a measure of response to fluctuating environmental conditions.

- By assessing the integrated response to highly variable pollutant inputs, biological communities provide a practical approach for monitoring runoff source impacts and the effectiveness of best management practices.
- Routine monitoring of biological communities can be relatively inexpensive, particularly when compared to the cost of assessing toxic substances.
- The public is highly interested in the status of biological communities as a measure of environmental health.

In the broadest sense, biological community assessment embraces monitoring of both habitat features and biota in both the plant and animal kingdoms. Habitat features of importance to aquatic life are very numerous and even extend outside of the aquatic environment itself to the riparian zone. Primary producers are found among the macrophytic rooted plants and the periphyton attached to surfaces in the water. Benthic macroinvertebrates are both consumers of aquatic and terrestrial primary production and food sources for fish. Clearly, the potential subjects for monitoring biological communities are so numerous and diverse that designing a feasible monitoring program that will give needed answers highly depends on formulating carefully considered, comprehensive objectives as covered in Chapter 5.

There has been much progress in aquatic bioassessment through benthic macroinvertebrate monitoring. This segment of the community directly represents the ability of the resource to sustain life, which habitat assessment does not. Relative to fish, the invertebrates are much less mobile. They therefore register conditions in a particular location better and are considerably easier to monitor. Progress accelerated when researchers developed indices representing overall benthic macroinvertebrate communities instead of attempting to interpret the significance of measures on individual species or genera.

Beginning in the 1980s, researchers, agencies in some states, and the U.S. Environmental Protection Agency began to develop and codify procedures to guide biological monitoring and assessment. The early developments culminated in issuance of Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish (Plafkin et al., 1989), updated as Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish (Barbour et al., 1999). The Rapid Bioassessment Protocols (RBPs) give complete coverage of monitoring the three biotic communities named in the title, as well as habitat. Burton and Pitt (2002) summarized key provisions of the habitat, benthic macroinvertebrate, and fish protocols, as well as related methods from the state of Ohio, in their Appendices A to C. This chapter does not repeat the extensive material in these references but helps guide potential users to the resources they can find within the protocols, with specific attention to habitat and invertebrate monitoring. It also exemplifies some of the subject matter with case studies representing individual approaches found to work well. The chapter also briefly covers aquatic toxicity assessment, which is not a part of the RBPs.

Reference Conditions for Biological Monitoring

The issue of reference conditions is critical to the interpretation of biological surveys. The term "reference" is more appropriate when applied to a foundation for comparison in studies within the natural environment than "control," which is commonly used in laboratory experimentation. A reference site is a location unaffected by the variable(s) whose effects are to be measured at a "treatment" or "test" site according to the objectives adopted for the monitoring program. In this context, the term "treatment" refers to the influence created by the variable(s) of interest. If the reference site is carefully chosen, equivalent measurements of conditions there and at the test location should elucidate the type and extent of effects created by the test variable(s). As with so much else in monitoring program design, selecting a good reference site depends strongly on clear, complete objective statements. In practice, test variables in urban water resources work are often measures of impact, such as the quantity and water quality of a discharge, or of BMP performance.

Barbour et al. (1996a) described two general types of references, site-specific and regional. The first type represents measurements of conditions either upstream of an intervention, such as a discharge or BMP installation, or from a "paired" watershed. The appendix to Chapter 5 explores paired watershed monitoring program design in some detail. Site-specific references are established at the outset of a monitoring program, in relation to its specific objectives. Regional references, on the other hand, consist of measurements from a population of sites relatively unaffected by the usual influences being studied within a fairly homogeneous region and habitat type. These sites usually serve as bases for comparison for various monitoring programs within the region for which they are suited.

Two major concerns always attend the designation of reference sites. One regards the similarity in all conditions except the test variables between the reference and treatment sites. Dissimilarity introduces potential alternative explanations for any observed effect. The second question is the degree that influences to be tested could still have on a reference site. This latter question is particularly evident in urban water work, where the urban influences could be so all-pervasive that there are no true references. In reality, this matter must often be resolved by settling for the "best attainable" conditions (Horner et al., 2002), locations that are not "pristine" but where human presence is sufficiently muted to offer the ability to distinguish clearly what are the effects of stronger influences.

Barbour et al. (1996a) noted the advantages of site-specific upstream references: (1) if carefully selected, the habitat quality is often similar to that measured downstream, thereby reducing complications in interpretation arising from habitat differences; and (2) impairments due to upstream influences from other sources are already factored into the reference conditions. Where feasible, effects should be bracketed by establishing a series or network of sampling stations at points of increasing distance from the test location, although this strategy can greatly raise costs. These stations will provide a basis for delineating impact and recovery zones. In significantly altered systems (e.g., channelized or heavily urbanized streams), suitable reference sites are usually not available. In these cases, historical data or simple ecological models can be used to establish reference conditions, although with less confidence than afforded by direct measurement.

While site-specific reference conditions represented by the upstream-downstream, or paired-site approach are desirable, Hughes (1995) pointed out three problems associated with their use: (1) they provide limited capacity for extrapolation beyond the site-specific; (2) they hence involve a substantial assessment effort that is likely to have little value in future monitoring; and (3) in many cases there are too few reference sites, often only one, for statistical assessment of measurement uncertainties.

Regional reference sites can overcome these disadvantages of site-specific references. The concept of systematically regionalizing reference site establishment got a major boost with Omernik's (1987) ecoregional framework for interpreting spatial patterns in state and national data. The geographical framework is based on regional patterns in land-surface form, soil, potential natural vegetation, and land use. Geographic patterns of similarity among ecosystems can be grouped into ecoregions and sub-ecoregions. Naturally occurring biotic assemblages, as components of the ecosystem, would be expected to differ among ecoregions but be relatively similar within a given ecoregion.

Establishing and characterizing a set of reference locations represents a substantial investment for a region. But this investment can pay off over time in more revealing and certain interpretations of monitoring findings than those that are possible with complete reliance on site-specific referencing. Nevertheless, site-specific references will still be required to meet certain objectives.

Habitat Monitoring

Introduction

An evaluation of habitat quality is crucial to achieving many objectives in the assessment of ecological integrity. Raven et al. (1998) pointed out that habitat and biological diversity are closely linked. In the most general sense, "habitat" incorporates all aspects of the physical and chemical environment, along with the interactions among living organisms. This broad spectrum makes the number of possible monitoring subjects a very long list, which for feasibility must be pared in relation to the objectives being pursued. Usually, the definition of habitat is narrowed to the quality of the internal (to the water body) and external riparian environments that influence the structure and function of the aquatic community. The RBPs adhere to this definition. This more restricted definition still leaves many possible conditions for consideration as monitoring subjects, and the attendant need to apply objectives to focus on the most crucial ones.

The presence of an altered habitat structure is one of the major potential stressors of aquatic systems (Karr et al., 1986). A degraded habitat, often from hydrologic modification, can sometimes obscure investigations of the biological effects of contaminated water, sediments, or both. Habitat monitoring should be strongly considered, along with physical, chemical, and biological monitoring, when the objective is to distinguish such effects. Habitat knowledge is essential to pairing stations for study (i.e., upstream and downstream or in paired watersheds). Because conditions are usually not identical from site to site, some habitat data can help in interpreting measurements from paired sites. Where physical habitat quality at a test site is similar to that of a reference, detected impacts can be attributed to water quality factors or other stressors. In the opposite situation of dissimilar habitats, the location with more degraded habitat could be limited more by that condition than other stressors. With all its potential value, though, habitat monitoring cannot replace or be a surrogate for biological measurements when the objective is to discern the condition of biotic populations or communities. Biological quality cannot be safely deduced from habitat conditions and must be measured directly, if that is the focus of objectives.

The following subsections on monitoring program elements preserve the same terminology with regard to "sampling" that was used in preceding monitoring chapters. Of course, tasks in habitat monitoring frequently do not involve sampling in the same context as collecting a parcel of water or sediment. Instead, habitat monitoring usually involves tasks like measuring dimensions and observing and then describing (and also perhaps scoring) environmental attributes. Nevertheless, this chapter maintains the terminology, both for consistency in language and to emphasize that the same set of decisions must be made in properly designing habitat monitoring programs as in any other monitoring effort.

What to Sample

As pointed out in the introduction, potential variables for habitat monitoring are very numerous. Researchers and agency staff based at local, state, and federal levels have collectively devised many standard variable lists and protocols for guiding habitat monitoring. Locally and regionally derived lists typically represent the concerns of the area (e.g., anadromous fish spawning and rearing habitat features in the Pacific Northwest, conditions for resident warm-water and cold-water fish species in the Upper Midwest) and usually should not be transferred in whole to other places.

The RBPs prescribe a series of descriptive, measured, and scored variables in their habitat monitoring protocol. The descriptive set consists of:

- Stream characterization;
- Watershed characterization;
- Riparian vegetation;
- Aquatic vegetation; and
- Sediment and substrate.

Measured quantities are:

- Instream features (measured or estimated dimensional characteristics, mostly);
- Large woody debris; and
- Basic water quality variables (temperature, conductivity, dissolved oxygen, pH, and turbidity).

The RBPs recommend scoring the following variables in four categories from optimal to poor quality for organism support, with five numerical scores to represent distinctions within each category. In some cases, variables do not apply to both high- and low-gradient streams but only one.

- Epifaunal substrate and available cover – Includes the relative quantity and variety of natural structures in the stream, such as cobble (riffles), large rocks, fallen trees, logs and branches, and undercut banks available as refugia, feeding areas, or sites for spawning and nursery functions of aquatic macrofauna.
- Embeddedness – Refers to the extent to which rocks (gravel, cobble, and boulders) and snags are covered or sunken into the silt, sand, or mud of the stream bottom. Generally, as rocks become embedded, the surface area available for macroinvertebrate and fish shelter, spawning, and egg incubation decreases.
- Pool substrate characterization – Evaluates the type and condition of bottom substrates found in pools. Firmer sediment types (e.g., gravel, sand) and rooted aquatic plants support a wider variety of organisms.
- Velocity/depth combinations – Patterns of velocity and depth are included for high-gradient streams under this parameter as an important feature of habitat diversity. The best streams in most high-gradient regions will have all four

patterns present: (1) slow-deep, (2) slow-shallow, (3) fast-deep, and (4) fast-shallow.

- Pool variability – Rates the overall mixture of pool types found in streams according to size and depth. The four basic types of pools are large-shallow, large-deep, small-shallow, and small-deep.
- Sediment deposition – Measures the amount of sediment that has accumulated in pools and the changes that have occurred to the stream bottom as a result of deposition.
- Channel flow status – The degree to which the channel is filled with water.
- Channel alteration – A measure of large-scale changes in the shape of the stream channel.
- Frequency of riffles or bends – A way to measure the sequence of riffles and thus the heterogeneity occurring in a stream. Riffles are a source of high-quality habitat and diverse fauna.
- Channel sinuosity – Evaluates the meandering of the stream. A high degree of sinuosity provides for diverse habitat and fauna, and the stream is better able to handle surges when it fluctuates as a result of storms.
- Bank stability – Measures bank erosion or the potential for erosion.
- Bank vegetative protection – Measures the amount of vegetative protection afforded to the stream bank and the near-stream portion of the riparian zone.
- Riparian vegetative zone width – Measures the width of natural vegetation from the edge of the stream bank out through the riparian zone.

For example, categories and scores for embeddedness are shown in Table 7-1.

As mentioned earlier, work in various places has identified habitat variables providing the most crucial information for regional biota. A systematic, objective process of singling out, from the multitude of possibilities, those variables giving the best return of important information permits streamlining habitat monitoring. For example, May (1996) used a partial least-squares correlation analysis to identify the most effective physical habitat measures of Puget Sound area lowland stream quality, in relation to the ability to support benthic macroinvertebrate communities. The resulting variables are:

- Large woody debris frequency;
- Large woody debris volume;
- Glide habitat (as percent of wetted area);
- Pool habitat (as percent of wetted area);
- Pool frequency;
- Cover on pools (vegetation cover as percent of total pool area);
- Stream bank stability; and
- Embeddedness.

Several investigators have developed regional indices representing habitat quality based on multiple variables. An index is useful to express relative habitat quality among different locations in the same or different streams. The indices have generally been composed of scores assigned to observations (and in some cases measurements) in categories, usually a simplified version of the RBP scoring illustrated above. Since these indices are not a mathematical combination of measurements themselves, they have customarily been labeled "qualitative." The Ohio Environmental Protection Agency's (1989) Qualitative Habitat Evaluation Index (QHEI) incorporates substrata, instream cover, channel morphology, riparian width and cover, and pool, glide, and riffle characteristics. May's Qualitative Habitat

Table 7-1: Embededness Categories and Scores

Habitat Parameter	Condition Category			
	Optimal	Suboptimal	Marginal	Poor
Embeddedness	Gravel, cobble, and boulder particles are 0-25% surrounded by fine sediment. Layering of cobble provides diversity of niche space.	Gravel, cobble, and boulder particles are 25-50% surrounded by fine sediment.	Gravel, cobble, and boulder particles are 50-75% surrounded by fine sediment.	Gravel, cobble, and boulder particles are more than 75% surrounded by fine sediment.
Score	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
Source: Barbour et al., 1999				

Index (QHI) is made up of 15 variables scored in four quality categories and combined additively.

Where to Sample

The usual variability of habitat and impossibility of monitoring throughout the ecosystem inevitably raises the question of where it is best to monitor. Barbour et al. (1999) advised in the RBPs that, when sampling water bodies with complex habitats, a complete inventory of the entire reach is not necessary. They add, however, that the sampling area should be representative of the reach, incorporating riffles, runs, and pools if these habitats are typical of the stream in question. Mid-channel and wetland areas of large rivers, which are difficult to sample effectively, can be avoided. Sampling effort should be concentrated in near-bank habitats where most species occur.

When to Sample

If habitat knowledge is essential to the objectives, it should be monitored simultaneously with biological sampling. Simultaneous monitoring saves effort and therefore costs. While habitat is not invariable over time, it is likely to vary less, and more slowly, than the biota of interest. Therefore, it is appropriate for the biological monitoring schedule to control timing of habitat work.

How Many Samples to Take

Overall, the same representativeness and statistical considerations covered in Chapter 5 and (for water quality and sediment monitoring) in Chapter 6 also apply to habitat monitoring. When true sampling is required, composite sampling, as opposed to individual small replicates, is the norm for RBP investigations to characterize stream reaches. However, taking too few samples for a composite can be a major source of variance. Replication is strongly encouraged for precision evaluation of the methods.

How to Sample

As with guidance on what to sample in habitat monitoring, there are numerous guides on performing the various monitoring tasks. The RBPs contain field data sheets and associated text guidance. Prior to formulating his own system, May (1996) assessed more than 10 general purpose protocol documents, most from the Pacific Northwest, and a number of additional special purpose procedures. He incorporated the most appropriate features from these resources to arrive at a comprehensive procedure applicable to wadeable streams supporting or potentially supporting anadromous salmon in and near urban areas. The reader launching a habitat monitoring program has the choice of proceeding with the national RBPs, at least at first, or seeking out already developed regional methods. Such methods may already have adapted the RBPs for regional circumstances. These methods should be evaluated for the intended purposes, and either used if applicable or adapted with guidance by the objectives and the modifications made by others when they needed different techniques.

Chapter 6 emphasized the importance of careful, complete field notes while taking water quality samples. Field notes are, if anything, even more important for habitat monitoring, since they constitute the only record of many measurements and observations of habitat attributes. Any reader contemplating or performing habitat monitoring should consult the Chapter 6 discussion of field notes. The safety of the record should be ensured in all cases, but particularly for habitat monitoring, by keeping more than one copy of field notes in different locations.

Quality Assurance/Quality Control

Habitat monitoring QA/QC is less developed and formalized than for water quality and sediment monitoring. The RBPs recommend the following steps:

- Train field personnel in the assessment techniques being used.
- Calibrate the judgment criteria for each habitat variable and for the stream settings in which they will be assessed. Calibration involves determining if generally recommended scoring systems apply in these settings by employing them, as a preliminary step, across a representative range

of conditions and then adjusting them if warranted.

- Make periodic checks of assessment results using pictures of monitoring reaches and discussions among the personnel involved. A more advanced form of this step would be independent assessments by participants, a form of replication for precision evaluation.

How to Analyze Data

To a large extent, routine data analysis in habitat monitoring accompanies or is a nearly immediate outgrowth of monitoring itself. The scoring systems and indices described above represent straightforward means of analyzing raw data. The Snohomish County [Washington State] Public Works Department (2002) took a somewhat more advanced approach to index formulation, using the procedure for developing indices of biological integrity (IBIs) to derive an index of habitat integrity (IHI). The following section on Benthic Macroinvertebrate Monitoring covers the IBI procedure. The variables identified by Snohomish County as being most instrumental were:

- Fine sediment ratio – Bed surface ratio of fine sediment (< 6.3 mm) to larger particles;
- Unstable banks ratio – Ratio of unstable to stable bank length;
- Hydrologically modified banks ratio – Ratio of hydrologically modified to unmodified bank length;
- Pool functional area ratio – Ratio of pool areas to non-pool areas;
- Pools per unit bankfull width; and
- Large woody debris and stumps per unit bankfull width.

Benthic Macroinvertebrate Monitoring

Introduction

Benthic macroinvertebrates live in close association with the bed (benthic area) of a water body, are visible to the human eye, and do not possess an internal skeleton. They include the juvenile forms of insects, which typically emerge to the terrestrial world as adults; mollusks (e.g., snails, clams); crustaceans (e.g., crayfish, shrimps, amphipods); and various worms. They have some but not a high degree of mobility. As biological monitoring subjects, benthic macroinvertebrates have a number of advantages (Barbour et al., 1999):

- Macroinvertebrate assemblages are good indicators of localized conditions. Because many have limited migration patterns or a sessile mode of life, they are particularly well suited for assessing site-specific impacts (e.g., in upstream versus downstream studies).
- Macroinvertebrates integrate the effects of short-term environmental variations. Most species have a complex life cycle of approximately one year or more. Sensitive life stages respond quickly to stress; the overall community responds more slowly.
- An experienced biologist can often detect degraded conditions with only a cursory examination of the benthic macroinvertebrate assemblage. Macroinvertebrates are relatively easy to identify to family; many taxa intolerant of human-induced stresses can be identified to lower taxonomic levels with relative ease.
- Benthic macroinvertebrate assemblages are made up of species that constitute a broad range of trophic levels and pollution tolerances, thus providing strong information for interpreting cumulative effects.
- Sampling is relatively easy, requires few people and inexpensive gear, and has minimal detrimental effect on the resident biota.
- Benthic macroinvertebrates serve as a primary food source for fish, including many recreationally and commercially important species.
- Benthic macroinvertebrates are abundant in most streams. Many small streams (first and second or-

ders), which can have a diverse macroinvertebrate fauna, only support a limited fish fauna.

- Most public agencies that routinely collect biosurvey data focus on macroinvertebrates. Many regions already have background macroinvertebrate data.

Where to Sample and How Many Samples to Take

As with all other monitoring, allocation of sampling effort should be based on the set objectives and weighing of variability and costs to acquire the desired information with a sufficient level of assurance. Benthic macroinvertebrates generally vary greatly in spatial dimensions in response to the substrata, depth, velocity, overhanging and aquatic vegetation, and other conditions. Therefore, a statistical analysis will often dictate a very large number of samples to arrive at a reliable estimate of population sizes. There has been a definite trend away from population estimates and the diversity indices that were applied in analysis of population data, except for basic research purposes. The environmental management field has moved instead toward benthic indices of biotic integrity (B-IBIs). The section *How to Analyze Data* below covers the concepts and methods associated with B-IBIs.

If the objective is to develop a B-IBI or use an existing one, the first question is whether to sample a single habitat type (e.g., a stream riffle) or multiple habitats (e.g., beneath overhanging vegetation, in addition to a riffle). The RBPs present protocols for both options. The original RBPs (Plafkin et al., 1989) emphasized the sampling of a single habitat, a riffle or run, as a means to standardize assessments among streams having those habitats. The revised RBPs (Barbour et al., 1999) still considered this approach to be valid, because macroinvertebrate diversity and abundance are usually highest in cobble substrate (riffle/run) habitats. Where cobble substrate is the predominant habitat, this sampling approach provides a representative sample of the stream reach. However, some streams naturally lack much of that substrate. In cases where the cobble substrate represents less than 30 percent of the sampling reach in reference streams, one or more other habitats should be sampled. Habitats to sample should be selected based on the habitat availability in the reference state, and not in potentially impaired streams. Absence in an impaired stream of a habitat that occurs in the reference will, of course, influence the results, which is appropriate in comparing conditions.

The RBPs define five habitat types that support benthic macroinvertebrates (Barbour et al., 1999):

- Cobble (hard substrate) – Prevalent in riffles (and runs), which are a common feature throughout most mountain and piedmont streams; dominant in many high-gradient streams;
- Snags – Accumulated woody debris that has been submerged for a relatively long period (not recent deadfall and not large logs, which are generally difficult to sample adequately);
- Vegetated banks – Submerged lower banks having roots and emergent plants associated with them;
- Submerged macrophytes – Aquatic plants rooted on the bottom of the stream; and
- Sand (and other fine sediment) – Usually the least productive macroinvertebrate habitat in streams, although the most prevalent in some streams.

The RBPs recommend using a kick net for single-habitat sampling and a D-frame net for multiple-habitat work (refer to *How to Sample* below). The protocols prescribe the types and numbers of locations each net type should be used in. They specify compositing all samples to represent the habitat.

B-IBIs have been developed principally through single-habitat (riffle) monitoring programs using Surber samplers (refer to *How to Sample* below). Using a statistical bootstrap algorithm, Fore and Karr (unpublished manuscript cited by Karr and Chu, 1997) analyzed how many samples taken in this way are needed for a relatively precise quantification of metrics making up the B-IBI. They concluded that using the mean of three replicates taken within the riffle habitat is sufficient and that using five replicates yields little additional precision.

Another consideration is sufficient sampling to obtain an adequate number of invertebrates to be representative. There is disagreement on this point, with Karr and Chu (1997) recommending the collection, identification, and counting of at least 500 individuals per habitat for B-IBI metrics computations, a larger number than that cited in the RBPs. These authors believe that sampling sufficient organisms is far more important than the way in which sampling is organized. It is certainly true that larger numbers give more precision but also that taxonomic work is time-consuming and therefore expensive. Like all other decisions in monitoring program design, this one ultimately comes down to the objectives, environmental variability, and available budget. The dilemma illustrates once more the importance of considering and defining these factors at the outset of the program and then relying on them in making decisions.

With most objectives aimed at impact or BMP assessment, ideal sampling locations for the three replicates consist of rocks 5 to 10 cm (2 to 4 inches) in diameter resting on pebbles in a water depth of 10 to 40 cm (4 to 16 inches) within the main flow of the stream (Karr and Chu, 1997). The three locations should be selected through a random process. On grid paper, map a fairly homogeneous reach approximately 50 to 100 meters (164 to 328 ft) in length, if the riffle is at least this extensive. Each square represents a potential Surber sampling spot of 1 ft square. Eliminate any grids that lie beneath undercut banks, overhanging vegetation, or other bank influences. Number the remaining grids. Use a random number generator to select three grids for sampling.

Upon occasion, urban water resources monitoring programs may have objectives more far-reaching than the routine ones. In these cases, a more sophisticated sampling program design could be warranted. For example, variability within a habitat type could be explored by sampling along several transects within it. These transects could be placed randomly or purposely in locations representing different substrata. A sample allocation strategy more sophisticated than the simple random approach outlined above (e.g., stratified random sampling, systematic sampling) could better apply in these cases.

When to Sample

Many invertebrates are found in aquatic systems throughout the year. Still, seasonal factors cause shifts in numbers and relative dominance. Insects emerge in response to temperature and light. Large flows can wash out organisms and deplete the community for a time. Therefore, timing should be considered relative to objectives. If the objective is to determine the maximum production capability of the system, sampling should occur when conditions are stable and elevated temperature and light are stimulating biological activity, but before there is substantial emergence (i.e., spring or early summer). If the objective is to compare conditions among streams, sampling should occur in all of them over a short time span. If the intention is to make comparisons over a period of years, monitoring must be scheduled for the same time each year.

As an example, monitoring of human development effects in the Pacific Northwest has concentrated on sampling each September for several good reasons (Karr and Chu, 1997): water flows are generally fairly stable and safe for field work then, before the fall and winter rains, and invertebrates tend to be abundant. Sampling in September also minimizes disturbance of the spawning redds of anadromous salmonids.

How to Sample

In the RBPs, Barbour et al. (1999) described five sampling devices commonly used in macroinvertebrate assessment work (all with the standard 500-μm mesh size nytex screen):

- Kick net – 1 x 1 meter (3.3 x 3.3 ft) net attached to two poles; most efficient for sampling cobble substrate where velocity of water will transport dislodged organisms into net; designed to sample 1 m^2 of substrate at a time; can be used in any depth from a few centimeters to just below 1 meter;
- D-frame dip net – Frame 0.3 meter (1 ft) wide x 0.3 meter (1 ft) high and shaped like the letter "D" attached to long pole; net is cone- or bag-shaped for capture of organisms; can be used in a variety of habitat types, either as a kick net or for "jabbing," "dipping," or "sweeping";
- Rectangular dip net – Frame the same size as a D-frame net and also attached to a long pole; net is cone- or bag-shaped; sampling is conducted similarly to the D-frame device;
- Surber sampler – Frame 0.3 x 0.3 meter (1 x 1 ft) placed horizontally on cobble substrate to delineate a 0.09 m^2 (1 ft^2) area; vertical section of frame has the net attached and captures the dislodged organisms from the sampling area; restricted to depths of less than 0.3 m (1 ft); and
- Hess sampler – Cylindrical metal frame approximately 0.5 m (1.6 ft) in diameter sampling

an area of 0.8 m^2 (8.6 ft^2); an advanced design of the Surber sampler intended to prevent escape of organisms and contamination from drift; restricted to depths of less than 0.5 m (1.6 ft).

The RBPs give complete advice on using the kick net and D-frame dip net samplers recommended for single- and multiple-habitat work, respectively. Important techniques in using a Surber sampler are:

1. Sample from downstream to upstream to avoid early disturbance of later sampling spots.
2. Stand downstream of the sampling point, place the sampler with the net opening facing upstream, and brace the sampler's frame in place firmly.
3. Starting with rocks closest to the net opening, rub rocks by hand to dislodge invertebrates into the net; place rocks in a bucket of water for later inspection and hand picking of any remaining animals.
4. Thoroughly disturb the pebble layer with a small rake or large spike to a depth of at least 10 cm (4 inches) for at least 1 minute; collect any rocks appearing and save them in the bucket.
5. Lift the frame off the bottom slowly and tilt the net up and out of the water while keeping the open end upstream.
6. Invert the net (and removable receptacle, if the net has one) into a white wash pan; repeatedly dip the net in the stream to concentrate debris in the bottom and empty into the pan each time.
7. Take great care to capture all individuals in the net, using a magnifying glass and forceps to spot and remove animals as necessary.
8. With a small amount of water in the pan, pick through the collection to find all invertebrates and place them in a sample jar about half full of ethyl alcohol preservative and sitting in the pan to avoid spillage; again take great care to capture all individuals, using a magnifying glass and forceps as necessary.
9. Pick invertebrates from the rocks and bucket using the same care and place them in the sample jar.
10. Properly label the sample jar (see *Quality Assurance/Quality Control* below).

How to Handle Samples

Once preserved in 100-proof ethyl alcohol, invertebrate samples are stable for a reasonable period until identification. Because they do deteriorate, the taxonomic work should occur as soon as possible and certainly within months of collection.

The main issue in sample handling is whether or not to subsample to reduce the burden of sorting and identification. Authorities are divided on the wisdom of subsampling. As pointed out earlier, Karr and Chu (1997) believe in the necessity of a sample of 500 animals minimum, which removes subsampling as a consideration unless the collection is particularly rich. Courtemanch (1996) argued against subsampling but recommended a volume-based procedure if it must be done. The RBPs (Barbour et al., 1999) embrace subsampling and cite justification for it. They present a fixed-count approach based on a 200-organism subsample but applicable to any size (e.g., 100, 300, 500). The subsample must be sorted and preserved separately from the remaining sample for quality control checks.

Quality Assurance/Quality Control

QA/QC procedures are less developed for biological than for water quality and sediment monitoring but more complete than for habitat monitoring. The following recommendations are primarily derived from and based on the RBPs (Barbour et al., 1999).

For QA/QC of field work:

1. Prepare sample labels on a medium resistant to deterioration in alcohol with the sample identification code, date, stream name, sampling location, and collector's name. Place labels inside sample containers. Label the outside of the container with the same information. Also, include this information on chain-of-custody forms.
2. After sampling has been completed at a given site, all nets, pans, etc. that have come in contact with the sample should be rinsed thoroughly, examined carefully, and picked free of organisms or debris. Any additional organisms found should be placed into the sample containers. The equipment should be examined again prior to use at the next sampling site.

3. Duplicate sampling at 10 percent of the sites, minimum, to evaluate precision and repeatability of results for the sampling technique and collection personnel.

Recommended laboratory QA/QC procedures are:

1. Laboratory QA/QC personnel or a qualified co-worker should examine at least 10 percent of the sorted samples in each lot. (A lot is defined as a special study, basin study, entire index period, or individual sorter.) The worker will examine the grids chosen and the tray used for sorting and look for organisms missed by the sorter. Organisms found will be added to the sample vials. If the QA/QC worker finds fewer than 10 organisms (or 10 percent in larger subsamples) remaining in the grids or sorting tray, the sample passes; if more than 10 (or 10 percent) are found, the sample fails. If the first 10 percent of the sample lot fails, the worker will check a second 10 percent of the lot. Sorters in training will have their samples 100 percent checked until the trainer decides that training is complete.
2. After laboratory processing is complete for a given sample, all sieves, pans, trays, etc., that have come in contact with the sample should be rinsed thoroughly, examined carefully, and picked free of organisms or debris. Add organisms found to the sample residue.
3. Maintain a voucher collection of all samples and subsamples. These specimens should be properly labeled, preserved, and stored in the laboratory for future reference. A taxonomist (the reviewer) not responsible for the original identifications should spot check samples corresponding to the identifications on the bench sheet.
4. A reference collection of each identified taxon should be maintained and verified by a second taxonomist. The word "val." and the first initial and last name of the person validating the identification should be added to the vial label. Specimens sent out for taxonomic validations should be recorded in a "taxonomy validation notebook" showing the label information and the date sent out. Upon return of the specimens, the date received and the finding should also be recorded in the notebook, along with the name of the person who performed the validation.
5. Record information on samples completed through the identification process in the "sample log" notebook to track the progress of each sample within the sample lot. Update tracking of each sample as each step is completed (i.e., subsampling and sorting, mounting of specimens, taxonomy).
6. Maintain a library of basic taxonomic literature to aid identification of specimens, and update it as needed. Taxonomists should participate in periodic training on specific taxonomic groups to ensure accurate identifications.

How to Analyze Data

There has been a certain loss of confidence in diversity indices for interpretation of trends responding to impacts of human actions and management strategies. As a result, several different kinds of community indices have been introduced. These indices more fully exploit the data resulting from community monitoring and relate more closely to community structure and processes than do diversity indices. Terms applied to the indices include invertebrate community index (ICI) and benthic index of biotic integrity (B-IBI). Indices of these types are composed of metrics representing aspects of both elements and processes within the macroinvertebrate assemblage. Although these indices have been regionally developed, they are typically appropriate over wide geographic areas, usually with some modification of metrics for regional circumstances (Barbour et al., 1995).

Fore and Karr (1994) outlined the general procedure for B-IBI development as follows:

1. Develop metrics appropriate for the geographic area that respond to known sources of human influence.
2. Test the metrics developed in Step 1 with a second, independent data set.
3. Develop an index based on proven metrics; test the index on a third, independent data set.
4. Fine-tune the index.

The key to successful B-IBI development is selection of metrics. The most effective metrics are those having ecological relevance, exhibiting response across a range of human influence, and distinguishing well between relatively pristine and degraded sites. Four studies published from 1995 through 1997 that tested potential metrics in detail serve as a basis for general recommendations, presented in Table 7-2 (DeShon, 1995; Barbour et al., 1996b; Fore et al., 1996; Smith and Voshell, 1997). While these metrics were found to have wide applicability, their utility should be checked regionally and replaced by others if they are found to be superior.

Some basic statistical and graphing procedures are useful in the initial identification of potential metrics. If an index is being constructed to study biological response to increasing urbanization, for example, bivariate correlation analyses relating all potential metrics to an urbanization measure can show the strongest associations. The seven to ten metrics with the highest correlations should be reviewed to judge if any represent very similar attributes and might be dropped. Also, metrics in the next tier of correlation coefficients should be reviewed to see if they represent different attributes that might be an asset to index development. Based on these reviews, decide on the composition of metrics that should be used for an initial trial B-IBI.

The next step in index development is scoring metrics. Continuing with the example, plot each of the tentatively selected metrics versus the urbanization measure. Through visual assessment of the graphs, assign ranges of the urbanization variable representing relatively high to low development, typically in five steps, although there may be situations where one less step or one or two more are appropriate. Again working visually, determine what range of each biological metric is consistent with each urbanization range. Assign scores

Table 7-2: Definitions of Best Candidate Benthic Metrics and Predicted Direction of Metric Response to Increasing Perturbation

Category	Metric	Definition	Expected Reaction to Impairment
Richness	Total # Taxa	Measure of richness of macroinvertebrate Taxa	Decrease
	# EPT Taxa	# of taxa in the EPT insect orders	Decrease
	# Ephemeroptera Taxa	# of mayfly taxa	Decrease
	# Plecoptera Taxa	# of stonefly taxa	Decrease
	# Trichoptera Taxa	# of caddisfly taxa	Decrease
Composition	% EPT	% mayfly, stonefly, and caddisfly larvae	Decrease
	% Ephemeroptera	% mayfly nymphs	Decrease
Tolerance/Intolerance	# Pollution Intolerant Taxa	Richness of perturbation-sensitive species	Decrease
	% Tolerant Taxa	% macrobenthos tolerant of perturbation	Increase
	% Dominant Taxa	Measure of the dominance of the most numerous taxon. Can also be calculated for 2nd, 3rd, 4th and 5th most dominant taxa.	Increase
Feeding	% Filterers	% macrobenthos which filter water or sediment for FPOM	Variable
	% Grazers and Scrapers	% macrobenthos that scrape or graze at periphyton	Decrease
Habit	# of Clinger Taxa	# insect taxa	Decrease
	% Clingers	% insects with fixed retreats or adaptations to attach to surface in moving water	Decrease

Source: Barbour et al. 1999, as compiled from DeShon 1995, Barbour et al. 1996b, Fore et al. 1996, Smith and Voshell 1997 to each metric range from 1 for the lowest interval to 5 for the highest. The B-IBI is the sum of all metric scores, with the maximum possible B-IBI being the product of the number of metrics and the maximum score (5). The tasks outlined here complete Step 1 in the general procedure. Steps 2 and 3 should follow.

In fine-tuning the index (Step 4), it is advisable to perform some statistical and numerical tests to evaluate the index's performance for its intended purpose. These tests can involve linear regressions and multivariate techniques like discriminant function analysis and logistic regression to create models relating B-IBI and independent variables like the urbanization measure. During these analyses, metrics can be removed and added to the index (e.g., R^2, the coefficient of determination for regressions) to see if these alterations result in improvement in measures of model effectiveness.

Toxicity Assessment

Introduction

Toxicity assessment is not a routine activity in urban water monitoring. Many assessment procedures exist, although some have limited usefulness in the usually variable environment of waters affected by runoff from diffuse landscape sources. Therefore, this coverage will just summarize the field in a general manner. Burton and Pitt (2002) provide considerable detail in their Chapter 6 and Appendix D.

Toxicity assessment options can be classified as:

- Toxicity screening procedures – An instrumental analysis to express toxicity to a microorganism of an environmental sample relative to a control sample;
- Whole-effluent toxicity (WET) tests – Controlled laboratory exposure of a test species to various strengths of a natural or wastewater sample;
- Sediment toxicity tests – Controlled laboratory exposure of a test species in vessels containing sediments as well as water, generally over a somewhat extended period;
- In situ toxicity tests – Confined exposure of a test species in the natural environment; and
- Tissue analysis – Measurement of substances in organism tissues to determine their bioaccumulation.

Toxicity Screening Procedures

The most common screening procedure is the Microtox test. Microtox is a trademark of AZUR Environmental. The Microtox Acute Toxicity Test is a 15-minute exposure, metabolic inhibition test that uses freeze-dried luminescent bacteria (Vibrio fischeri NRRL B-11177) to assess the acute toxicity of water, soil, or sediment samples. The test system is comprised of the Model 500 Analyzer, Microtox reagent and test solutions, a personal computer, and MicrotoxOmni™ Software for capturing and analyzing test data. Toxicity is expressed in terms of reduction in light output with exposure to the sample relative to a control. If a series of dilutions of the sample has been tested, which is the standard protocol, and if light reduction exceeds 50 percent, EC50 (the 50 percent effective concentration) can be calculated to express the sample dilution that reduces light output by half compared to the control.

Microtox is a convenient and relatively inexpensive way of identifying the potential for generalized toxicity to aquatic biota of interest. However, it does not provide evidence of the source(s) of toxicity or of actual toxicity to those organisms. To achieve this, more probing analyses must be performed with more specific objectives. Still, the procedure provides urban water resource managers with a tool to identify the need for and plan additional work.

Whole-Effluent Toxicity (WET) Tests

WET tests are bioassays most commonly performed on species long considered to be good laboratory test specimens. The species most often used are the alga Selenastrum capricornutum; the invertebrates (zooplankton) Daphnia magna, Daphnia pulex, and Ceriodaphnia dubia; and the fish Pimephales promelas (fathead minnow). Juvenile rainbow trout (Salmo gairdneri) have also been used quite frequently. The U.S. Environmental Protection Agency (1991) recommended simultaneous WET tests on a fish, an invertebrate, and an alga. National Pollutant Discharge Elimination System (NPDES) permits sometimes require at least fathead minnow and zooplankton bioassays.

WET tests are performed in laboratory containers, most commonly with acute exposures to a number of test organisms over 48 or 96 hours, although longer, chronic tests certainly can be and often have been conducted. A series of dilutions of the sample from 100 percent to 0 (control) strength elucidates a range of effects. In acute tests, the effects are normally simply lethality or non-lethality. Their results are usually expressed in terms of the LC50, the sample concentration lethal to 50 percent of the organisms present, often with the length of the testing period also indicated (e.g., LC50-48 h, LC50-96 h). Chronic assays reveal degrees of sublethal effects, such as effects on mobility, growth, or reproduction.

WET tests have the advantage of being guided by well-standardized procedures under a high degree of control. However, they represent a very artificial environment that makes it difficult to extrapolate results to natural systems. They use organisms that are rarely inhabitants of the ecosystem of interest and omit inter-species interactions (e.g., competition, predation) present in the natural environment. WET tests reflect only one type of stressor, contaminants dissolved in water, and omit the effects of other key stressors, like hydrology-driven phenomena and sediment contamination. Very importantly for urban water resources investigations, they miss the pulse exposures of stormwater runoff.

Some work has been done to address these shortcomings but has not advanced far. For example, Herricks and colleagues experimented with time-scale toxicity testing (Herricks, Milne, and Johnson, 1994, 1998; Brent and Herricks, 1998, 1999). This research was designed to assess the effects of brief exposures, such as would occur in storm runoff episodes, based on sublethal responses during a post-exposure observation period. The results exhibited both delayed effects and, sometimes, organism recovery, which suggests that toxicity tests used to monitor brief exposures should use environmentally relevant exposure durations and post-exposure observations.

Sediment Toxicity Tests

Methods for long-term, chronic testing of sediment toxicity are relatively new (U.S. Environmental Protection Agency, 2000). Using invertebrates as test specimens, these procedures require 42 days or longer to run and are thus relatively expensive. They are mentioned here just to alert urban water resources managers, who may encounter more attention to sediment contamination in the future.

In Situ Toxicity Tests

The basic approach to in situ toxicity testing is to confine test species in a container, expose them to the environment being studied, and observe their lethal or sublethal responses. A number of media exist for in situ toxicity testing, including instream artificial substrates, baskets containing rock or mesh, glass slides, side-stream chambers, and cages. All of these devices require protection from high flows, and they must be secured to the bed of the water body (Burton and Pitt, 2002).

In situ testing has a number of advantages. Like other types of field analysis, this type of testing avoids the extensive artificiality of laboratory conditions and the difficulties of extrapolating laboratory results to the field. Naturally, it also incorporates environmental conditions that are difficult or impossible to produce in the laboratory, such as sunlight; suspended solids; diurnal effects of oxygen and temperature; spatial and temporal variation of physicochemical constituents; stressor magnitudes, frequencies, and durations; presence of natural substrata; and other factors (Burton and Pitt, 2002). Of particular importance in urban water resources investigations, the test organisms are subject to the multiple stresses and actual exposure patterns of episodic stormwater runoff events.

There are also some disadvantages associated with in situ toxicity testing. Test species can be affected by transportation stress and starvation in enclosures. Protecting test media from high flow can artificially alter the effects of flow on organisms. They may not experience the same sediment transport patterns as free-living individuals. Containment removes interactions with other species and other members of the same species. Unprotected devices are also subject to disturbance by humans and animals.

In situ tests provide the best information about toxicity when used in conjunction with the results of other monitoring. In accordance with program objectives, these additional assessments might include monitoring of water quality, sediment quality, or both; habitat conditions; and benthic macroinvertebrates.

Tissue Analysis

Inorganic and organic chemicals accumulate in the tissue of organisms through chronic exposure to polluted waters and sediments as well as through the ingestion of food. The goal of any evaluation of bioaccumulation is to relate body-residue concentrations of toxins to effects in aquatic species. Accomplishing this goal allows the response to toxins to be linked directly to sources of contamination. While many factors affect the bioavailability of toxins in the environment, it is the amount of exposure in the receiving species that causes the toxic response. Despite the potential usefulness of this approach, it has not been pursued much through development of standard test procedures and long-term studies (Burton and Pitt, 2002).

An alternative to the use of test species are semi-permeable membrane devices (SPMDs). SPMDs are polymeric tube bags containing a lipid compound, which mimics uptake and concentration of contaminants in living tissue. Comparisons of concentrations of target compounds in SPMDs and test species have shown that they often provide similar results (Burton and Pitt, 2002).

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CHAPTER 8

Impact Avoidance

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Avoiding impacts to aquatic ecosystems can only be achieved through careful site design and implementation of source control and management practices. However, any one stormwater management tool is unlikely to achieve the stormwater management objectives for a given development on its own. It is therefore necessary to consider the objectives early in the design process, when competing demands can be carefully balanced and an integrated approach achieved. This approach reduces both the need for and the size of treatment devices, as well as their construction and maintenance costs and obligations. Achieving an overall stormwater management objective by using a treatment train approach that combines a number of different tools or practices is essential to successful program implementation.

It is important to recognize that limiting hydrological modification is just as important as water quality treatment if aquatic resource protection is to be achieved. Hydrologic change also influences the whole range of environmental features that affect aquatic biota – flow regime, aquatic habitat physical structure, water quality, biotic interactions, and food sources. In addition to the hydrological change resulting from urban development, there are changes to the runoff delivery system. Soil compaction or impervious surfaces convert what was once subsurface groundwater flow to overland surface water flow. Thus the precipitation over a small watershed reaches the stream with a typical delay of just a few minutes, instead of what once was a lag of hours, days, or even weeks. The result is a dramatic change in flow patterns in the downstream channel, with the largest flood peaks doubled or more and the more frequent storm discharges increased as much as tenfold.

Many of the effects of stormwater are, by themselves, relatively small. When considered on a watershed basis, however, their cumulative effect is substantial, such as in the case of flooding due to gradual increases in upstream impervious areas. To manage these effects, we need to understand them on a watershed basis, where the effects are discernible, but prevent them on an individual site basis, where the physical changes to the hydrological cycle are made. This is the role of watershed management plans: as a range of approaches to achieve overall watershed objectives, they are a key tool for integrated stormwater management.

This chapter is divided into two major components: site (and watershed) resource protection and enhancement, and source control. The section on site resource protection and enhancement includes discussion of riparian buffers, upland forests, wetlands, steep slopes, and the importance of soils. The section on source control provides discussion of site practices, including low impact design to avoid adverse impacts from residential and commercial areas.

It is important to reiterate that both components can and should be applied on a site-by-site basis as well as on a watershed basis to provide maximum downstream aquatic system protection.

The final section of this chapter sets out information on various new approaches.

Site Resource Protection and Enhancement

The site resources referred to here are those natural features or site characteristics that, to a large extent, provide a benefit to receiving systems just by existing. They serve the general public by continuously reducing peak rates and volumes of stormwater runoff, provide water quality treatment, and prevent damage to improved or natural lands either on site, where the site resources exist, or downstream of those resources.

Site resources have intrinsic and other values for habitat and biodiversity beyond their stormwater functions. These include a wide variety of items, but those discussed here are considered primary resources that should be recognized and considered in site development and use. The following site resources (some of which are less obvious than others) are considered important primarily for their stormwater management benefits. They will be discussed in further detail in the following sections.

- Terrestrial ecology and landscape form;
- Headwater streams;
- Wetlands;
- Floodplains;
- Riparian buffers;
- Vegetation;
- Soils;
- Slopes/topography;
- Other natural features; and
- Linkage with site development.

Site resources often overlap. For example, a riparian buffer may lie within a floodplain, or, conversely, a forested area may form part of a riparian buffer. In this chapter, they are discussed individually, although their benefits may be, and generally are, overlapping and cumulative.

Most readers are probably already familiar with the following terms and discussion. Recognition of their values cannot be overemphasized, but they are often overlooked during the site development phase. Too often, we consider "low impact" approaches such as swales, bioretention, infiltration, or rain gardens to be stormwater management practices. Those practices impact the natural environment less than more traditional practices such as ponds, but they are only part of the solution. There has to be a fundamental shift in how we use land if we are to protect aquatic resources. Too often, we shape the land to fit a style of development. We have to start thinking about how to shape our use to fit the land. Any approach is only as strong as its weakest component. Doing half of what is needed is good from an evolutionary context, but resource protection may not result.

Terrestrial Ecology and Landscape Form

Where natural features are located on a site is just as important as the characteristics of the natural features themselves. There are several basic principles of ecology that can be used to improve the quality of receiving environments. These principles, detailed below, apply to all site resources:

- Retain and protect native vegetation (forest, regenerating forest, wetlands) – these ecosystem types have important intrinsic values and provide different habitats for native flora and fauna as well as different ecological functions.
- Allow natural regeneration processes to occur (e.g., pasture => scrub => forest; wet pasture => wetland).
- Undertake weed and pest control of invasive, non-native species to improve the natural succession potential of native vegetation. Allow natural processes and seed dispersal mechanisms to occur.
- Replant and restore with native plants to provide vegetation cover that is characteristic of what would once have been there and/or that reflects other local remnants in the area.
- Restore linkages with other natural areas or ecosystems (e.g., by using waterways and riparian areas, linking fragmented forest remnants, linking wetland ecosystems and freshwater ecosystems to terrestrial forest/scrub remnants).
- Our knowledge is limited, and we do not know what we are doing yet. When we design a bridge or a building, we include a factor of safety. We do not normally include a factor of safety for

the environment, and that lack of additional protection is reflected in the continuing decline of aquatic resources.

It is important to retain natural areas (including scrub, forest, and wetlands) on a site for their biological diversity and intrinsic values, which include the following:

- They are important as characteristic examples of biodiversity in a region or district.
- They contain a diversity of species or ecosystem types.
- They may contain rare or special features or unusual ecosystem types.
- They are valuable as habitats for native species.
- They have the ability to sustain themselves over time (e.g., through available seed sources, active regeneration, the level of weeds and pests and outside influences controlled).
- They are of adequate size and shape to be viable.
- They provide a buffer to habitats or natural areas from outside influences. These may include scrub on edges of native wooded areas or intact sequences from estuarine to terrestrial, from freshwater to terrestrial, from valley bottom to ridge top. They also provide linkages with other natural areas (corridors for birds or invertebrates).

Long-term ecological viability is the ability of natural areas to retain their inherent natural values over time. This includes the ability of a natural area to resist disturbance and other adverse effects, as well as the ability of its component plant and animal species to regenerate and reproduce successfully. Complex ecosystems often have a messy or "wild" appearance to them as their complexity increases. A mature forest can take hundreds of years to develop, so seeing one indicates a lack of recent disturbance.

Headwater Streams

A stream is a natural body of water that includes a free-flowing area of concentrated flow, an area having pools of water, a spring outfall, and/or a wetland. In the context of a stream, the area of concentrated flow has defined banks and bottom, not including areas of sheet or shallow concentrated flow such as swales.

Nationwide, as shown in Table 8-1, 73 percent of the total length of all streams in the U.S. are first- and second-order streams. These streams tend to be filled

Example of wetlands and native wood areas.

in, enclosed, and developed over. If one of our goals is to protect third-order and larger streams, that goal cannot be attained if first- and second-order streams are destroyed. Imagine, if you will, that 73 percent of your arteries were clogged or significantly impaired: your state of health could certainly never be considered good. Unless first- and second-order streams are protected, expectations for larger streams have to be reduced. This thought could have a substantial impact on how we develop land and on our normal practice of enclosing first-order streams to allow development to proceed on top of what was once a natural system. Once we enclose a stream and build on top of it, that stream is gone. How many times have all of us remembered a stream that we played in as children and found that stream now gone? It is a sad but recurring story.

Wetlands

Wetlands include permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals adapted to wet conditions. They occur on water margins or on land that is temporarily or permanently wet. Wetlands are a major habitat for freshwater fish as well as for frogs, birds, and invertebrates. Almost half (42 percent) of the total U.S. threatened and endangered species depend upon wetlands for survival. Wetlands have unique hydrological characteristics that can be irreversibly modified by activities such as drainage.

There can be few other vegetation classes that have suffered as severely during human times as wetlands have. There are many reasons for this, mainly related to the fact that wetlands are on flat land, suited to agriculture, and generally display a vegetation that is held in low esteem by the average person. These changes have occurred despite the value of wetlands as wildlife habitats, regulators of flooding, their intrinsic values, and their benefits for recreation and scientific research. Nevertheless, a far larger area than that remaining today has been lost through drainage or filling.

The vast majority of wetlands are less than 10 acres in size. It is also reasonable to assume that there are many more in the less than two-acre category, but they are generally not reported. This is especially true in headwater areas of watersheds, where very small wetlands may be present.

It is important to recognize that even without the presence of humans, wetland systems are being modified and eliminated by a natural ecological ageing process: succession. The filling and conversion of wetlands into more terrestrial types of ecosystems occurs naturally, but at a relatively slow rate. The intervention of humans into the process accelerates this conversion process from a period of hundreds of years to a very short time frame that can be measured in years.

In addition to the beneficial values shown in Table 8-2, the above list can be expanded to incorporate stormwater quality treatment. Natural systems have

Table 8-1: Relationship Between Stream Order and Other Stream and Floodplain Measures for Nontidal Streams of the United States; Meters (m), Kilometers (km)

Stream					Floodplain	
Order	Number	Length (km)	%	Cum %	Width (m)	Area (km^2)
1	1570000	2526130	48.4	48.4	3	7578
2	350000	1295245	24.8	73.2	6	7771
3	80000	682216	13.1	86.3	12	8187
4	18000	347544	6.7	92.9	24	8341
5	4200	189218	3.6	96.5	48	9082
6	950	97827	1.9	98.4	96	9391
7	200	47305	0.9	99.3	192	9083
8	41	22298	0.4	99.7	384	8562
9	8	10002	0.2	99.9	768	7682
10	1	2896	0.1	100.0	1536	4448

Source: Brinson, 1993

complex mechanisms, and the following list describes the major processes occurring in wetlands that allow them to provide water quality enhancement functions. These mechanisms include:

- Settling/burial in sediments;
- Uptake of contaminants in plant biomass;
- Filtration through vegetation;
- Adsorption on organic material;
- Bacterial decomposition;
- Temperature benefits; and
- Volatilisation.

Floodplains

Floodplains occupy those areas adjacent to stream channels that become inundated with stormwater during large rainfall/runoff events. For the most part, rainfall is the main cause of flooding, although surges by wind-driven currents can exacerbate the problem or (in unique situations) actually cause the flooding problem. Flooding problems result from two main components of precipitation: the intensity and duration of rainfall, and its areal extent and distribution.

The form of the stream channel and its associated floodplain in part determine the size of the flood, particularly its depth and areal extent. A small watershed and wide floodplain will result in a shallow, but widespread flood. A deep channel and steep slopes, on the other hand, will result in deeper flooding, but on a small areal extent.

The many benefits that floodplains provide are partly a function of their size and lack of disturbance. But what makes them particularly valuable ecologically is their connection to water and the natural drainage systems of wetlands, streams, and estuaries. The water quality and water quantity functions they provide overlap with the landscape functions of tract size and ecosystem complexity to make them exceptionally valuable natural resources.

Table 8-2: Summary of Wetland Functions and Values

Function/Value	Description
Flood control	Attenuation of peak flows
	Storage of water
	Absorption by organic soils
	Infiltration to groundwater
Flow augmentation	Maintenance of streamflow during drought
Erosion control	Increased channel friction
	Reduction in stream velocity
	Reduction in stream scour
	Channel stability through vegetative roots
Water quality	Sedimentation
	Burial of pollutants in sediments
	Adsorption of pollutants to solids
	Uptake by plants
	Aerobic decomposition by bacteria
	Anaerobic decomposition by bacteria
Habitat for wildlife	Food
	Shelter/protection from predators
	Nursery area for early life stages
Fisheries habitat	Freshwater mussels, crayfish, fish
Food chain support	Food production (primary production)
Recreation/aesthetics	Enjoyment of nature
Education	Teaching, research

Floodplains provide flood storage and conveyance during periods when flow exceeds channel boundaries. In their natural state, they reduce flood velocities and peak flow rates by out of stream bank flow of stormwater through dense vegetation. They also promote sedimentation and filter pollutants from runoff. In addition, having a good shade cover for streams provides temperature moderation of streamflow. Maintaining natural floodplains also promotes infiltration and groundwater recharge while increasing or maintaining the duration of low surface streamflow. Another function of floodplains is the temporary storage of floodwaters. If floodplains were not protected, development would, through placement of structures and fill material in the floodplain, reduce their ability to store and convey stormwater when necessary. This, in turn, would increase flood elevations upstream of the filled area and increase the velocity of water traveling past the flow area that has been reduced by fill material. Either of these conditions could cause safety problems or significant damage to private property.

Natural floodplains are fertile and support a high rate of plant growth that in turn supports and maintains biological diversity. They provide breeding and feeding grounds for fish and wildlife and habitat for rare and endangered species.

Ground cover in natural wooded floodplains tends to be composed of leaf and dense organic matter. Organic soils have a lower density and higher water-holding capacity than mineral soils, due to their high porosity or the percentage of pore spaces. This porosity allows floodplain soils generally to store more water than mineral soils would in upland areas.

Riparian Buffers

Although reduction of pollutants can be a function of riparian buffers, they also contribute significantly to other aspects of water quality and physical habitat. Habitat alterations, especially channel straightening and removal of riparian vegetation, continue to impair the ecological health of streams more often and for longer time periods than do pollutants.

When considering riparian buffer systems, it is helpful to examine the variety of benefits that are gained by their protection or implementation.

Example of a Small Headwater Wetland

Temperature and Light

The daily and seasonal patterns of water temperature are critical habitat features that directly and indirectly affect the ability of a given stream to maintain viable populations of most aquatic species. Considerable evidence shows that the absence of riparian cover along many streams has a profound effect on the distribution of a large number of species of macroinvertebrates and fish.

In the absence of shading by a forest canopy, direct sunlight can increase stream temperatures significantly (up to 12° C), especially during periods of low streamflow in summer. Riparian buffers have been shown to prevent the disruption of natural temperature patterns as well as to mitigate the increases in temperature following upstream deforestation.

Habitat Diversity and Channel Morphology

The biological diversity of streams depends on the diversity of habitats available. Woody debris is one of the major factors in aquatic habitat diversity. It can benefit a stream by:

- Stabilizing the stream environment by reducing the severity of the erosive influence of streamflow;
- Increasing the diversity and amount of habitat for aquatic organisms;
- Providing a source of organic carbon; and
- Forming debris dams and slowing stream velocities.

Loss of the riparian zone can lead to loss of habitat through stream widening, where forest is not replaced by permanent vegetation, or through stream narrowing, where forest is replaced by grass. In the absence of woody vegetation, bank erosion and channel straightening can occur. The accelerated streamflow velocity allowed by straight channels promotes channel erosion that may exceed the overland sediment load entering the stream. This process can eventually lead to the development of wide, shallow streams that support fewer species.

Food Webs and Species Diversity

The two primary sources of natural food energy input to streams are litterfall from streamside vegetation and algal production within the stream. Total annual food energy inputs are similar under shaded and open canopies, but the presence or absence of a tree canopy has a major influence on the balance between litter input and primary production of algae in the stream.

Example of a wooded floodplain in a local park

Having a stream exposed to sunlight for most of the day promotes algal growth and proliferation of algal grazing species. This proliferation reduces species diversity. The diversity of the macroinvertebrate community in a stream protected by a riparian buffer is much greater than the diversity of a stream that does not have a riparian canopy. This diversity is important, because it occurs in a very small area that goes from lowland wetter soil conditions to upland fairly rapidly and thus promotes very different vegetative types. Also, riparian buffer areas are adjacent to streams and therefore to floodplains. Through periodic out-of-bank flow, floodplains are depositional zones for fertile sediments, which is why areas adjacent to streams have always been considered so productive from an agricultural perspective.

Pollutant Removal

Riparian vegetation removes, sequesters, or transforms nutrients, sediments, and other pollutants. The removal function depends on two key factors:

- The capability of a particular area to intercept surface and/or groundwater-borne pollutants; and
- The activity of specific pollutant removal processes (filtration, adsorption, biological uptake, etc.).

Sediment trapping in riparian forest buffers is facilitated by physical interception of surface runoff that causes flow to slow and sediment particles to be deposited. Channelized flow is not conducive to sediment deposition and can, because of its higher velocities, cause erosion in the riparian buffer.

Channel Stability and Flood Flow Protection

Streams are dynamic systems that are characterized by change. In-stream stability and stream bank erosion at a given point are heavily influenced by the land use and condition in the upstream watershed. However, vegetation – especially woody vegetation – is essential for stabilizing stream banks. Forested buffer strips have a direct effect on stream bank stability by providing not only deep root systems that hold the soil in place more effectively than grasses, but also a degree of roughness capable of slowing runoff velocities and spreading flows during large storm events. While slowing flood velocities may increase flood elevations upstream and in the buffer, downstream flood crest and damage may be significantly reduced. These processes are also critical for building floodplain soils.

Example of a small stream with riparian cover

Vegetation

Vegetation cover has changed considerably as man's influence on his environment has expanded, with the most dramatic changes occurring in the past century. Almost every kind of vegetation imaginable exists in the U.S., and each of them has suffered from our use of the land. As we have already discussed wetlands, it would be of value here to discuss forest land, which has a number of components whose characteristics determine its effectiveness in terms of water quantity and quality.

Stormwater Runoff Reduction

Woody vegetation and forest floor litter have a significant impact on the total volume of rainfall converted to runoff. Runoff volumes from forested areas are much lower than volumes from other land uses. This lesser volume in runoff acts to minimize downstream erosion and instability problems. This can clearly be shown by some of the runoff curve numbers listed in Table 8-3 that are provided in the Natural Resources Conservation Service (NRCS) Technical Release No. 55 for various land uses. Some of those curve numbers are provided here to demonstrate the differences that vegetation variety has, in conjunction with soil conditions, on curve numbers. The higher the curve number the greater the runoff.

Using the curve number approach, relationships can be drawn (hypothetically) regarding the amount of forest that would have to be planted to compensate for imperviousness. That ratio, depending on soils and slopes, can be approximately 6:1, meaning that it takes six times as much wooded area to compensate for a given amount of impervious surface. This considers only volume, but prevention of concentrated flow, absence of soil compaction due to development, etc. also need to be considered.

Soil Structure

Forest soils are generally regarded as effective nutrient traps. Most nutrients are retained (and recycled) in the leaf litter and shallow soil layers. The ability of a forest soil to remove nutrients in surface and groundwater is partially dependent on soil depth, ground slope, density of vegetation, permeability, extent and duration of any shallow water table, and its function as a groundwater discharge zone.

Organic Litter Layer

The organic litter layer in a forest buffer provides a physical barrier to sediment movement. It also maintains surface porosity, higher infiltration rates, increased populations of soil mycorrhizae (a symbiotic relationship of plant roots and the mycelium of fungi that aids in decomposition of litter and translocation of nutrients from the soil into the root tissue), and provides a rich source of carbon essential for denitrification. The organic soil provides a reservoir for storage of nutrients to be later converted to woody biomass.

A mature forest can absorb as much as 14 times more water than an equivalent area of grass. The absorptive ability of the forest floor develops and improves over time. Trees release stored moisture to the atmosphere through transpiration, while soluble nutrients are used for growth.

Table 8-3: Curve Numbers for Various Land Covers

Cover type/land use	Hydrological condition	A	B	C	D
Impervious areas		98	98	98	98
Woods	poor – no forest litter	45	66	77	83
	good – litter and brush	30	55	70	77
Pasture	good	39	61	74	80
Lawns/open space	good – full grass cover	39	61	74	80

Forested Areas

Trees have several advantages over other vegetation in improving water quality. They aggressively convert nutrients into biomass. They are not easily smothered by sediment deposition or inundation during periods of high water level. Their spreading root mats resist the development of gullies and stimulate biological and chemical soil processes. They produce high amounts of carbon needed as an energy source for bacteria involved in the denitrification process. A forest's effectiveness in pollution control will vary with the age, structural attributes and species diversity of its trees, shrubs and understory vegetation.

To consider the involvement of a forested area in water quality treatment, there are a number of functions that define that performance. These functions can be broadly defined as physical and biological functions and include the following:

Physical Function

The forest floor is composed of decaying leaves, twigs, and branches that form highly permeable layers of organic material. Large pore spaces in these layers catch, absorb, and store large volumes of water. Flow of stormwater through the forest is slowed down by many obstructions. Suspended sediment is further removed as runoff flows into the vegetation and litter of the forest floor. This sediment is readily incorporated into the forest soil. With a well-developed litter layer, infiltration capacities of forest soils generally exceed rainfall and can also absorb overland flows from adjacent lands.

Biological Function

Forest ecosystems serve as filters, sinks, and transformers of suspended and dissolved nutrients. The forest retains or removes nutrients in a variety of ways. It rapidly incorporates biomass, stores it long term, improves soil nutrient holding capacity by adding organic matter to the soil, reduces leaching of dissolved nutrients in subsurface flow from uplands by evapotranspiration, provides bacterial denitrification in soils and groundwater, and prevents erosion during heavy rains.

Soils

Soils possess several outstanding characteristics as a medium for life. They are relatively stable structurally and chemically. The underground climate is far less variable than above-surface conditions. The atmosphere remains saturated or nearly so, until soil moisture drops below a critical point. Soil affords a refuge from high and low extremes in temperature, wind, evaporation, light, and dryness. These conditions allow soil fauna to make easy adjustments to the development of unfavorable conditions. On the other hand, soil hampers movement. Except for organisms such as worms, space is important. It determines living space, humidity, and gases.

A wide diversity of life is found in the soil. The number of species of bacteria, fungi, protists, and representatives of nearly every invertebrate phylum found in soil is enormous. It has been estimated that approximately 50 percent of the earth's biodiversity occurs in soil. Dominant among the soil organisms are bacteria, fungi, protozoans, and nematodes.

As detailed by the NRCS soil classification systems, all soils are contained within the following four categories:

- Group A soils have low runoff potential and high infiltration rates even when thoroughly wetted. They consist chiefly of deep, well to excessively drained sands or gravels and have a high rate of water transmission (greater than 0.3"/hour).
- Group B soils have moderate infiltration rates when thoroughly wetted and consist chiefly of moderately deep to deep/moderately well to well drained soils with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission (0.15 – 0.3"/hour).
- Group C soils have low infiltration rates when thoroughly wetted and consist chiefly of soils with a layer that impedes downward movement of water and soils with moderately fine to fine texture. These soils have a low rate of water transmission (0.05 – 0.15"/hour).
- Group D soils have high runoff potential. They have very low infiltration rates when thoroughly wetted and consist chiefly of clay soils with a high swelling potential, soils with a permanent high water table, soils with a claypan or clay layer at or near the surface, and shallow soils

over nearly impervious material. These soils have a very low rate of water transmission (0 – 0.05"/hour).

Soils having greater infiltration rates also have reduced runoff potential. From a groundwater recharge and stormwater runoff perspective, prioritizing development on Group C and Group D soils would be more desirable than allowing development on Group A or Group B soils. If the overall development density was set at a given level, clustering that development on poorer soils would result in less of an increase in stormwater runoff than development on soils with greater infiltration rates. This would also have significant beneficial effects on groundwater recharge that could help maintain stream baseflows as a watershed develops. Maintaining highly permeable soils in open space areas would provide a better approach to baseflow maintenance than artificial infiltration in smaller selected areas that would come with long-term maintenance concerns.

Slopes/Topography

Steeper slopes also increase the erosion potential of the soil. Looking at the Universal Soil Loss Equation (USLE) (now the Revised Universal Soil Loss Equation – RUSLE), you can gain an understanding of the importance of slope in the calculation of soil loss. The slope length factor (LS) demonstrates that there is a direct relationship between slope and calculated soil loss. The Universal Soil Loss Equation (USLE) is a simple empirical formula that was developed approximately 30 years ago and derived from the theory of erosion processes. The general form of the equation is:

$$A = (R)(K)(LS)(C)(P)$$

where:

A = Calculated soil loss (tons/ha)

R = Rainfall energy factor

K = Soil erodibility factor

LS = Slope-length factor

C = Cropping management (vegetative cover) factor

P = Erosion practice factor.

The greater the slope, the greater the soil loss. In calculating the LS term, LS is based upon the length and steepness of a given slope. If the slope length is kept constant, its doubling (log-log relationship) causes the LS factor to approximately double. This means that a slope of 2 percent (100 m length) has an LS factor of 0.29, where a slope of 4 percent has an LS factor of 0.6. A slope of 16 percent has an LS factor of 5. A slope of 16 percent has 17 times the soil loss of a 2 percent slope, all other factors being equal. In other words, disturbance of steep slopes has a dramatic impact on site soil loss.

By identifying steeper slope areas in the initial stages of project planning for a new development, portions of a site that have increased potential for erosion can be identified. This process would allow for site development to occur in a less destructive manner or for more stringent erosion and sediment control practices to be implemented during site development.

Other Natural Features

Every site has natural features that would have substantial stormwater management benefits if they were integrated into the development approach. The previously discussed ones are important individually, but there are others that are important as well and should be integrated to provide a better site management approach.

Depression Storage and Evapotranspiration

Of the rainfall that strikes roofs, roads, and pervious surfaces, some is trapped in the many shallow depressions of varying size and depth present on practically all ground surfaces. The specific magnitude of depression storage varies from site to site. Depression storage commonly ranges from 1/8 to 3/4 inches for flat areas and from 1/2 to 1-1/2 inches on grasslands of forests. Significant depression storage can also exist on moderate or gentle slopes with some estimation for pervious surfaces being between 1/4" to 1/2" of water and even more on meadows and forest land. Steeper slopes would obviously have smaller values.

When using traditional hydrologic procedures, depression storage is contained in an initial abstraction term. This term includes all losses before runoff begins. It includes water retained by vegetation, evaporation, and infiltration. It is highly variable but generally correlated with soil and cover parameters.

Prior to urbanization, watersheds have a significant depressional storage factor. The urbanization process generally reduces that storage in addition to significantly modifying the land's surface. The combination of site compaction, site imperviousness, and reduced depression storage causes dramatic increases in downstream flood potential and channel erosion.

Information from one watershed study indicated that long-term average annual predicted runoff varied from less than 12" (18 percent of rainfall) to greater than 24" (greater than 35 percent of rainfall). The 12" coincided with subwatersheds under permanent forest cover, while the 24" coincided with subwatersheds in predominantly agricultural land use and on low infiltration soils. There is a clear statement in these statistics that significant volume reductions in runoff exist in forested watersheds compared to volumes of runoff from agricultural land cover.

The infiltration of water into the surface soil is responsible for the large abstraction (loss) of rainwater in natural areas. The infiltration capacity of most soils allows low-intensity rainfall to totally infiltrate, unless the soil voids become saturated or the underlain soil is much more compact than the top layer. High-intensity rainfalls generate substantial runoff because the infiltration capacity at the upper soil surface is surpassed, even though the underdrain soil might still be dry.

Natural Drainage Systems

Natural site drainage features exist on every site. The most common of these features is an already existing flow path for stormwater runoff. Water doesn't travel down a hill in a straight line. Straight channels or pipes are something that humans have developed to accelerate the passage of water downstream as quickly as possible. During site development, the tendency is to place water in conveyance systems, open or enclosed, which follow the shortest distance to site outfalls.

Shortening the flow distance effectively increases the slope that water travels on, accelerates the flow of water, and increases the ability of water to scour downstream receiving systems. When water travels over a meandering flow path, energy is dissipated, which reduces the erosion potential. Shortening flow lengths reduces energy expended and increases the available erosion-producing energy. Stream channels will meander regardless of the degree of human alteration. Replicating existing flow paths and lengths to the extent possible promotes channel stability and increases function and value.

The additional functions provided by meandering channels in comparison to straight channels are also simply related to the length of the aquatic resource and the time that the water is in contact with the various biotic and abiotic processing mechanisms: the additional length of meandering channels provides a greater total quantity of aquatic resource and its associated functions and values.

Soil Compaction

Areas have increased runoff after development for a number of reasons. The most important cause is usually the increased amount of pavement and roof areas. However, urban soils also undergo major modifications that result in increased runoff. These soil modifications may mostly affect infiltration, but other soil changes also occur. Specifically, reductions in the organic content of the surface soil layers and removal of plants will reduce the evapotranspiration losses and contribute to increases in runoff. This is especially important in areas where surface soils are relatively shallow and located above impermeable layers.

The soil compaction during construction and use likely causes most of the reduced infiltration capacity of urban soils. In addition, many more subtle changes will also occur. Many of these changes contribute to the reduction of measured infiltration, such as the replacement of native plants that typically have much deeper root systems with shallow-rooted grasses. The removal of the native soils results in the removal of organic matter, mature and deep-rooted plants, and the soils themselves, often exposing a deeper soil material that is much less able to allow infiltration or evapotranspiration. There are a number of options to address this concern:

- In areas of significant site disturbance, and where there is less than three feet of cut or where cuts or fills of at least three feet are intended to facilitate site development, the expected permeability of the soil may be reduced. Stormwater management calculations, which detail post-construction hydrology, should use a modified approach to soil classifications.

- In areas of significant soil disturbance, and where there is less than two feet of cut or fill, soil classifications are not modified, but the approved permit should contain a construction requirement to the effect that significantly disturbed soils in areas where those soils remain pervious should be chisel-plowed. Chisel-plowing will break the surface crust of the disturbed soil and allow for a greater infiltration rate. This would provide a good foundation for the placement of topsoil and prevent topsoil slippage on slopes that become saturated.
- Avoid compaction altogether by keeping equipment out of areas preserved for open space.
- Making soil amendments, or otherwise modifying soil structure and chemical characteristics, is becoming an increasingly popular stormwater management practice. However, little information is available to quantify benefits and problems associated with their use.

Linkage with Site Development

The above natural site features all provide stormwater management benefits if considered and integrated into the initial site development plan. They cannot be considered an afterthought of the site planning stage. If stormwater considerations are neglected until the overall site plan has been developed, there are too many site conflicts to provide an effective site management plan that protects aquatic resources. At this point, issues related to levels of imperviousness, location of utilities, and lot layouts prevent integration of site features into a development approach, and it is too late for aquatic resource protection.

It is vital that a pre-development site inventory plan be drafted and submitted with the site development and stormwater management plan. It would also be advantageous to have a narrative submitted that details what steps have been taken to incorporate natural site features into the site development plan. This approach will require a rethink away from the traditional site development approach that has existed for many years.

Source Control

Source control is often considered in a traditional context such as industrial site source control. At the same time, there are now other source control components that aim to eliminate the source of a pollutant from potentially entering the receiving system and control impacts on any kind of land use. This approach constitutes a considerable expansion of the traditional use of the term.

Traditional Source Control

Source control and management procedures attempt to reduce or avoid pollutants getting entrained in stormwater runoff. These practices assume that the pollutant source is necessary for the successful operation of the business or activity, and seek to control the release of pollutants or remove them before they come into contact with stormwater. For example, service stations inherently use trade oils and gasoline as their main business activity, but they are required to cover the service area and shut off stormwater pipes during tanker deliveries to prevent the discharge of petroleum products to the environment via stormwater drains.

The EPA advocates that businesses that handle chemicals or produce wastewater carry out an environmental audit to identify actual and potential pollutant sources. An action plan should then be developed to eliminate any actual pollution and minimize the risk of potential pollution.

Source control practices identify pollutant sources and construct physical works to prevent them from coming into contact with stormwater. The classic example is the above ground storage tank with a berm constructed around it. The berm volume is greater than the volume of the storage tank.

Other examples include:

- Physical control structures such as berming, spill containment;
- Covering stockpiles of soil, waste products;
- Directing washwater to sanitary sewers; and
- Covering "dirty" work areas such as truck washes or oil changing bays.

Numerous procedures can be designated as management practices, from local government initiatives to regularly removing gutter dusts before they get entrained in stormwater to industrial protocols for handling chemicals. The common factor is that there is a process to be followed that minimizes the risk of pollutant transfer to stormwater.

Local government initiatives include:

- Street vacuuming;
- Education initiatives; and
- Recycling.

Industry initiatives include:

- Refueling procedures;
- Chemical handling procedures;
- Staff training regarding proper disposal areas for wastes, chemicals, etc.; and
- Proper storage of chemicals, fuel etc.

Significant information on traditional source control is available from the EPA website, so further detail is not being provided here.

Eliminating the Pollutant Source

When considering a given pollutant, it is becoming more recognized that treatment represents the "ambulance at the bottom of the cliff." Treatment cannot be expected to remove all pollutants of concern nor to totally eliminate a particular one. Questions are increasingly being asked as to where specifically a pollutant found in a watershed is coming from. It may be that removing the pollutant source is more economical than attempting to remove the pollutant once it is in the water column.

A good example of a source control was the removal of lead from gasoline in the 1980s, a source control activity that has led to the reduction of lead levels in receiving environments. In the same regard, older roofs were recognized in the Baltimore NURP study as being a significant source of copper. Painting those roofs, using a different material if the roof has to be replaced, or preventing new copper roofs would be an effective approach to copper reduction. The Chesapeake Bay Program has been addressing nutrients and has targeted phosphorus for years. An effective approach to phosphorus reduction has been the elimination of phosphorus from detergents.

More and more we have to ask ourselves why a certain pollutant is being found. It may be that we can't eliminate it from a local context and must consider either regional or national initiatives to eliminate the source, but changes can only be made if we understand cause and effect. It is expected that significant efforts will be expended to consider more benign materials from a water quality perspective.

Source Control in the Broader Context

Sir Isaac Newton (1643-1727) theorized in one of his laws that "for every action there is an equal and opposite reaction." This law is very true, and the only way to reduce or eliminate reactions is to reduce or eliminate actions. The whole premise behind source control in the broader context is to reduce actions and thereby reduce the inevitable reactions. Using or disturbing less of a site results in less potential downstream impact. There are a number of names given to this approach, including low impact design, conservation design, water sensitive urban design, or sustainable urban drainage systems. These will be discussed in more detail later in the chapter.

Each proponent of a given approach will claim that theirs is the most encompassing of all essential elements, but the argument revolves more around details than around concepts. There are, however, a number of essential components that are needed in all of the approaches. They include the following:

- Reducing site disturbance;
- Reducing impervious surfaces;
- Distributing flow and reducing efficiency of flow conveyance;
- Implementing integrated stormwater management;
- Creating or protecting natural areas;
- Clustering development; and
- Reusing water (where possible).

Each of these items will be discussed individually with regard to their importance in the overall context. Having only one or several of these elements in place will provide a benefit, but overall resource protection will need to incorporate all of them to maximize benefits. Even with all of these elements, it is necessary to consider those items listed earlier in the chapter relating to site resource protection and enhancement. In addition, there may still be a need to provide structural stormwater management.

Reducing Site Disturbance

There are two contexts to consider in a discussion of overall site disturbance:

- Construction-generated sediment loads; and
- Permanent stormwater issues related to quantity and quality of runoff.

Erosion and Sediment Control

Any discussion of erosion and sediment control has to break the term down into its two basic components, "erosion" and "sediment control." When land is disturbed at a construction site, the erosion rate increases with removal of ground cover, normally vegetative, which protects soils from erosion. The major problem with erosion is the movement of soil off-site and the subsequent impact of sedimentation on the receiving environment.

The high yield from the urbanizing catchment stems from the considerable portion of its ground area that is bared for construction (in one watershed, for example, approximately 28 percent at the time of the study). The yield from the sub-watersheds undergoing 100 percent construction was estimated to be approximately 16,800 t/km^2/yr, or hundreds of times the yield from undisturbed or stable areas of the watershed.

Reducing the limits of site disturbance by leaving steeper areas natural, not exposing erodible soils, and maintaining a vegetative cover can all reduce the amount of work that sediment control practices must do and reduce downstream sedimentation. A simple way to consider how site development can affect sediment yield is to once again look at the Revised Universal Soil Loss Equation.

$$A = R \times K \times (L\,S) \times C \times P$$

where:

A = Soil loss (tonnes/hectare/year)

R = Rainfall erosion index (J/hectare)

K = Soil erodibility factor (tonnes/unit of R)

LS = Slope length and steepness factor (dimensionless)

C = Vegetation cover factor (dimensionless)

P = Erosion control practice factor (dimensionless).

Clearly, changing the vegetative cover, or reducing area of disturbance or slope being disturbed all have a significant effect on soil loss.

By clustering development on a portion of a site while protecting critical areas, overall site disturbance is reduced, which in turn reduces sediment yield.

In addition to sediment as a pollutant, site disturbance will also affect the quantity of water that leaves a construction site. A clear example of this are again the NRCS runoff curve numbers for average antecedent runoff conditions shown in Table 8-4.

Table 8-4: Runoff Curve Numbers Compared to Construction Curve Numbers

Cover type/land use	Hydrological condition	A	B	C	D
Woods	poor – no forest litter	45	66	77	83
	good – litter and brush	30	55	70	77
Meadow	-	30	58	71	78
Lawns/open space	good – full grass cover	39	61	74	80
Bare soil	-	77	86	91	94

A simple TR-55 analysis can show that runoff is dramatically increased during the construction phase of site development. Sediment control practices are very seldom designed to provide water quantity control, especially for channel erosion. Thus, as can be seen, the greater the area of disturbance, the greater the peak discharge and total volume of runoff. If stream channel protection is a program goal, the erosion that the permanent stormwater system is intended to reduce or prevent (even if using permanent source controls) may occur prior to implementation of those permanent controls.

Permanent Stormwater Management

The effects of urbanization on soil structure can be significant. A common approach to site development is to clear most, if not all, of the site being developed. Existing vegetated areas are often cleared even when in non-essential locations. The clearing and grading of areas that will remain pervious results in significant compaction of those areas. This compaction reduces expected infiltration rates and increases overland flow.

In addition to soil structure, forested areas and wetlands should be seriously considered for retention if aquatic resource protection is a program goal.

Reducing Impervious Surfaces

There have been many studies relating impervious surfaces to aquatic system impact. We are seeing now (as mentioned in the introduction) that impervious surface considerations alone are an imperfect barometer of ecosystem health. But even so impervious surfaces have a profound impact on the generation of stormwater runoff, the conveyance of that runoff to drainage systems, and the subsequent problems that result from a water quantity and quality standpoint.

Stormwater considerations have to be an integral component of site development if impervious surface coverage is to be addressed at all. Items such as road widths, amounts of off-street parking spaces, driveway lengths, roof areas, and sidewalks must all be given careful consideration if impervious surfaces are to be reduced.

Another problem related to impervious surfaces is the creeping increase of impervious surfaces once land has been developed. Residential properties designed to have a maximum impervious surface of 50 percent may find subsequent levels of 70 percent or more as people increase driveway widths, patios, etc. This impervious surface area creep can have a significant effect on local drainage design if storm drain pipes are sized for a level of imperviousness that is exceeded.

Example of a very wide street

Distributing Flow and Reducing Efficiency of Flow Conveyance

The construction of efficient conveyance systems accelerates the flow of water from the top of a watershed to streams, estuaries, and coastlines. The traditional approach of catch pits draining into piped systems and rapid delivery to the receiving system accelerates flow dramatically beyond natural drainage conveyance and prevents potential water loss through infiltration.

Efficient conveyance was a drainage goal of the 1960s and early 1970s, when water was considered the common enemy. A common pattern of watershed development has been for development to occur from the mouth of the watershed upwards over time. This generally resulted from initial development depending on travel by water. This historical approach meant that upstream landuse was generally pasture, agriculture or forest having natural flow of drainage across the land and into streams. As the upstream watershed developed, downstream flows were increased and the delivery of stormwater downstream was accelerated.

That approach has evolved over the past 30 years, first to on-line ponds and later to off-line ponds. As we have looked closer at cause and effect, we are now realizing that "inefficiency" at the top of the watershed will provide significant benefits over the traditional approach. There is greater recognition that disconnecting drainage systems from their outlet can provide downstream benefits.

An example of possible disconnection is the consideration of curbing on streets. With a local requirement for curbing, flow is immediately concentrated adjacent to the curb and travels along the curb until it must enter a catch basin. As a result, stormwater flows are concentrated at the top of the watershed, delivered to a catch pit, and placed into a stormwater pipe whose outfall is into a receiving system or, hopefully, into a stormwater management structure. The net effect is an increase in peaks, volumes, and pollutant delivery downstream. We need to look for opportunities to disconnect drainage systems to provide better resource protection.

Implementing Integrated Stormwater Management

Stormwater management has historically been considered an afterthought and has to be integrated into the overall site development planning process. There are two key components of stormwater design: prevention and mitigation.

Prevention

Prevention includes land-use planning, not only on a site basis but taking into account the relationship between the individual site and the sub-watershed. This is an important context when considering wildlife corridors or riparian buffers. In all cases, the goal should be to use the simplest approach possible. This relies on the use of natural site features (wetlands, forested areas, meadows) in conjunction with source control practices such as green roofs, rain gardens, swales, filter strips,

Example of an "ancient" approach to drainage revegetation, and water reuse. The key element of these source control practices is to reduce the total volume of runoff while providing water quality treatment. While initial efforts in stormwater design were directed at centralizing runoff for control and treatment, newer concepts aim to disperse the runoff as much as possible. An example of this would be the elimination of curbing in a subdivision, which would tend to disperse flows rather than concentrating them. This is a significant shift in thinking.

Mitigation

Mitigation has clearly been the focus of program implementation over the past 20 years. Practices that fit into this category include ponds, wetlands, filter systems, and hydrodynamic separators. The use of these practices assumes that water quantity and water quality cannot be addressed through prevention. They tend to represent the "ambulance at the bottom of the cliff."

While prevention is the most desirable outcome, some mitigation will always be necessary. The key point with mitigation is to reduce to the extent possible the amount of work that must be performed. In addition, not all mitigation practices are created equal. A given development may primarily have suspended solids issues, and a pond may prove effective here, while nutrients, metals, or organics may be a primary consideration elsewhere. Mitigation needs to be considered in the context of the problem that it is addressing.

Having a good understanding of prevention and mitigation allows us to consider aspects of prevention that would reduce downstream effects of stormwater runoff.

Creating or Protecting Natural Areas

In many site development situations, the creation of a meadow as open space would have significant storm-water management benefits for both water quantity and water quality. If well designed and constructed, the area could become an attractive amenity to a community and enhance the value of the properties.

In a similar fashion, reforesting of steep slopes or protection of natural woody vegetation elsewhere on a site would have long-term benefits in a stormwater context. Whereas traditional stormwater practices clog or fill in over time, thus reducing their effectiveness, areas that have been revegetated become more effective over time and bring with them a reduced maintenance responsibility. Forested areas intercept rainfall and have an organic leaf and branch ground cover that acts to retain water. In addition, trees use and store nutrients for

Subdivision without curbing or catch pits long periods of time and, as discussed above, moderate temperatures during the summer.

As wildlife corridors could be re-established as well, there could be significant wildlife benefits.

Wetlands are also very valuable and productive ecosystems whose maintenance or enhancement would have significant benefits. These benefits, as discussed above, include flood control, low streamflow augmentation, erosion control, water quality, and habitat.

Clustering Development

As a source control tool, clustering is very important and may be considered, in conjunction with protection or enhancement of site resources, a keystone of the overall source control. Without clustering, protection of important site features is impossible. From a stormwater management standpoint, clustering minimizes stormwater and pollutant load generation and is clearly preventive in nature.

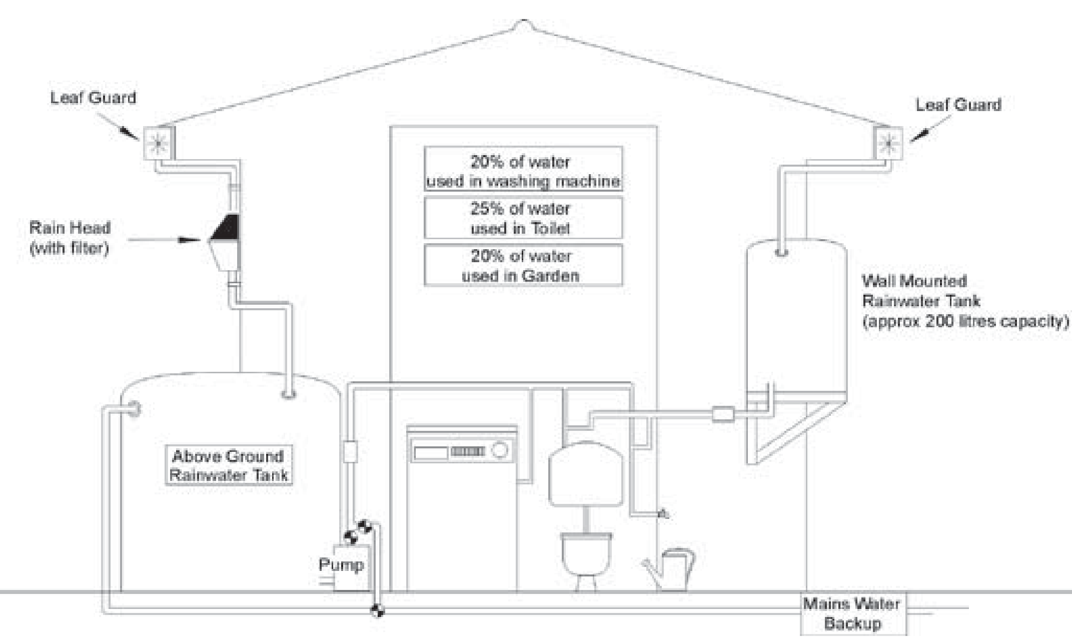
Although some density bonuses may be considered to encourage use of clustering, clustering in a strict sense usually begins after the decision on overall site density has been made. In some cases, clustering may provide different types of development, including single- and multi-family development. Clustering may involve lot design and arrangement only, or it may involve changing the types of residences. The challenge is to maximize benefits such as open space in conjunction with developer-desired outcomes.

Clustering benefits include:

- Reduced imperviousness;
- Reduced pollutant generation;
- Preservation of natural site values;
- Habitat and wildlife values;
- Passive recreation and open space amenities; and
- Cost reduction.

In consideration of clustering, it is important to overcome barriers to its use. These barriers may include minimum lot sizes, inflexibility by local jurisdictions, the time that the permitting process may take and uncertainty by developers that the proposal would be acceptable. Local government needs to revisit code requirements to facilitate implementation of innovative approaches.

Commonly used components of a water tank



Reusing Water

At this point, it would be beneficial to discuss the use of rainwater tanks as a stormwater management source control practice. They are primarily water quantity management devices but do have minor water quality benefits, depending on the amount of atmospheric deposition in a given area or the pollutant load that may result from the roof itself (zinc, copper, etc.). They have been used for centuries for supplying household and agricultural water.

Approximately 60 percent of domestic water use goes to toilet flushing, laundry, and garden watering. If roof runoff could be stored and used for those purposes, it would not contribute runoff during storm events and represent a reduced volume downstream. This could be part of an overall strategy for stormwater management in which roof areas do not contribute runoff. Benefits could be even greater for industrial sites that use water in their daily operations.

Rainwater tanks are not a stand-alone solution for quality and quantity issues in a watershed, but they can be implemented as a part of an integrated approach toward:

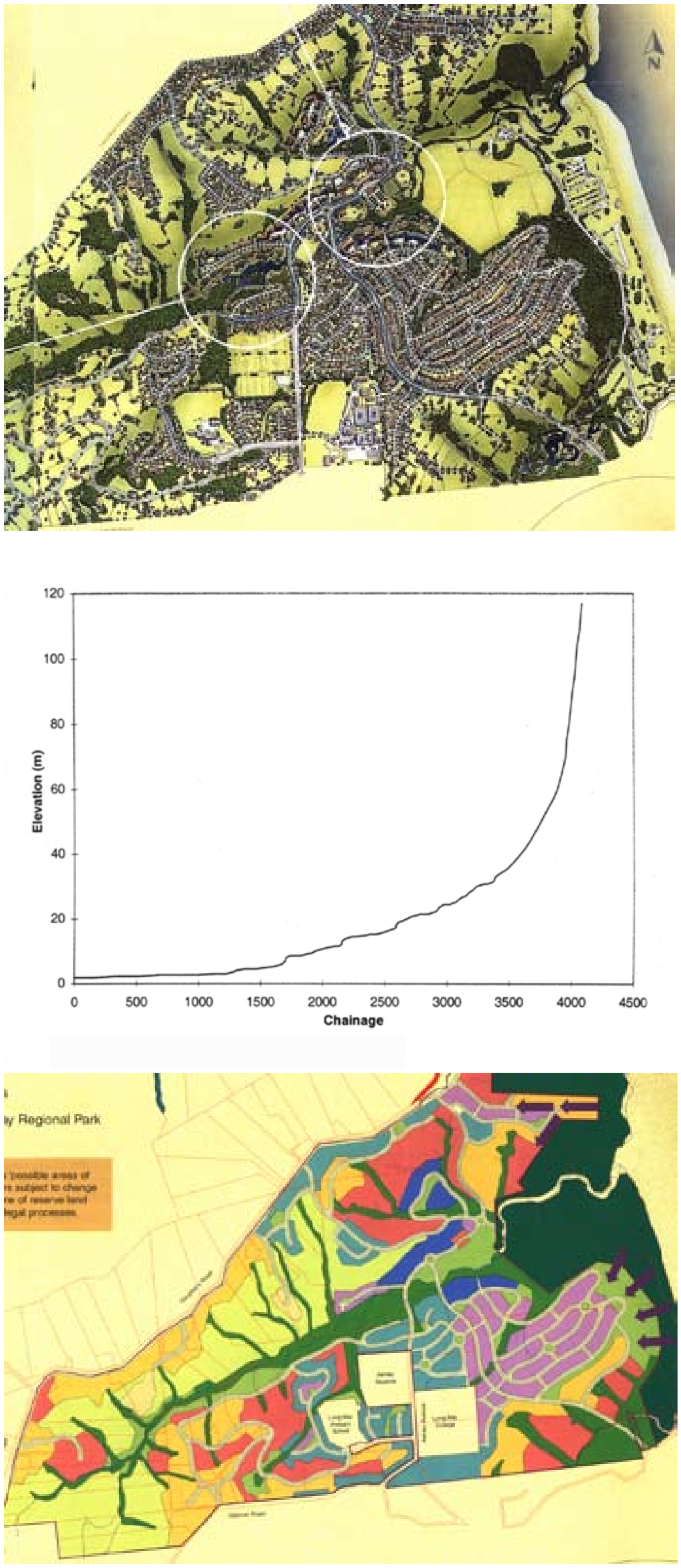
- Reducing stormwater volumes entering the receiving waters through the use of stormwater captured and used on site;
- Reducing flows into downstream stormwater treatment practices;
- Reducing peak stormwater flows from the sub-watershed by providing permanent or temporary storage;
- Reducing sanitary sewer overflows by reducing the rates and volumes of stormwater that enters directly or indirectly into sanitary sewers;
- Reducing roof-generated pollutants entering water bodies; and
- Reducing demand for potable water, which leads to more effective use of water resources.

Water tank capturing roof runoff

Rainwater tanks can be used in residential, commercial and industrial developments. The applications include the following:

- Treating roof runoff and accordingly reducing the size of the downstream treatment devices. In this case, the roof runoff, after storage in the tank system, would enter the receiving waters separately, while the ground runoff would be routed via the downstream treatment practice. An example of this would be industrial or commercial sites, where the roofs are treated by tanks while parking areas are treated by rain gardens or swales. Another example are high-density subdivisions, where tanks address roofs and the rest of the area is treated by wetlands.
- Managing stormwater in infill developments where the existing drainage system capacity is already exceeded for the design storm (generally 1 in 10 year capacity). There are different types of rainwater tanks to suit the available space and required volume.
- In conjunction with other practices, working toward hydrological neutrality in order to mitigate adverse effects of a development.
- Providing treatment, peak attenuation, and non-potable water supply benefits as multi-purpose devices. When coupled with adequate roof areas, they become financially self-supporting for reasonably large non-potable water demands.

Figure 8-1: Watershed Information – Intended Growth, Stream Slope, Sensitive Areas



An Example from a Watershed-Wide Context

While implementation of source controls on a site-by-site basis is essential, the optimum level for consideration of source control is on a watershed basis. If clustering is valuable on an individual site basis, its consideration on a watershed scale could have huge benefits in terms of existing resource protection and enhancement, while the desired level of development could still be accounted for. The following example shown in Figure 8-1 is put forth to demonstrate the value of source control from a watershed perspective. Expected development of this watershed will accommodate 8,000 new residents.

The headwaters of the watershed are on steeper land, and the intention is to protect first-order streams and avoid mass earth movement during the developing phase. As a result, density is approximately one house per acre. In other words, those areas have a lot yield based on an average density of one house per acre, but clustering is encouraged. Average lot sizes are expected to be approximately 20,000 ft^2, which means that overall site development will disturb about half of each site. The gullies and first-order streams are protected and revegetated where necessary. Site stormwater controls will be rain gardens, swales, filter strips, and water reuse for each house. Using this approach, traditional stormwater practices such as ponds or wetlands are not necessary.

Downstream of the headwaters, the land flattens out. All perennial streams are protected and riparian corridors re-established. Lots are approximately 10,000 ft^2, and development will take a more conventional approach, although rain gardens and water tanks will still be used on individual lots. Due to increasing imperviousness, stormwater management will take the form of constructed wetlands located on ephemeral watercourses that provide water quality and quantity control.

The lower area of the watershed will be intense development, including commercial, institutional, high-density residential, and town development. Again, all of the perennial streams are protected and riparian corridors established. Stormwater management in this portion of the watershed will use constructed wetlands as the primary stormwater practice. Upon completion of the development approach, there will be pedestrian movement adjacent to the riparian corridors throughout the watershed.

A major focus of this approach is the protection and enhancement of aquatic resources. The watershed has historically been pasture with significant stock access to streams that has severely impacted on aquatic resources. Through substantial revegetation throughout the watershed, especially in the headwaters, removal of fish blockages where they currently exist, and implementation of stormwater management throughout the watershed while accommodating significant urban growth, it is hoped that aquatic resource values can improve as urbanization occurs.

Various New Approaches

There are a variety of approaches around the world that have a similar foundation in minimizing our impact on receiving systems. They all differ in certain aspects that would make them interesting to investigate further.

Low Impact Design (LID)

LID's basic tenet is to create a hydrologically functional landscape that mimics the natural hydrologic regime. This objective is accomplished by:

- Mimimizing stormwater impacts to the extent practicable. Techniques include reducing imperviousness, conserving natural resources and ecosystems, maintaining natural drainage courses, reducing the use of pipes, and minimizing clearing and grading.
- Providing runoff storage measures dispersed uniformly throughout a site's landscape with the use of a variety of detention, retention, and runoff practices.
- Maintaining pre-development time of concentration by strategically routing flows to maintain travel time and control the discharge.

University of Maryland rain garden monitoring site

- Implementing effective public education programs to encourage property owners to use pollution prevention measures and maintain the on-lot hydrologically functional landscape management practices.

LID does not rely on the conventional end-of-pipe or in-the-pipe structural methods but instead uniformly or strategically integrates stormwater controls throughout the urban landscape.

Conservation Design

Conservation design is a design approach to site development that protects and incorporates natural site features into the stormwater management program. There is a subtle difference between conservation design and LID in that a primary emphasis of conservation design is to incorporate natural site features into the site development process and thereby reduce or eliminate the need for structural stormwater management. As a central tenet, maintenance of natural site features plays a greater role in conservation design than it does in LID. Clustering development on a smaller portion of a site provides greater retention of natural site features that assist in stormwater management. This is a small difference, though, that can be incorporated into LID, but it is more clearly stated in the conservation design approach.

The site features to be protected and incorporated are similar to those discussed earlier in the chapter and include the following items:

- Wetlands;
- Floodplains;
- Forested areas;
- Meadows;
- Riparian buffers;
- Soils; and
- Other natural features.

In short, the point of conservation design is to do more with less. Design principles (which are identical to the ones used in LID) include:

- Achieving multiple objectives;
- Integrating stormwater management and design early into the site planning and design process;
- Prevention rather than mitigation;
- Managing stormwater as close to the point of origin as possible, minimizing collection and conveyance; and
- Relying to the maximum on natural processes within the soil mantle and the plant community.

Water Sensitive Urban Design (WSUD)

WSUD is similar to LID and conservation design and has been developed in Australia to address issues there. WSUD is a philosophical approach to urban planning and design that aims to minimize the hydrological impacts of urban development on the surrounding environment. Stormwater management is a subset of WSUD directed at providing flood control, flow management, water quality improvements, and opportunities to harvest stormwater to supplement potable water

Bioretention in Sydney

Rain garden in Auckland

for non-potable uses (that is, toilet flushing, garden irrigation, etc.).

Key planning and design objectives of WSUD are:

- Protecting and enhancing natural water systems in urban developments;
- Integrating stormwater treatment into the landscape by incorporating multiple-use corridors that maximize the visual and recreational amenity of developments;
- Protecting water quality draining from urban development;
- Reducing runoff and peak flows from urban developments by employing local detention measures and minimizing impervious areas;
- Adding value while minimizing drainage infrastructure development costs.

WSUD recognizes that opportunities for urban design, landscape architecture, and stormwater management infrastructure are intrinsically linked. The practices that promote long-term success of a stormwater management scheme are called Best Planning Practices (BPPs) and Best Management Practices (BMPs). They can apply to greenfield land development sites, redevelopment sites in built-up areas, and, in some instances, to retrofits in fully urbanized watersheds. The scale of application can range from individual houses, streetscapes and precincts, to whole watersheds.

Sustainable Urban Drainage Systems (SUDS)

SUDS is a design approach for urban drainage in England and Wales that includes long-term environmental and social factors in decisions about drainage. It takes account of the quantity and quality of runoff and of the amenity value of surface water in the urban environment. Many existing urban drainage systems can cause problems of flooding, pollution or damage to the environment and are not proving to be sustainable.

Drainage systems can be developed in line with the ideals of sustainable development by balancing the different issues that should be influencing the design. Surface water drainage methods that take account of quantity, quality, and amenity issues are collectively referred to as SUDS. These systems are more sustainable than conventional drainage methods because they:

- Manage runoff flowrates, reducing the impact of urbanization on flooding;
- Protect or enhance water quality;
- Are sympathetic to the environmental setting and the needs of the local community;
- Provide a habitat for wildlife in urban watercourses; and
- Encourage natural groundwater recharge (where appropriate).

They do this by:

- Dealing with runoff close to where the rain falls;
- Managing potential pollution at its source now and in the future; and
- Protecting water resources from point pollution (such as accidental spills) and diffuse sources.

Low Impact Urban Design and Development (LIUDD)

Low impact urban design and development is a design approach used in Auckland, New Zealand whose concepts and approach are similar to all of the above-mentioned ones. LIUDD presents an alternative approach to site and watershed development from a stormwater management perspective. Its basis lies in the recognition that the volume of stormwater discharged from a site may be of equal importance to limiting pollution discharge. The low impact urban design and development approach is another stormwater management tool for reducing the adverse impacts of stormwater runoff. There are two primary areas of interest addressed in this design approach:

- Erosion and sediment control during construction, and
- Permanent stormwater management.

The Auckland approach recognizes that much of the technical information for LIUDD has been developed from a design perspective but is doing considerable work to address the institutional barriers to successful implementation of LIUDD. Local land use plans tend to have enough flexibility to allow for case-by-case implementation of the approach; the major problem regards codes of practice or engineering standards. These can include minimum street width, requirements for curbing, side walks, and other criteria that prevent LIUDD from being implemented. Significant efforts have gone into removing these barriers where they exist.

In addition, there are several case studies of LIUDD approaches on a watershed basis. Monitoring of watercourses for quantity, quality, and biology has been done prior to development initiation. It will take approximately 10 years for ultimate development to occur in these watersheds, and monitoring will be done both during and post-construction to evaluate the benefits of LIUDD on a watershed basis.

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CHAPTER 9

Erosion and Sediment Control

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Erosion and sediment management practices have not evolved to the extent that stormwater management practices have over the past 10 years. There are, however, other components that have more recently begun to emerge. These components include considering temporary and permanent revegetation, phasing work to limit open areas, using chemical flocculation of sediment traps to provide enhanced sediment discharge reduction (especially of clay soils), and also looking at innovative ways of offsetting the residual impacts that result from sediment yields. If improved treatment is to be provided, however, more attention needs to be given to advancing erosion and sediment control practices. It is also important to note that erosion and sediment control is increasingly being looked at as an essential component of the overall site development package. The linkages between this early stage of site development and the longer-term approach to stormwater management can not be separated if we are to protect our downstream aquatic environments.

For erosion and sediment control programs, technology must continue to improve and approaches must be further refined if aquatic resource protection is to be realized. An effective stormwater management program is not going to achieve its goals if the receiving systems are severely impacted during the construction phase of a project. In addition to significant sediment loads, the construction phase of site development can increase the total volume and peak rates of stormwater exiting a site and cause downstream channel instability concerns. It is a positive step, therefore, that the Phase II program is also emphasizing erosion and sediment control on smaller sites as an essential permit component.

While this chapter can be read alone, it should be considered in conjunction with the rest of the manual to achieve a full appreciation of the development cycle and all the aspects that contribute to it.

Regulatory Nature of Erosion and Sediment Control

Phase II of the National Pollutant Discharge Elimination System (NPDES) specifies that a program to reduce pollutants has to be developed, implemented, and enforced in any stormwater runoff from construction activities that result in a land disturbance of greater than or equal to one acre. Reduction of stormwater discharges from construction activity disturbing less than one acre must be included in your program if that construction is part of a larger common plan of development or sale that would disturb one acre or more.

The program must include the development and implementation of, at a minimum:

- A regulatory mechanism to require erosion and sediment controls, as well as sanctions to ensure compliance, to the extent allowable under state, tribal or local law;
- Requirements for construction site operators to implement appropriate erosion and sediment control best management practices;
- Requirements for construction site operators to control waste such as discarded building materials, concrete truck washout, chemicals, litter, and sanitary waste at the construction site that may have adverse impacts on water quality;

- Procedures for site plan review that incorporate consideration of potential water quality impacts;
- Procedures for receipt and consideration of information submitted by the public; and
- Procedures for site inspection and enforcement of control measures.

It is important to note that examples of sanctions to ensure compliance include non-monetary penalties, fines, bonding requirements, and/or permit denials for non-compliance. The EPA recommends that procedures for site plan review include the review of individual pre-construction site plans to ensure consistency with local erosion and sediment control requirements.

Procedures for site inspections and enforcement of control measures could include steps to identify priority sites for inspection and enforcement based on the nature of the construction activity, topography, and the characteristics of soils and receiving water quality. Additional educational and training measures for construction site operators should also be considered. Phase II of the NPDES program provides for regulatory flexibility that would require pollution prevention plans for non-sediment generated pollutants.

Phase II of the NPDES municipal requirements provides an excellent platform from which erosion and sediment control can be more effectively managed.

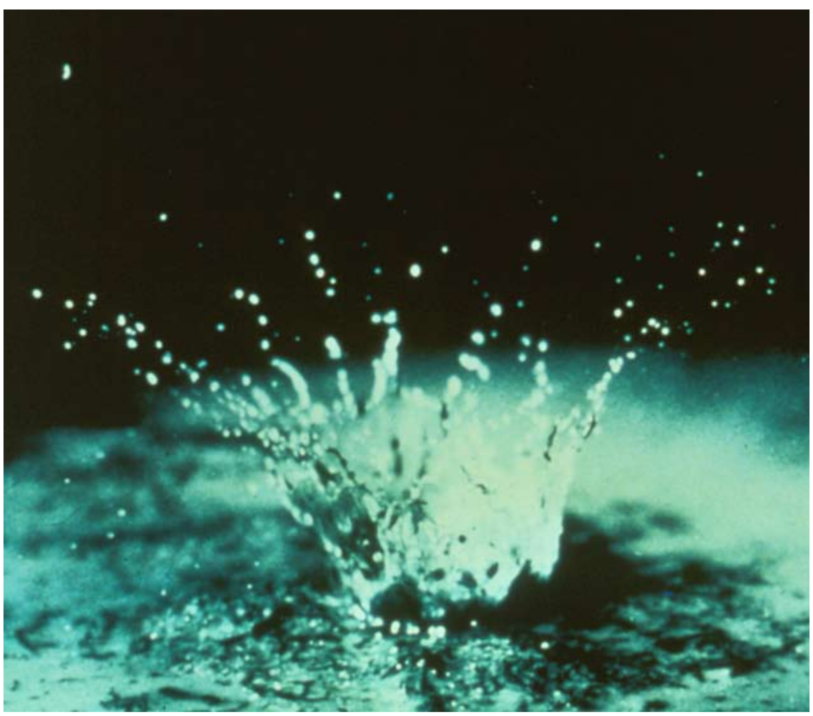
Principles of Erosion and Sediment Control

Erosion and sediment management must always be thought of as two separate components, erosion control and sediment control.

Erosion is the process whereby the land surface is worn away by the action of water, wind, ice, or other geological processes. The resultant displaced material is known as sediment, with sedimentation being the deposition of this eroded material. Accelerated erosion is primarily caused by human activities and is a much more rapid process than natural erosion.

The basic erosion process consists of detachment, transport, and sedimentation, with water often being the key eroding agent and transport medium. When considering erosion, the following seven main types need to be looked at:

- Splash erosion is commonly caused by raindrop impact. This impact can break up the soil surface with a net effect of moving soil particles down the slope.
- Sheet erosion occurs when intensity of rainfall exceeds the infiltration rate. Sheet erosion refers to the uniform removal of soil in thin layers by the forces of raindrops and overland flow.
- Rill erosion is the removal of soil by runoff moving in concentrated flows. The velocity and the turbulence of the flow increase in these concentrated flow paths, with the resultant energy detaching and transporting soil particles.
- Gully erosion is the next step from rill erosion, where gullies form that are usually distinguished by being greater than 300 mm in depth. The potential for gullies to transport significant amounts of sediment is large, and from an erosion control standpoint, they should be avoided.
- Tunnel erosion is the removal of subsurface soil by subsurface water, while the surface soil



Raindrop impact

remains intact. This produces large cavities beneath the ground surface that can eventually lead to collapse of the surface material.

- Channel erosion occurs once the water in concentrated flow reaches the stream system. This erosion is essentially caused when the water velocity increases such that scouring or undercutting of the stream banks occurs. Channel erosion is noted to have a direct relationship with watershed urbanization, with increased flows and increased erosion occurring once a watershed is urbanized.
- Mass movement is the erosion of soil or rock by gravity-induced collapse. It can be triggered by heavy rainfall and increased groundwater pressure, or by streams undercutting the base of a slope where works are occurring.

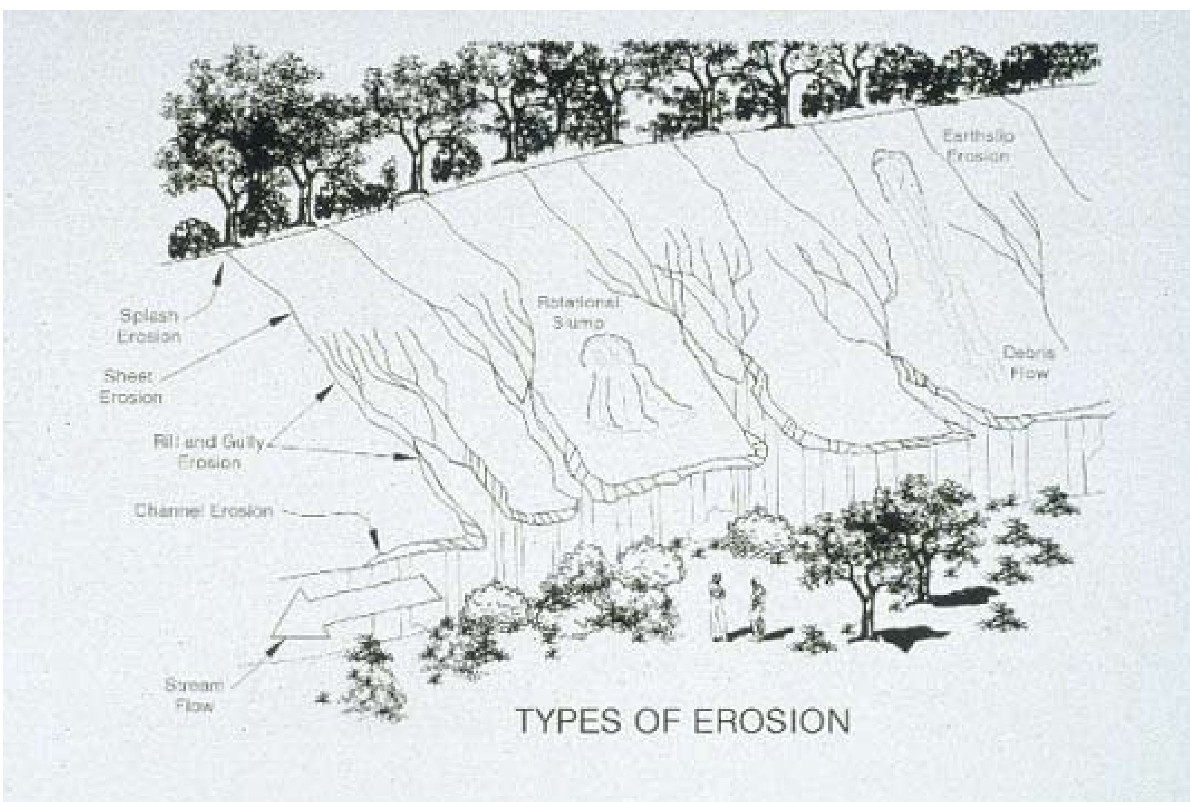
It is also important to understand the factors that influence the erosion process. These four factors (discussed below) are climate, soil characteristics, topography, and ground cover. They also form the basis of the Universal Soil Loss Equation, which will be discussed in detail later in the chapter.

Climate

Climate is a key factor, with rain being the driving force of erosion. The erosive power of rain is determined by rainfall intensity and the droplet size. The annual pattern of rainfall and temperature change is also critical in that it determines the extent and growth rate of vegetative cover, the key tool in prevention of erosion.

Soil characteristics

All soil characteristics, including texture, organic matter content, structure, and permeability are important. Sand, silts, and clays are the major soil particle classes, and it is critical to understand the soils you are working with to be able to assess the erodibilty of these different particles. Organic matter is critical in improving soil structure and increasing the permeability and water-holding capacity of the soil. Soil permeability in itself is important, with soils with higher permeability producing less runoff than soils with a low permeability. Soil structure is also important in that compacted soil will result in runoff as opposed to infiltration.



In this context it should be noted that soil compaction is also a major issue that needs to be considered. Soil compaction during construction and use is likely to cause most of the reduced infiltration capacity of urban soils. This aspect of construction activities is often not given due attention, in spite of the fact that the reduced infiltration is known to result in increased runoff and associated effects (Pitt et al., 2002).

Topography

Topography is important primarily from a slope length and angle perspective. The shape of the slope also plays an important part, with the base typically being more susceptible to erosion than the top due to runoff arriving at the base at a faster, more concentrated rate. Reference should be made to section 8.9 of this chapter for an illustration of slope versus sediment yields.

Ground Cover

Ground Cover includes vegetation and surface treatment such as mulches and geotextiles. This aspect is the most important and effective form of long-term erosion control. Good ground cover provides direct instant protection, slows runoff, and maintains the soil's ability to absorb water.

Evapotranspiration

Evapotranspiration is a further factor often not considered. In some areas, however, it is critical in that minimal rainfall and high evapotranspiration in the summer period can lead to soil moisture deficit. This becomes a critical factor when the question of establishing vegetative cover for erosion control arises, because it can lead to the necessity of considering alternative methodologies to establish a vegetative cover.

Once the principles of erosion and sedimentation are understood, it is much easier to also understand the importance of erosion control and sediment control. Erosion control is based on prevention of erosion in the first instance and includes controls such as revegetation, contour slope drains, project phasing, and time frame limitations. The specific designs of some erosion control mechanisms are discussed later in this chapter.

Sediment Control is based on prevention of sedimentation and of sediment leaving the site in question. Sediment control is never 100 percent effective, but with effective erosion control, it can go a long way toward minimizing downstream effects of sediment discharge.

When assessing construction operations, the emphasis must always be on the prevention of erosion in the first instance. Only after this has been fully assessed should the operation consider the sediment control options for the site. These options may have numerous components, but they will always include perimeter controls. It is important, however, to recognize that sediment control can also include controls such as sediment traps and ponds within a site that may reduce reliance on the perimeter controls installed.

Assessing Sediment Generation of Construction Sites

The most important physical property of a soil particle is its size. The size of the particle can be determined in a number of ways. The nominal diameter refers to the diameter of a sphere of the same volume as the particle, and the sieve diameter is the minimum length of the square sieve opening through which a particle will fall. Recognizing the size of material on an earthworks site can increase the awareness of how easy or difficult it can be to remove sediment once it is in suspension. This helps target the erosion and sediment controls. Clay is considered to be less than 0.002 mm in diameter, silt between 0.002 and 0.063 mm, and sands greater than 0.063 mm. While sands and silts are more erodible than clays, they settle easier, whereas clays, a cohesive material that can form quite strong bonds once in suspension, are very difficult to trap with sediment control mechanisms. This places the emphasis on sites with clay soil dominance on erosion control methodologies.

The Universal Soil Loss Equation (USLE) is a simple model that was originally developed for agricultural practices and is now recognized as a suitable sediment yield estimation tool for activities such as earthworking operations. Rather than providing an accurate estimate of actual total sediment yield, the most beneficial use of the USLE has proven to be the identification of variations of sediment yields across a particular site. To achieve this, it is critical that a site is divided up into logical sectors, based on gradient, slope length, and surface cover. Other factors to consider are the proximity and nature of the receiving environment. Once completed, the USLE will allow the erosion and sediment control methodology to be tailored to suit the variations across the site.

The USLE is based on the following factors:

Rainfall Erosion Index (R)

This factor is a measure of the erosive force and intensity of the rain in a normal year. It is based on the energy and the maximum 30-minute intensity for all major storms in an area during an average year. It is derived from probability statistics resulting from analyzing rainfall records of individual storms.

Soil Erodibility Factor (K)

This represents the ability of the surface to resist the erosive energy of rain. Texture is the principle factor affecting K, but structure, organic matter, and permeability also contribute. Adjustments are made to the K factor as the site works progress, reflecting the percentage of clay, silts and sands within a soil structure. In calculating the K factor, an allowance is also made for the percentage of organic matter that is contained within the soil.

Length-Slope Factor (LS)

This is a numeric representation of the length and slope angle of a site. It is the ratio of soil loss per unit area on a site to the corresponding loss from a 22.1-meter-long experimental site with a 9 percent slope. Representative slope length and gradients are assessed for the separate sediment sources and depicted in a table. It should be noted that the potential sediment generation on a site increases geometrically with an increase in gradient. It is therefore essential that bare area and slope length are minimized on steeper gradients. This may be achieved by staging works, progressively stabilizing completed areas and installing contour drains to reduce slope lengths.

Ground Cover Factor (C)

This is the ratio of soil loss under specified conditions to that of a bare site. Where the soil is protected against erosion, the C Factor will reduce the soil loss estimate. This factor also takes into account the effectiveness of the vegetation and mulch in preventing the detachment and transport of soil particles.

Erosion Control Factor (P)

This factor reflects the roughness or smoothness of the earthworks surface with the rougher surface having a lesser value. As examples, bare soil that is compacted and smooth would have a P factor of 1.3 while a rough irregular surface such as contour plowing would have a value of 0.8. The lower value results in reduced erosion.

Once the values for R, K, LS, C, and P have been derived, the value for estimated sediment generated can be calculated. To estimate the quantity of sediment likely to be discharged to the receiving environment, it is necessary to multiply this result by the areas of exposure, the sediment delivery ratio, the sediment control measure efficiency, and the duration of exposure. Areas of the site, or the entire site, that are demonstrated to exhibit high sediment yields can then be managed accordingly.

It is important to also be aware that the sediment that is generated will be mobilized as either bedload or as suspended sediment. Bedload is moved at or near the bottom of the stream, while suspended sediment is mixed with the waters of the receiving environment.

Impacts of Sediment Discharge on Receiving Environments

Irrespective of the erosion and sediment controls employed, construction activities lead to sediment generation and to a subsequent sediment discharge with, among others, visual, recreational, and ecological impacts. These activities can be appropriately managed by the respective authorities through a range of tools inclusive of both regulatory and educational initiatives. One of the key tools in the educational component is a specific guideline for erosion and sediment control.

The range of guidelines available typically provide a comprehensive guide for erosion and sediment control, detail the specific policies and rules applying to the site in question, and essentially work toward minimization of adverse environmental effects of sediment discharge through appropriate use and design of specific measures.

The guidelines should detail both principles and practices emphasizing the importance of both non-structural and structural measures to be implemented on sites.

In terms of the regulatory component, permits are key tools that are utilized to minimize impacts from sediment discharge. It is also recognized that compliance inspections of these permits are an important aspect of ensuring that environmental objectives are achieved. Associated with this key tool is an enforcement role which, when combined with all other relevant program aspects, will provide a suitable implementation mix.

There are many effects associated with sediment discharge. Runoff from construction sites is by far the largest source of sediment in urban areas under development. Soil erosion removes over 90 percent of sediment by tonnage in urbanizing areas where construction activities occur. The following values illustrate some of the measured sediment loads associated with construction activities found across the United States.

- York County Soil and Water Conservation District 1990 – Sediment loading rates vary from 36.5 to 1000 tons/acre/year, which is 5 to 500 times greater than those from undeveloped land.
- Franklin County, Florida – Sediment Yields
    - Forest: less than 0.5 ton/acre/year
    - Rangeland: less than 0.5 ton/acre/year
    - Construction Site: 30 tons/acre/year
    - Established Urban: less than 0.5 ton/acre/year
- Washington Department of Ecology, 1989 – Erosion rates range from 50 to 500 tons/acre/year for construction activities. Natural erosion rates from forest are 0.01 to 1.0 ton/acre/year.
- Wisconsin Legislative Council, 1991 – Erosion rates range from 30 to 200 tons/acre/year, which is noted to be 10 to 20 times those of croplands.

As summarized above, the huge potential for sediment to be generated from land bared through construction activities is significantly greater than for many other land uses.

A study undertaken in Auckland, New Zealand measured sediment yields from various land uses over time and predicted the average annual soil loss for these land uses. Construction sites were shown to have a predicted average annual soil loss of up to 400 times that of a pastoral site. The study also demonstrated that sediment yields would increase markedly with larger storm events. The significantly higher levels of sediment that occur as a result of land disturbance activities need to be identified within the specific program and to have the appropriate policy backing to ensure that these issues can be addressed.

The following effects can result from sediment discharges.

Biological Effects

Large amounts of sediment in a waterway are harmful to fish and other aquatic life. Aquatic life can be physically smothered by a build-up of sediment in the stream bed. Aquatic life not actually covered by deposits of silt can sustain damage to their gill and mouthparts due to the abrasive nature of the silt. The juvenile stages of many species are particularly vulnerable. Sedimentation may also significantly alter habitats, for example by destroying spawning grounds.

Algae, the major food supply for stream life, can be scoured off the rocks in the stream bed by sediment. Other links in the food chain may also be affected and the surviving animals forced to migrate elsewhere if they can.

Turbidity (cloudiness of the water) from suspended solids in the water may stop animals feeding because they cannot see their prey. It can also affect aquatic life by increasing heat absorption and therefore the temperature of the water. It also stops light penetrating the water, slowing down photosynthetic activity and subsequent plant and algae growth.

Other Pollutants

Sediment transports other pollutants such as lead, hydrocarbons, agricultural nutrients, and toxic substances into streams and harbors. There they can accumulate and affect aquatic life. Control of the pollutants transported by sediment is simply achieved by controlling the generation and movement of sediment itself.

Stream Blockage

Sediment deposition can lead to the infilling of affected water bodies. This in turn can lead to a reduction in their hydraulic efficiency, an increase in susceptibility to flooding, and restrictions to access. While such sediment deposition has environmental impacts, the removal works also have potential for serious environmental effects.

Effects on Consumable Water Resources

High loadings of suspended solids affect the use of water for irrigation, stock, and domestic water supplies. Sediment in irrigation water clogs pump filters and sprinkler nozzles. In domestic and stock water supplies, it can lead to unacceptable drinking quality. Removing sediment from drinking water can be an expensive operation. Furthermore, sediment can form a threat to the useful life of dams.

Aesthetic Values

Sediment discharges into streams, lakes, or coastal waters detract from their aesthetic qualities. Clean, clear water is perceived as being much more conducive to recreation than "dirty," sediment-laden water. The purely scenic value of water bodies such as key harbor areas is enhanced by their degree of clarity.

Damage to Property and Public Utilities

Construction activities can inundate lower-lying properties or roadways with sediment if adequate control measures are not in place.

Effect of Sediment on Matters of Cultural Significance

Construction activities often disturb items and matters of cultural and archaeological importance. The effects can vary from a direct effect on these matters, such as significant destruction or alteration of a physical site, through more indirect effects such as impacts on cultural values.

Erosion Control Measures

Erosion control mechanisms typically include the following measures.

Earth Dike

A temporary berm or ridge of compacted soil, located in such a manner as to channel water to a desired location.

Its purpose is to direct runoff to a sediment trapping device, thereby reducing the potential for erosion and off-site sedimentation. Earth dikes can also be used for diverting clean water away from disturbed areas.

Earth dikes are often constructed across disturbed areas and around construction sites such as graded parking lots and subdivisions. The dikes shall remain in place until the disturbed areas are permanently stabilized.

Runoff Diversion Channel/Berm

A non-erodible channel or berm for the conveyance of runoff constructed to a site-specific cross section and grade design. To either protect work areas from upslope runoff (clean water diversion), or to divert sediment-laden water to an appropriate sediment retention structure.

Contour Drain

A temporary ridge or excavated channel, or a combination of ridge and channel, constructed to convey water across sloping land on a minimal gradient. To

Example of an earth dike flow diversion

periodically break overland flow across disturbed areas in order to limit slope length and thus the erosive power of runoff, and to divert sediment-laden water to appropriate controls or stable outlets.

Benched Slope

Modification of a slope by reverse sloping to divert runoff to an appropriate conveyance system. The purpose is to limit the velocity and volume, and hence the erosive power of water moving down a slope and therefore minimize erosion of the slope face.

Contour drain

Slope benches

Rock Check Dam

Small temporary dam constructed across a channel (excluding perennial watercourses), usually in series, to reduce flow velocity. May also help retain sediment. The primary purpose of a rock check dam is to reduce the velocity of concentrated flows, thereby reducing erosion of the channel. While trapping some sediment, they are not specifically designed to be utilized as a sediment retention measure.

Top Soiling

The placement of topsoil over a prepared subsoil prior to the establishment of vegetation. This serves to provide a suitable soil medium for vegetative growth for erosion control while providing some limited short-term erosion control capability.

Temporary and Permanent Seeding

The planting and establishment of quick-growing and/or perennial vegetation to provide temporary and/or permanent stabilization on exposed areas. Temporary seeding is designed to stabilize the soil and to protect disturbed areas until permanent vegetation or other erosion control measures can be established. It may be used where the area to be stabilized needs temporary stabilization but is not yet up to final grade and requires further earthworks.

Hydroseeding

The application of seed, fertilizer, and a paper or wood pulp with water in the form of a slurry which is sprayed over the area to be revegetated. To establish vegetation quickly while providing a degree of instant protection from rain drop impact.

Examples of rock check dams

This form of revegetation facilitates the establishment of vegetation on steep slopes and also allows a mixture of appropriate seeds to be utilized dependent upon the site conditions. It is, however, dependent upon the availability of appropriate hydroseeding contractors with suitable machinery.

Mulching

The application of a protective layer of straw or other suitable material to the soil surface to protect it from the erosive forces of raindrop impact and overland flow.

Straw mulch being applied

Turf being applied to topsoil

Mulching also helps to conserve moisture, reduce runoff and erosion, control weeds, prevent soil crusting, and promote the establishment of desirable vegetation.

Turfing

The establishment and permanent stabilization of disturbed areas by laying a continuous cover of grass turf. Provides immediate vegetative cover in order to stabilize soil on disturbed areas.

Geosynthetic Erosion Control Systems

The artificial protection of channels and erodible slopes utilizing artificial erosion control material such as geosynthetic matting, geotextiles or erosion matting. Immediately reduces the erosion potential of disturbed areas and/or reduces or eliminates erosion on critical sites during the period necessary to establish protective vegetation. Some forms of artificial protection may also help to establish protective vegetation.

Stabilized Construction Entrance

A stabilized pad of aggregate on a filter cloth base located at any point where traffic will be entering or leaving a construction site. To assist in minimizing dust generation and disturbance of areas adjacent to the road frontage by giving a defined entry/exit point.

Pipe Drop Structure/Flume

A temporary pipe structure or constructed flume running from the top to the bottom of a slope. A pipe drop structure or a flume structure is installed to convey surface runoff down the face of unstabilized slopes in order to minimize erosion on the slope face.

Stabilized construction entrance

Level Spreader

A non-erosive outlet for concentrated runoff constructed so as to disperse flow uniformly across a slope. To convert concentrated flow to sheet flow and release it uniformly over a stabilized area to prevent erosion.

Surface Roughening

Roughening a bare earth surface with horizontal grooves running across the slope or tracking with construction equipment. To aid in the establishment of vegetative cover from seed, to reduce runoff velocity and increase infiltration, and to reduce erosion and assist in sediment trapping.

Rock Outlet Protection

A section of rock protection placed at the outlet end of culverts, conduits, and channels.

Flow spreading

Surface roughening

Its purpose is to reduce the velocity and energy of water such that the flow will not erode the receiving downstream reach.

This practice applies where discharge velocities and energies at the outlets of culverts, conduits, and channels are sufficient to erode the next downstream reach.

Sediment Control Measures

Sediment control mechanisms typically include the following measures.

Temporary Swale

A temporary excavated drainageway.

Its purpose is to prevent runoff from entering disturbed areas by intercepting and diverting it to a stabilized outlet or to intercept sediment-laden water and divert it to a sediment trapping device.

Conditions where this practice applies:

- To divert flows away from a disturbed area and to a stabilized area;
- To shorten overland flow distances intermediately across disturbed areas;
- To direct sediment-laden water along the base of slopes to a trapping device; or
- To transport off-site flows across disturbed areas such as rights-of-ways.

Examples of pipe and flume downdrains

Sediment Retention Trap and Pond

A temporary device formed by excavation and/or embankment construction in order to intercept sediment-laden runoff and provide an impoundment for suspended sediment to settle out. To treat sediment-laden runoff and reduce the volume of sediment leaving a site in order to protect downstream environments from excessive sedimentation and water quality degradation.

Sediment ponds and traps

Sediment ponds and traps

Silt Fence

A temporary barrier of woven geotextile fabric used to intercept runoff, reduce its velocity, and impound sediment-laden runoff from small areas of disturbed soil. To detain flows from runoff so that deposition of transported sediment can occur through settlement. Silt fences can only be used to intercept sheet flow. They cannot be used as velocity checks in channels or placed where they will intercept concentrated flow.

Super Silt Fence

A temporary barrier of geotextile fabric over chain link fence used to intercept flows, reduce their velocity, and impound sediment-laden runoff from small catchment areas. To reduce runoff velocity and allow the deposition of transported sediment to occur.

Silt fence

Super silt fence

Straw Bale Barrier

Temporary barrier of hay bales used to intercept and direct sediment-laden surface runoff from small areas to a sediment retention facility so that deposition of transported sediment can occur.

Stormwater Inlet Protection

A barrier across or around a cesspit (stormwater inlet) that is designed to intercept and filter sediment-laden runoff before it enters a reticulated stormwater system via a cesspit.

Earth Dike

A temporary berm or ridge of compacted soil (inclusive of topsoil) to create impoundment areas where ponding of runoff can occur and suspended material can settle before runoff is discharged.

Sump/Sediment Pit

A temporary pit constructed to trap and filter water for pumping to a suitable discharge area. The design is based on a perforated vertical standpipe placed in the center of a pit that is then backfilled with aggregate.

Perimeter Dike/Swale

A temporary ridge of soil excavated from an adjoining swale located along the perimeter of the site or disturbed area.

Its purpose is to prevent off-site storm runoff from entering a disturbed area as well as prevent sediment-laden storm runoff from leaving the construction site or disturbed area.

A perimeter dike/swale is constructed to divert flows from entering a disturbed area, along tops of slopes to prevent flows from eroding a slope, or along the base of slopes to direct sediment-laden flows to a trapping device. The perimeter dike/swale shall remain in place until the disturbed areas are permanently stabilized.

Non-Sediment-Related Pollutants from Construction Sites

Concrete washings, water blasting, equipment washing, concrete and tile cutting are all works occurring at construction sites that can pollute waterways unless care is taken. These pollutants are known to cause problems due to their highly alkaline nature. They contain oxides, heavy metals, or, possibly, petroleum products.

There are, however, practices that can be followed to ensure that the effects from these activities do not create adverse effects. These can include the following:

- When washing operations generate fine sediments (silts or clays), make sure the wash water is confined, filtered or diverted across to a soak area. If discharge is necessary, it should go into a sanitary sewer, not the stormwater system.
- Do not wash equipment on site unless there is a designated washout area where wash water soaks into the ground or is treated.
- When waterblasting, contain dirty waste runoff. Chemical additives should not be discharged to the stormwater system. Utilize filter cloth to filter out paint flakes and sediment prior to discharge.
- Slurry from directional drilling must be allowed to settle, with the water soaking to ground or taken off-site to an appropriate disposal location.

Inlet protection

Sediment dewatering pit

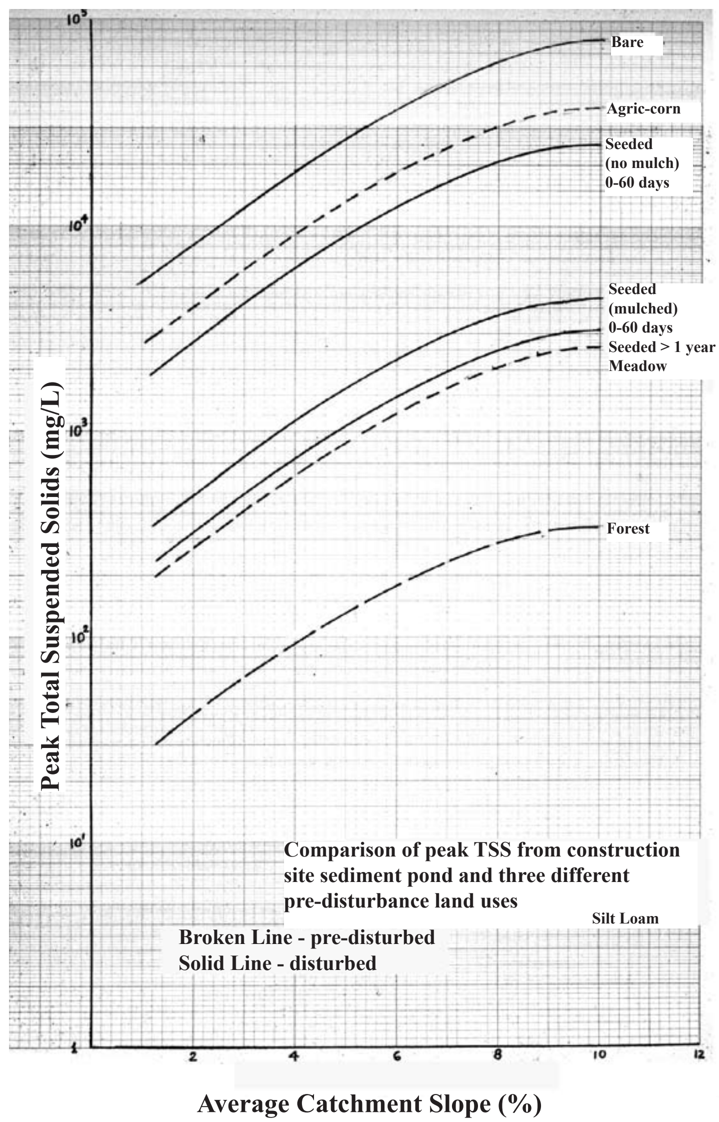
Innovative Practices

While the previously detailed erosion and sediment controls are typical traditional controls and will be reasonably effective for most sites, there are many circumstances where more specific innovative practices should be employed. Erosion and sediment controls need to be significantly advanced in this area to ensure protection of the receiving environment.

Innovative practices should be considered on all sites, particularly those that show high sediment yields, and include the following structural and nonstructural measures.

A study done in the U.S. in the mid-1980s (Maryland Department of Natural Resources) looked at a number of parameters that could affect the discharge of sediments from a construction site. One component of the study was to look at the benefits of temporary stabilization techniques as an erosion control tool. The results of this project demonstrated the various stabilization techniques and the impact of slope on sediment loadings. They are illustrated in Figure 9-1. It is important to note that there is a clear trend not only for sediment loading to increase with slope angle, but also for it to decrease as vegetative cover increases.

Figure 9-1: Relationship Between Vegetative Cover and Slope Versus Sediment Yield



Mulching

It is acknowledged that when land is disturbed for construction and other activities, the rate of erosion increases as unvegetated surfaces are subjected to raindrop impact and overland flow. While the benefits of stabilization of soils to minimize erosion have been supported by research for some time, it is important to recognize that the application of mulch has been somewhat inconsistent. It is critical that mulching becomes an integral part of any earthworks site and forms a part of all erosion and sediment control plans.

Investigation results demonstrate that:

- Established grass cover and mulching topsoil surfaces are the most effective way of reducing sediment discharge.
- Clay- and silt-size particles typically form the greatest proportion of sediments discharged from construction sites that have effective sediment controls, and the discharge of these particles is minimized through mulching the surface in question.
- Mulched topsoil areas produce up to 95 percent less sediment discharge than bare subsoil surfaces.

Chemical Treatment of Runoff

One method of enhancing the retention of suspended sediment in earthworks runoff is the use of flocculant. In recent advancements, liquid flocculant can be added directly to sediment retention pond inflows via a rainfall-activated system. The flocculant causes individual particles to be destabilized (neutralizing electrical charges that cause particles to repel each other), accelerating the coagulation and settlement of particles that may otherwise be discharged from the pond.

The key purpose of using flocculation is to treat sediment-laden runoff to an extent greater than standard sediment control practices and to reduce the volume of sediment leaving a site.

Flocculation may be used to enhance the retention of sediment on earthworks sites where there are concerns about the scale of works, potential effects on sensitive receiving environments, or cumulative discharges, or where it may not be feasible to construct standard sediment control practices.

Flocculation using the system illustrated below is simply incorporated into the design of a sediment retention pond. The catchment draining into the pond needs to be considered carefully throughout the period of flocculation, as the components making up the flocculation system are sized on the catchment characteristics, including area and soil type. The rainfall-activated flocculation system outlined in this section is based on the use of polyaluminum chloride (PAC). Other aluminum coagulants, including alum (aluminum sulphate), may be suitable for use; however, methodologies may need to be adapted to produce appropriate outcomes.

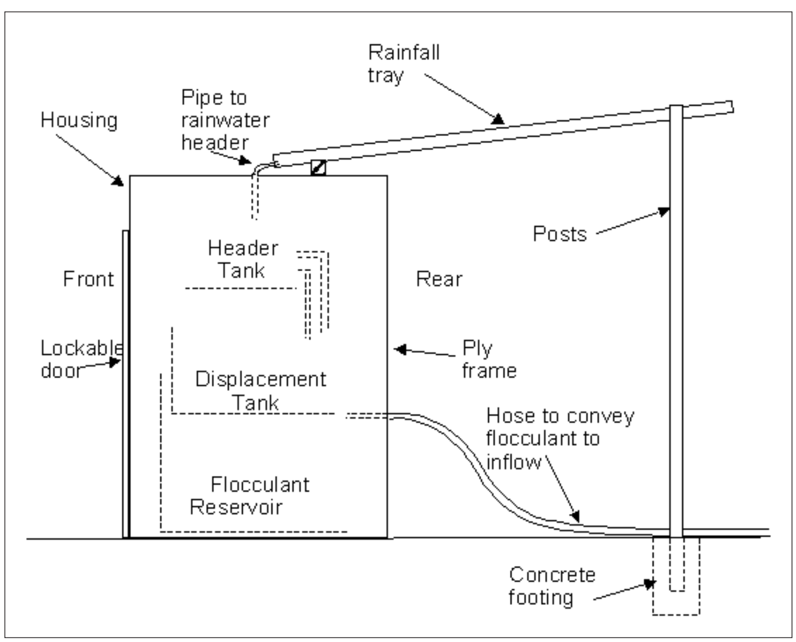
The general components of the flocculation system include a rainfall catchment tray, header tank, displacement tank, and flocculant reservoir tank, as detailed below.

Rainfall from the watershed-sized rainfall tray drains to a header tank. The header tank provides storage capacity to avoid dosing during initial rainfall following a dry period and to attenuate dosing at the beginning and end of a rainstorm (to simulate the runoff hydrograph). The header tank provides:

- Zero flocculant discharge until a pre-selected quantity of rain has fallen, to allow for initial infiltration and saturation of dry ground before runoff commences;
- A slow start to the dosing rate to allow for the response time of runoff flowing off the site at the beginning of a storm; and
- An extension of the dosing period beyond the rainfall period to provide treatment of runoff that occurs following cessation of rainfall.

From the header tank, the rainwater discharges by gravity into a displacement tank which floats in the flocculant reservoir. As the displacement tank fills with rainwater, flocculant is displaced through the outlet in the reservoir tank and then flows by gravity to the

Example of an on-site chemical flocculation system



Schematic of a chemical flocculation system dosing point. The dosing point should be selected in an area of high turbulence in the pond inflow channel.

The zero (flocculant) discharge rainfall volume can be adjusted manually for site characteristics by adding or removing water from the header tank.

It is important that the pH of the soils in the area in question is understood. The pH of soils should be tested prior to and during construction, as the exposure of different soil horizons may alter the runoff pH. Dosing with aluminum-based flocculant should cease where the pH drops below 5.5, as the toxicity of the aluminum fraction of the flocculant increases below this level, potentially placing at risk receiving environment organisms.

The use of flocculation will lead to more frequent sediment retention pond maintenance. The sediment containing the flocculant is not considered to be toxic, as the aluminum is bound up with the soil particles. It is common practice for the accumulated sediment to be dried on site and incorporated into fills.

Flocculants can provide an alternative to traditional sediment control practices. Dependent upon the nature of the site, they can ensure that adequate controls are implemented when site conditions restrict options.

Performance of flocculants for sediment removal has proved the approach to be very effective at removal of pond sediments, especially for finer silt and clay particles. Monitoring of 21 sediment ponds demonstrated that efficiencies ranged between 90 and 99 percent removal of suspended sediments. In circumstances where flows exceeded pond design criteria, efficiencies were also notably enhanced.

The key features of this approach are the following:

- They are simple to install and maintain.
- They do not require electrical power.
- They are rainfall-activated.
- They dose the critical storm.
- They are easily transportable and reusable.
- They are cost-effective (around $1,500 plus maintenance and operational costs).
- They require no dedicated staff.

Low Impact Design (LID)

The principles behind LID are based upon using an analysis of existing site conditions as a baseline from which to commence site planning. The site conditions provide an inventory of the full range of natural systems such as soils, geology, vegetation, habitat, and the cultural and archaeological factors associated with the site. The more the complex and integrated nature of the conditions is understood, the better the earthworks and building program can be fitted on the site with reduced impact. LID is similar to erosion control in that it is a preventative approach reducing the amount of sediment generated by practicing the principles through planning processes.

Only after a full site analysis and inventory has been undertaken can the erosion and sediment control plan be developed and full control mechanisms considered.

While considered innovative, LID should be built into all erosion and sediment control methodologies. It needs to be the first step considered in the management of a site.

In addition to the LID principles detailed above, a further innovative consideration is one associated with limiting the season within which construction activities can occur. A policy of this nature is utilized in Auckland, New Zealand, where construction activities over a certain size require specific approval to continue work over the winter

Roof area of flocculation unit

months. The rationale behind this practice is the increase in rainfall which is expected over the winter period in addition to lower ground temperatures and reduced evapotranspiration, which creates some difficulties in establishment of vegetative cover for stabilization. Dependent upon the site vicinity to receiving environments, many construction activities can not continue under these conditions due to the increased sediment yields and the difficulties that will be experienced.

While the practice of not working over this high-risk period is one that needs to be considered on a site-by-site basis, it does provide a further tool in the erosion control tool box and can go a long way toward prevention of sediment discharge and the associated problems.

Watershed-Wide Considerations

While this chapter focuses primarily on development activities, it is important that the issue of sediment discharge from development activities is not considered in isolation. Pollution (sediment) budgets on a watershed-wide basis may assist in determination of the prime sources of pollution within the watersheds of concern. This concept is one that needs further consideration, and while it would provide good generic information, it is recognized as being a difficult study to undertake.

The above demonstrates some of the innovative practices that currently exist and can be employed as part of an erosion and sediment control plan. There are many more practices in this category, and programs need to consider them in association with the traditional measures available.

Ten Commandments of Erosion and Sediment Control

It is important that all the principles and practices that apply be put into the context of a program and site development. To assist in this process, a set of "commandments" can be considered on all development sites. These "commandments" have been adopted by many programs and are utilized on a regular basis when erosion and sediment control are considered. They provide a checklist and also demonstrate the key aspects that should always be considered.

1. Minimize Disturbance

Some parts of sites should not be opened up, and where construction is required, ensure this is undertaken carefully to avoid sensitive areas. These sensitive areas include wetlands, streams, and steep slopes. This component is clearly linked to the planning phase of all developments, and if you can plan to only undertake earthworks on land that is suitable for this activity, you will go a long way toward reducing accelerated erosion. LID also

Minimal disturbance for earthmoving

attempts to limit total site disturbance. Working with existing site contours, as opposed to mass earthworking, will reduce overall site sediment discharge.

2. Stage Construction

Where possible, stage construction such that it is undertaken in manageable segments that can include revegetation and therefore limit the erosion potential. Sites that expose the whole area at one time constitute a considerable risk with a significant potential for erosion of large sediment yields.

3. Protect Steep Slopes

Associated with minimization of disturbance, steep slopes should always be avoided. This is clearly demonstrated by the Universal Soil Loss Equation, according to which these slopes can generate the biggest percentages of sediment yields. Runoff should also be diverted away from these slopes. Where slopes will be disturbed and revegetation is required, techniques over and above those traditionally utilized will need to be considered.

4. Protect Watercourses

Again associated with the minimization of disturbance, existing streams and drainage patterns need to be identified and protected as part of the planning phase of the development cycle. These systems are critical components of our receiving environments. If work is required that disturbs these areas, specific careful management is required.

5. Stabilize Exposed Areas Rapidly

The best way to prevent erosion is to fully stabilize the soils to prevent raindrop impact and scour. This may require stabilization during the development as well as stabilization at the completion of the earthworking phase. To provide the vegetative cover required, it may be necessary to look further than tradition conventional grass sowing and move toward methodologies such as straw mulching as a standard practice.

6. Install Perimeter Controls

The key behind effective sediment control is to treat only the runoff that is required to be treated. Essentially, treat only dirty water and keep clean water clean. The best way of achieving this is to employ perimeter controls that divert clean water safely to a point of discharge. Perimeter controls can also act to ensure that dirty water does not leave the site, as well as direct sediment-laden water to the necessary controls within the site.

7. Employ Detention Devices

A further critical feature associated with erosion control measures is the use of sediment traps and ponds. These work on the principle of detaining sediment-laden runoff, and while never 100 percent effective, they are a critical component. These measures need to be designed to local standards and must be able to withstand the rainfall conditions for the area while not overtopping.

8. Register and Attend Local Contractor or Designer Education Courses

9. Make Sure the Erosion and Sediment Control Plan Evolves as Site Development Proceeds

10. Assess and Adjust

These last three commandments refer to ensuring that you have appropriately trained personnel on site, that your erosion and sediment control plan adjusts as the site evolves, and that continual monitoring and assessment of the site occurs.

These "ten commandments" can be easily transferred to any program. It is important to note the emphasis they place on nonstructural techniques. Since these principles focus on the prevention of erosion in the first instance, they take pressure off the sediment control measures. This does not mean that the sediment control measures are not necessary, but that maintenance and overloading of these structures is not as frequent if there is less sediment entering.

Essential Program Elements

For the erosion and sediment program to be successful, it is critical that its institutional aspects evolve along with its technical aspects.

The ultimate goal of any erosion and sediment control program is to minimize or reduce adverse water quantity and quality impacts. This cannot be accomplished without an effective, efficient, and comprehensive institutional foundation. There must be adequate legal authority, performance standards, design assistance and guidance, program funding and staffing, commitment to enforcement, comprehensive approaches to research, and program evaluation and evolution. All of these program elements must have a solid institutional foundation that exists prior to any practices being constructed. In addition, regardless of the best of intentions, the program must have political support, which is translated into funding and other necessary program support components.

Essential program elements include:

- Basic goals;
- Authority and implementation structure (relationship and linkage to other local government programs);
- Performance standards;
- Exempted and waived activities;
- Design guidelines and assistance;
- Inspection procedures;
- Funding;
- Staffing;
- Educational activities;
- Compliance and enforcement;
- Maintenance; and
- Evaluation and evolution.

Design Plans

Detailed plans should be submitted and reviewed for larger site disturbances. The threshold size of the site in question should relate to the use of practices that require engineered design. If sediment traps are used, detailed designs should be submitted for review, and formal local agency approval should be obtained prior to works commencing. This threshold could also relate to areas where watershed size exceeds the ability of a particular practice to treat the runoff. An example of this is a silt fence, where concentrated flow would exceed design criteria.

Inspection Documentation and Frequency

An integral part of the inspection process is the question of how often inspections are considered necessary. During the construction process, site conditions can change rapidly, and assurance of adequate site control may necessitate frequent site visits by the inspector. Inspection frequency needs to be flexible, corresponding to shifts in the intensity of activity going on at construction sites.

When active construction is occurring, erosion and sediment control inspections should be conducted on a specified, appropriate frequency. This frequency will depend on the level of activity and may be developed during the construction period. When work on the site stops temporarily, inspections should still occur periodically to ensure that work has not resumed and that erosion and sediment controls are being maintained and still working. That inspection can be a 'windscreen inspection' to verify that construction has not initiated. It would be good to conduct a walk-around inspection at least once a month or after significant rainfall events to ensure that site controls are still functioning as required.

After completing the inspection, the inspector should leave an inspection report with the contractor, sending a copy to the developer and possibly the property owner. The report should serve as a site report card, clearly documenting proper installation and maintenance of site controls as well as any deficiencies in site control implementation. If there are areas of non-compliance, the inspection report initiates a "paper trail," which is integral to successful enforcement actions. To improve the effectiveness of inspections, it is important to establish standard, well-documented inspection procedures. These procedures should specify in detail the actions an inspector conducts at a site, set out options and list steps to be taken when site compliance is inadequate, and establish a process to be initiated if there is a disagreement on site.

Clarification of Roles on Site

A clear, formal statement of individual responsibility always benefits program implementation. The agreement should clearly define roles and levels of responsibility. When setting out these roles, it is important to be cognizant of the legal responsibilities of each authority.

Adequacy and Use of Guidelines

Program implementation will only be effective if there is a 'level playing field' where everyone has an equal responsibility to implement erosion and sediment control practices. There are several approaches that need to be jointly considered:

- Educational activities;
- Mandatory requirements where the regulated community recognizes their obligations; and
- Inspection and enforcement. A site presence will demonstrate to contractors and developers that program implementation is important. Enforcement procedures are also important in public recognition of program responsibility and obligation.

Adequacy of Erosion and Sediment Control Implementation

The purpose of program implementation is to ensure good site control to minimize sediment discharge into receiving waters. Program structure can take many forms, but it is the end result of resource protection that is the reason for program implementation.

Erosion and sediment control practices can be considered individual components of an environmental treatment train. Site control cannot rely upon one single practice to provide effective control. It will take a number of practices working in conjunction with one another to achieve that goal.

Permit Processing

Permits are a key tool utilized to enable erosion and sediment control objectives to be transferred and implemented on specific sites. Therefore, processing of permits is a very important first step in ensuring that these objectives are reflected on the ground and that all relevant parties understand the key issues and solutions associated with the development of a site.

The Erosion and Sediment Control Law is the key to minimizing adverse effects from construction and therefore needs to be considered "up front" by the developer and appropriately approved by the respective local authorities. This will ensure that controls required are documented and form a condition of the permit, thus making compliance and enforcement an easier task. In terms of the type and level of information required, this should include: contour plan, measures proposed, design criteria and justification, construction sequence, staging details, and maintenance. By including all this information on one plan, the contractor can also access the detail without second-guessing.

An environmental effects analysis is a second area of information that could be supplied as part of the permit application. This enables the developer to gain an understanding of the values of the receiving environment and adjust the erosion and sediment control plan accordingly to reflect this. The process of the developer undertaking this task also has a significant educational effect. Not only will it eventually change the attitude of developers toward erosion and sediment control, but it will also ultimately lead to significant improvements to site implementation. It also needs to be mentioned that the immediate receiving environment, such as on-site stream systems, needs careful consideration.

It is important that the development cycle is considered as a whole and sight is not lost of other key areas. The obvious area in this regard is the long-term stormwater discharge that is detailed in other parts of this document. There is little value in implementing a satisfactory erosion and sediment control program only to find all the benefits compromised by ineffective stormwater management over the long term.

Environmental Goals and Clear Guidance

Development and documentation of clear goals for the various receiving environments is critical. This should also include detail of the steps that can be taken to achieve these goals. This information could be included within an updated Erosion and Sediment Control Guideline.

Permit Inspection and Enforcement

Ensuring compliance with program requirements makes for a level playing field where all the players have the same responsibilities. Failure to take enforcement action can lead to widespread problems on other sites.

Enforcement is made more difficult by the fact that no one wants to be considered a "bad guy." Inspectors and program administrators need to recognize that, at times, they will have to act as "policemen." To facilitate these actions, the programs framework should specify the procedures, options, and remedies to be followed by staff when conducting compliance and enforcement activities.

As-built Requirements/ Pre-construction Meetings

As-built plans, especially for sediment ponds or traps, are extremely helpful in assessing the adequacy of implementation. Sediment ponds or traps are structural practices, and their performance depends to a large extent on adequacy of construction. Will there be leakage around the outfall pipe, has compaction been adequately done, and are design elevations reflected in construction? Sometimes these questions cannot be answered by an inspector, and having as-built requirements will provide further assurance that construction was adequately done.

Pre-construction meetings between the inspector, developer, designer, and contractor can provide an excellent starting point, where any questions regarding site implementation, timing, and phasing can be resolved. Important elements of erosion and sediment control implementation can be emphasized. Pre-construction meetings are important in getting a project off to a good start. Too often, site construction gets quickly out of control, and getting effective erosion and sediment control implementation becomes extremely difficult.

Mandatory or Voluntary Educational Programs

Generally, educational programs should be voluntary. However, there are certain program elements that can benefit from, or even depend on, mandated educational programs. An example of this would be a requirement that a responsible person from an individual contracting company attend an erosion and sediment control training program. This program could last one day and explain why implementation of erosion and sediment control is important and how to construct individual practices. The States of Delaware and Maryland, for example, have a mandatory contractor certification program that requires every site contractor to have at least one individual responsible for site controls attend a course in erosion and sediment control. To date, thousands of people have attended these programs, and they have proven popular with attendees.

Individuals attending these programs generally enjoy outdoor, water-related activities, and relating these activities to the program's goals leads to a more personal commitment by attendees. This greatly enhances program effectiveness.

If an educational program is mandatory, it must be available on a regular basis. This allows individuals who need this training to attend sessions and carry out their function under the program. Educational activities for the general public generally cannot be offered on a regular basis but rather when opportunities become available.

Educational programs aimed at the construction industry present a special challenge because of the constant turnover of employees. This implies a need for courses to be held on a more frequent basis. A contractors' course can last from a half day to one day. It needs to stress general information about erosion and sediment control as well as problems and solutions, along with information about the contractors' specific responsibilities and obligations.

While design guidance manuals or guidelines are extremely important, consultants can benefit greatly from periodic workshops on design aspects. These workshops can explain the rationale behind practice selection and design criteria, provide supplementary, up-to-date information on designing practices, and include case studies that illustrate good and bad examples of design and use of practices. Having a good relationship with the design community reduces potential problems in all aspects of program implementation.

Workshops conducted on a periodic basis could be used not only to disseminate information, but also to obtain feedback regarding program implementation, conduct case studies, bring in experts to discuss a specific issue, and demonstrate new strategies or products.

Program Requirements

It has been emphasized throughout this chapter that erosion and sediment control is an important part of the development cycle. It needs to be undertaken with consideration of all the aspects previously mentioned.

There are, however, the big "Cs" of an erosion and sediment control program that, similar to the ten commandments, should always be considered. They are:

- COMPREHENSIVE management of land use, water resources, and infrastructure throughout a catchment is necessary.
- CONTINUITY of erosion and sediment control and stormwater management programs over a long period of time will be required to address these problems.
- COOPERATION between all statutory bodies, the public sector and the private sector is essential to prevent and solve problems.
- COMMON SENSE in our institutional approach is essential.
- COMMUNICATION is crucial: between entities involved in program implementation; between implementing agencies and those being regulated; with politicians to gain their support; and with the general public to convey how normal activities can cause pollution and how to become part of the solution.
- COORDINATION of efforts for cost-effective implementation to maximize benefits is indispensable.
- CREATIVITY in technology and in our approach to solving this complex problem is critical.
- CASH in terms of program support and implementation of necessary controls is essential.
- COMMITMENT to solving these problems is of the utmost importance. Whether our children will have clean water, a high quality of life, and a vibrant economy will depend on our sincerity of effort.

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CHAPTER 10

Structural Stormwater Management Facilities

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Structural Stormwater Management Facilities

The full range of stormwater best management practices (BMPs) currently available to manage urban runoff includes both nonstructural measures and structural facilities. Nonstructural stormwater management measures, which can influence the amount of runoff and associated pollutants that are produced from a storm event, are discussed in detail in Chapter 7. This chapter presents technical information on the wide range of structural stormwater management facilities that are presently available to the developers and administrators of urban runoff management programs.

Unlike their nonstructural counterparts, structural stormwater management facilities generally do not influence the quantity and quality of stormwater runoff initially produced by rainfall. Instead, they respond to that initial runoff in a variety of ways, depending upon their basic operating principles and structural characteristics. These principles and characteristics include the relative ability of each to control runoff quantity and improve runoff quality, the mechanisms and materials they use to do so, the means by which they discharge their outflows, and their method of construction. As a result, these basic principles and characteristics can also be used to group the numerous types of structural facilities into broader categories as well as identify key design, construction, operation, and maintenance needs of each.

Compared to the 1994 *Fundamentals of Urban Runoff Management* that was the forerunner of this book, presenting a detailed chapter on structural stormwater management facilities is more difficult today. This is due to a number of reasons. In the intervening decade, the range of available types of structural facilities and the number of design variations have grown considerably. In addition, research into structural facility components, performance, operation, cost, construction, and ancillary impacts has also grown by a similar amount. These two factors have, in turn, increased the amount of available information that could be included in such a chapter. In addition, this knowledge growth rate is expected to continue in the future, increasing the likelihood that any information included in the chapter will quickly become obsolete. Finally, unlike 1994, there are many excellent structural facility design manuals that are readily available today from other public and private sources.

Therefore, rather than providing detailed design, construction, and maintenance specifications for each of the many structural stormwater management facilities currently available, this chapter will assist readers in selecting the most appropriate facilities, whether its for a particular development site or an entire urban runoff management program. It will do so by reviewing the basic operating principles and associated component and site needs of each facility type. It will then use this information to organize the large number of individual types into more general categories that should make it easier to both understand and select the optimum structural facilities. Finally, the chapter will provide an overview of key planning considerations and design requirements for each type of facility. This information will be supplemented by references to specific structural facility design manuals prepared by others that can be used to obtain for more detailed information.

Providing assistance in selecting the most appropriate structural stormwater management facility or facilities is important for several reasons. In addition to addressing the specific requirements of a development site or the overall needs of an urban runoff management program, it should be noted that the EPA's Stormwater Phase II Final Rule requires operators of small municipal separate storm sewer systems (or MS4s), which include municipalities and highway agencies, to develop and implement a runoff management program that utilizes best management practices (BMPs) to address post-construction runoff from land development and redevelopment sites. These BMPs are to include a combination of nonstructural measures and structural facilities, which the MS4 program developers are required to identify. NPDES permitting authorities, typically comprised of state governments and tribes, are also encouraged to develop a menu of BMPs to assist MS4 program developers in this process. As such, it can be seen that selecting the most appropriate structural BMPs for a runoff management program is an important component of the program's overall success.

Prior to the presentations on BMP selection, however, the chapter will begin with a discussion of two key design parameters that pertain to all structural stormwater management facilities:

1. The maximum rainfall for which runoff must be treated.
2. The minimum level of runoff treatment that must be provided.

In the past, the maximum rainfall amount, often referred to as the facility's design storm, has at times been the source of confusion and misunderstanding, which has affected the performance of both specific structural facilities and the overall urban runoff management program that created them. Similar confusion over required or attainable treatment levels during that design storm (and smaller events) has also adversely impacted both facility and program performance, albeit for somewhat different reasons. The discussions will attempt to reduce some of this confusion and promote better understanding of these two fundamental program parameters.

Design Storms

As described in the EPA's Stormwater Phase II Final Rule, urbanization can adversely impact streams, ponds, lakes, and other water bodies in two general ways. The first is caused by an increase in the type and amount of pollutants in stormwater runoff, which can harm aquatic life through both direct physical contact and their food chain. These impacts are described in detail in Chapter 3. The second cause is the increase in the volume of runoff that is delivered to and must be conveyed by water bodies. As described in Chapter 2, excessive increases in runoff volume can cause erosion and scour which, in turn, can harm aquatic life through a loss of both habitat and food sources, even without an increase in pollutant loading. In addition, excessive volume increases can cause flooding that can damage property and threaten human safety.

As a result, an effective urban runoff management program must address the impacts of land development and redevelopment on both runoff quality and quantity. However, while these requirements are easily stated and understood in general terms, there are some specific program requirements that must be determined in order to achieve them. One of these requirements is the exact amounts of rainfall and/or runoff that the program must address, both for runoff quality and quantity purposes. However, in attempting to do so, some complications arise. First, as discussed in Chapter 2, the amount of rain produced by a given storm event can range from a trace amount to a foot or more, depending upon location, season, and other meteorological factors. The amount of resultant runoff from these rainfalls can also vary, depending not only upon the specific rain depth but a number of other factors unassociated with the rainfall. The fact that rain events generally vary in a random way from storm to storm further complicates our efforts to select appropriate rainfall or runoff amounts.

One could turn to the EPA's Stormwater Phase II Final Rule for guidance. However, a review of the requirements for post-construction stormwater management reveals only general language regarding the need to develop a stormwater program that will reduce runoff pollution from land development and redevelopment projects to the maximum extent practicable. There is no mention of maximum or design storm depths, durations, or frequencies upon which to base specific program requirements, other than some discussion of controlling pollutant impacts on an average annual basis. As stated elsewhere in the final Rule, this lack of specificity on EPA's part is, in fact, deliberate, since it allows states and other regulated entities to develop program requirements and design parameters that specifically address their unique stormwater needs and problems. It also allows the EPA to improve and expand their stormwater regulations in an iterative manner, with greater specificity included in future versions of the Rule based upon the knowledge gained from previous ones. Unfortunately, while such customization and ongoing improvement of a nationwide rule is commendable, it also complicates the present task facing local program developers.

We can begin to find some answers by dividing up the urban runoff management program into the two major components described above: runoff quantity and runoff quality control. These are addressed separately below.

Runoff Quantity Control

From a risk perspective, increases in runoff quantity in the form of flooding can have direct impacts on all forms of life, including human. Therefore, while the potential for a loss of human life due to development-induced flooding may be relatively small, the effect or cost of such a loss is unquestionably large. As a result, the maximum design storm level for runoff quantity control must also be large in order to reduce the risk to an acceptably small amount. For that reason, many urban runoff management programs typically select large maximum design storm levels to address the quantitative runoff effects of land development and redevelopment. In keeping with the requirements of the National Flood Insurance Program (and many associated state floodplain management programs), the 100-year storm has typically been used as the maximum design storm for the control of runoff quantity increases.

However, to effectively manage runoff quantity increases caused by land development and redevelopment, it is not sufficient to simply exercise control over the maximum design storm. That is because runoff quantity increases caused by land development can also be expected to occur during smaller storms, and such increases, combined with their increased frequency of occurrence, can also cause significant flood and, in particular, erosion damage. In fact, on a percentage basis, the runoff volume increase experienced during smaller storms is typically greater than that which occurs during the maximum design storm. As a result, runoff quantity control must extend over a range of storm events up to and including the maximum design storm, which should be seen only as the upper limit of a range of necessary control.

For these reasons, it is common for an urban runoff management program to require runoff quantity control for a number of specific storm frequencies. For example, the New Jersey Stormwater Management Rules promulgated by the state's Department of Environmental Protection require runoff quantity control for the 2-, 10-, and 100-year storm events. In Maryland, the state's Department of the Environment requires runoff quantity control for the 1-, 10-, and 100-year storms in certain areas of the state to provide channel, overbank, and extreme flood protection. Many other states, counties, and municipalities have similar quantity control requirements.

However, although an appropriate range of design storms have been identified for runoff quantity control, the necessary level or degree of control must still be determined. To do so, we must first consider what runoff quantities are actually increased by land development and what can be done to either prevent or mitigate them. As discussed in detail in Chapter 2, land development and redevelopment projects that increase site imperviousness and drainage system efficiency will cause increases in both the site's total runoff volume and the peak runoff rate. As described in Chapter 2, both research and analysis have shown that increases in either of these quantities can cause downstream flooding, erosion, and habitat damage. As such, to be effective, it will be necessary for an urban runoff management program to address the increases in both runoff volume and peak runoff rate. And since increases in peak runoff rate are caused, at least in part, by runoff volume increases, it should be possible with a single set of controls to affect both quantity parameters.

Which returns us to the question of what level or degree of quantity control is necessary. An obvious first choice would be to require no increases in total runoff volume or peak runoff rate between pre- and post-developed site conditions, applied to a range of storm events up to a maximum 100-year storm. And as discussed in Chapter 8, Impact Avoidance, this is perhaps the most effective way to prevent the environmental damage and safety threats posed by runoff quantity increases. However, while limiting post-development peak runoff rates to pre-developed levels has proven to be readily achievable at most development sites regardless of site characteristics, similar control of post-development runoff volumes has proven to be more difficult. This is due, in part, to the different processes needed to control runoff rate and volume and, in part, to the very nature of land development itself. Regarding the different processes, reduction of post-development peak runoff rates can be accomplished through the temporary storage and slow release of the developed site's runoff. This can be accomplished through the construction of stormwater detention basins and related facilities that typically do not require any special site conditions other than sufficient space for construction and a downstream discharge point that can safely accept the basin's outflow. Both of these can usually be provided at most development sites with, at worst, a limited loss of developable land.

Reducing post-development runoff volumes to pre-developed rates, however, can be significantly more difficult if not impossible to achieve at many development sites. For unlike peak rate reductions that only require temporary storage of runoff, runoff volume reductions require what could be described as permanent runoff reductions. That is to say, reducing post-developed runoff volumes to pre-developed amounts requires the increased volume caused by the development to be infiltrated into the site's soils and not to be allowed to leave the site. While such infiltration can be possible at sites with granular, highly permeable soils with deep groundwater and bedrock levels, it can be difficult or impossible to achieve on sites with relatively impermeable soils and/or shallow depths to groundwater or bedrock. Even at permeable soil sites, achieving the required infiltration rates and volumes may be difficult due to the fact that the development has increased impervious coverage of the site, consequently reducing the area of pervious cover over which the infiltration can occur. Therefore, while the total volume of infiltration required under post-development conditions is essentially equal to pre-developed conditions, it must be achieved over a smaller area. This effectively increases the required soil infiltration rates, sometimes to unachievable levels. This problem can be compounded by excessive groundwater mounding, which can affect the infiltration facility itself and/or adjacent structures or systems such as basements and septic system disposal fields. Finally, the potential for groundwater contamination by the stormwater-borne pollutants infiltrated with the site's runoff may also prevent the use of infiltration.

As a result, the goal of maintaining pre-developed site runoff volumes, however desirable, has proven elusive in many areas. As a consequence, alternative quantity control measures have been sought. The most popular to date appears to be simply requiring post-developed peak runoff rates to not exceed pre-developed ones for a range of storm events through the use of on-site stormwater detention. However, despite both its popularity and apparent logic, research conducted in New Jersey and elsewhere has shown that, in addition to ignoring runoff volume increases, such a requirement can be, at best, ineffective and, in certain instances, actually detrimental: in a number of cases, downstream peak runoff rates turned out to be greater than those that would have occurred if the requirement (and associated on-site detention) had not been imposed. The results of one key research effort are summarized below.

The South Branch Rockaway Creek Stormwater Management Study was conducted by the Natural Resources Conservation Service (NRCS) and the New Jersey Department of Environmental Protection (NJDEP) in 1986. Among its many findings, the study demonstrated how ineffectual and, at certain locations, harmful a policy of maintaining post-development runoff rates at pre-developed levels can be. The study analyzed the 12.3 square mile watershed in west-central New Jersey under present (i.e., 1986) and ultimate development conditions. It then simulated the effects of a runoff quantity control policy that simply required no increase in predeveloped peak runoff rates for a land development site.

The results of that simulation are summarized in Tables 10-1, 10-2, and 10-3. Shown in the tables are the estimated peak 2-, 10-, and 100-year discharges at three of the study's eight points of analysis for both present and ultimate levels of development. The three points of analysis are located at the upper, central, and lower portions of the watershed as shown in Figure 10-1. The upper location (Point 1) has a total drainage area of approximately 0.6 square miles and has only a single subarea discharging to it, while the central and lower locations (Points 4 and 8, respectively) have increasingly larger drainage areas and, as such, receive the outflows from increasingly greater numbers of subareas. Point 4 has a total drainage area of approximately 7.3 square miles and receives runoff from 16 watershed subareas, while Point 8, which is located near the mouth of the South Branch, has a total drainage area of 12.3 square miles and receives runoff from all 23 subareas delineated for the study.

The tables contain three sets of peak discharges at each location for each storm frequency. The first set of peak discharges represents existing development levels at the time of the 1986 study, while the second set represents ultimate development of the watershed in accordance with current zoning but without any runoff rate controls. The third set of peak discharges also represents ultimate development conditions but with the requirement that peak runoff rates from future developments could not exceed those under existing development levels. These requirements, which were contained in the current land development ordinances of the watershed's municipalities, were achieved in the study by allowing each watershed subarea to represent a future development site, each with its own on-site detention basin that achieved the required peak rate reduction for the 2-, 10-, and 100-year storms.

As can be seen in all three tables, ultimate development of the South Branch Rockaway Creek watershed will cause increases in existing 2-, 10-, and 100-year peak runoff rates at all three points of analysis summarized in the tables. For example, Table 10-1 indicates that the existing 2-year peak runoff rate at Point 1 is estimated to increase from 136 CFS to 186 CFS under ultimate development conditions. Similarly, Table 10-3 indicates that the existing 100-year peak runoff rate at Point 8 is estimated to increase from 3840 CFS to 4660 CFS under ultimate development conditions. Similar increases can be seen in the tables for all points of analysis and storm events.

Further review of the tables also indicates how effective the watershed's 1986 peak runoff rate controls will be. As described above, these runoff quantity controls required peak developed site outflows not to exceed those under existing development for the 2-, 10-, and 100-year storms. At Point 1 (which is located in the upper portion of the watershed at the outlet of a single development site), this requirement will effectively control the peak increases caused by that development at that location for all three storm events. For example,

Table 10-1: Summary of Peak 2-Year Discharges, South Branch Rockaway Creek Stormwater Management Study

Point of analysis	Existing peak discharge (CFS)	Ultimate development without peak controls		Ultimate development with peak controls	
		Peak (CFS)	% of existing peak	Peak (CFS)	% of existing peak
1	136	186	136%	132	97%
4	594	901	152%	726	122%
8	419	690	165%	665	159%

Table 10-2: Summary of Peak 10-Year Discharges, South Branch Rockaway Creek Stormwater Management Study

Point of analysis	Existing peak discharge (CFS)	Ultimate development without peak controls		Ultimate development with peak controls	
		Peak (CFS)	% of existing peak	Peak (CFS)	% of existing peak
1	472	558	118%	464	98%
4	2100	2660	127%	2350	112%
8	1770	2280	129%	2250	127%

Table 10-3: Summary of Peak 100-Year Discharges, South Branch Rockaway Creek Stormwater Management Study

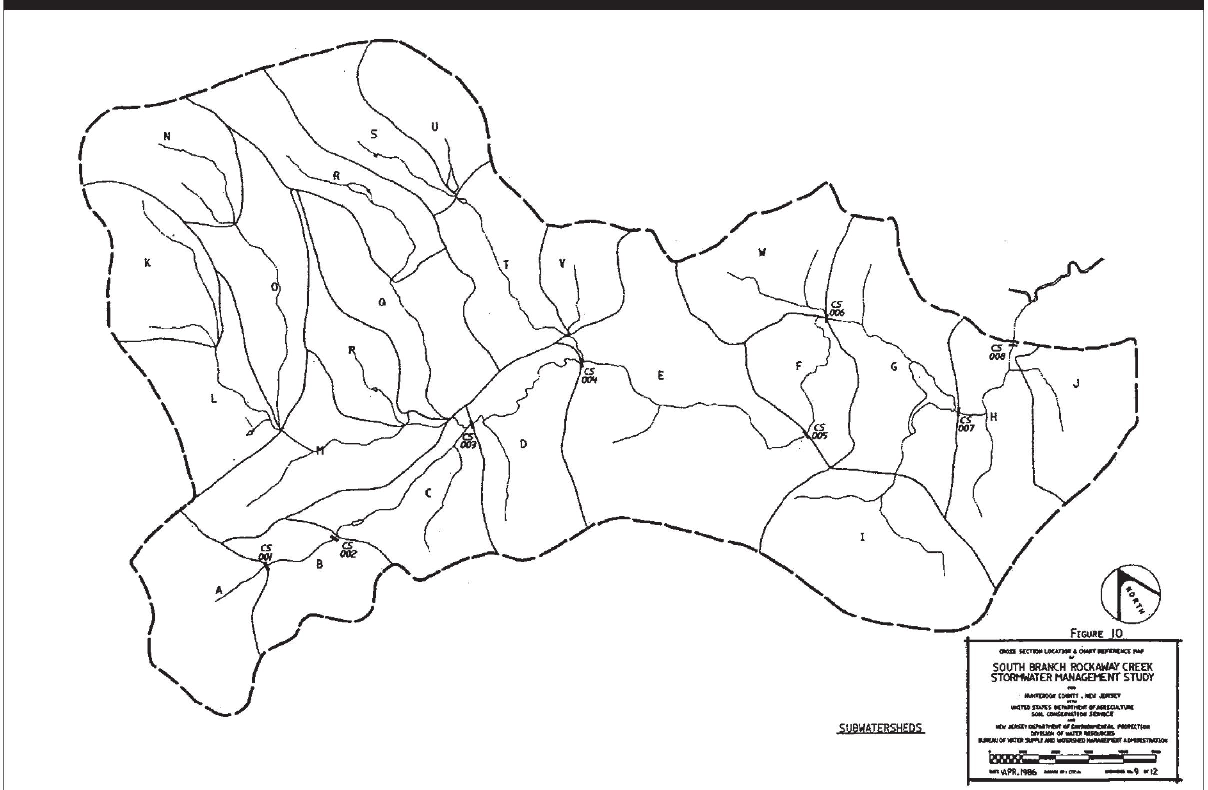
Point of analysis	Existing peak discharge (CFS)	Ultimate development without peak controls		Ultimate development with peak controls	
		Peak (CFS)	% of existing peak	Peak (CFS)	% of existing peak
1	896	1020	114%	882	98%
4	4320	5330	123%	4750	110%
8	3840	4660	121%	4710	123%

as shown in Table 10-1, the 2-year peak runoff rate at Point 1 following development in accordance with the peak rate controls described above would be 98 percent of the existing 2-year peak rate. Similar peak rates are achieved at Point 1 for the 10- and 100-year storms as shown in Tables 10-2 and 10-3, respectively.

However, further review of the tables indicates that the effectiveness of the peak rate requirement will decrease as the South Branch continues downstream through the watershed, receiving runoff from more development sites and their on-site detention basins. For example, Table 10-1 indicates that the developed 2-year peak runoff rate with peak rate controls at Point 4, which is located in the central portion of the watershed, will be 726 CFS or approximately 22 percent greater than the existing 2-year peak rate of 594 CFS. Continuing downstream, the developed 2-year peak runoff rate with peak rate controls at Point 8, which is located at the lower end of the watershed, will be 665 CFS or approximately 59 percent greater than the existing 2-year peak rate of 419 CFS.

Similar results can be seen in Tables 10-2 and 10-3 for the 10- and 100-year storms. As with the 2-year storm, the peak rate controls for both events are fully effective at Point 1 which is immediately downstream of a single development site and on-site detention basin. However, at lower locations in the watershed with multiple development site and on-site basins contributing flow, the developed peak rates begin to exceed the existing rates and approach those for developed conditions without peak rate controls. For example, as shown in Table 10-2, the developed 10-year peak runoff rate at Point 8 with peak rate controls will be 2250 CFS, which is not only 27 percent greater than the existing 2-year peak rate but only 30 CFS or 2 percent less than the estimated 10-year peak rate at this location without any peak rate controls. This loss of effectiveness is greatest for the 100-year storm at Point 8 where, as shown in Table 10-3, the peak developed runoff rate with peak rate controls of 4710 CFS will not only exceed the existing peak rate of 3840 CFS but also the developed peak rate without controls of 4660 CFS. In other words, not only have the peak rate controls failed to maintain existing peak runoff rates throughout the watershed following ultimate development, they have actually caused developed peak rates to exceed those that would have occurred if no controls had been imposed.

Figure 10-1: South Branch Rockaway Creek Watershed Points of Analysis

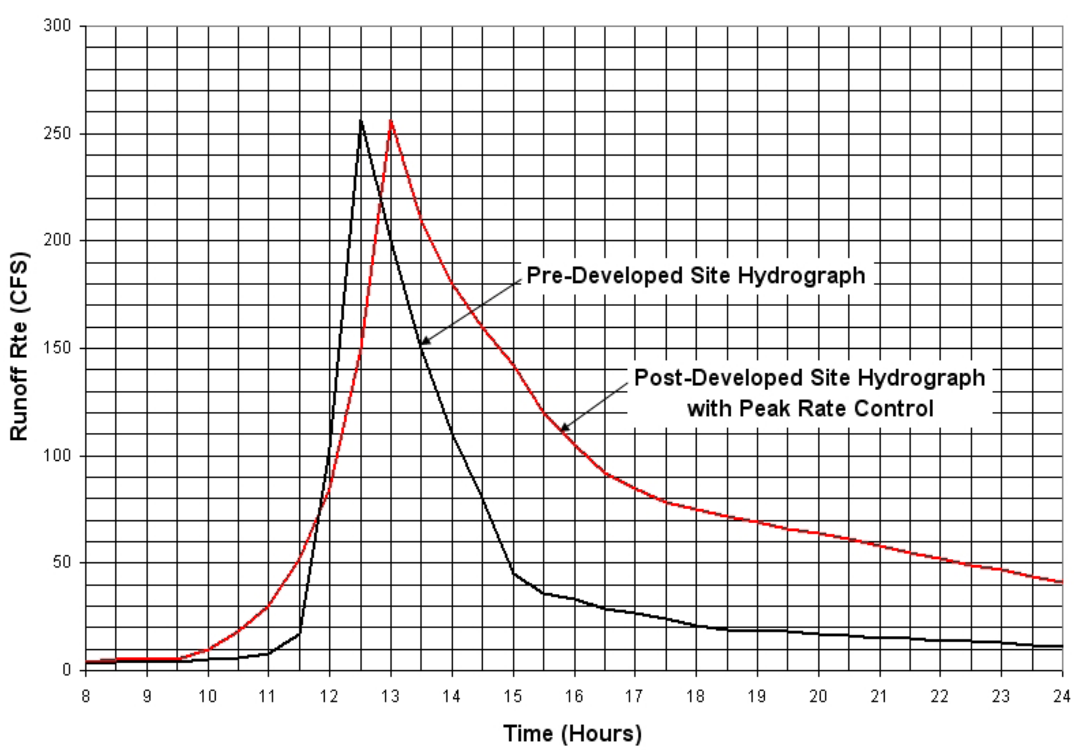


From the above, it can be seen that selecting a runoff quantity control that simply requires peak developed site runoff rates not to exceed those under existing or pre-developed conditions may prove ineffective. This ineffectiveness may even reach levels where the peak runoff rates produced by the controls, which were intended to prevent the adverse runoff quantity impacts of land development, actually exceed those that would have occurred if the controls had not been imposed. As a result, instead of preventing the adverse runoff quantity impacts of land development, the peak rate controls actually made them worse.

Based on an analysis of the South Branch Rockaway Creek study and the development of similar studies, several reasons can be found for these unfortunate results. They include alterations in the timing of runoff from land development sites caused by both the site's more efficient drainage systems and its on-site detention facility that achieves the required peak rate reductions. An additional and perhaps more significant reason for the ineffectiveness of the peak runoff rate controls is a failure to recognize and address the runoff volume increases that are also typically caused by land development. A simplified explanation of this complicated effect is illustrated in Figure 10-2. The figure depicts the runoff hydrograph from a development site under both pre-developed and developed conditions. The developed condition hydrograph is based on reducing the peak rate of runoff created by the development to a rate equal to the pre-developed peak. This is the same peak rate control used in the South Branch Rockaway Creek Stormwater Management Study discussed above and has been achieved by temporarily storing and slowly releasing the runoff hydrograph from the developed site similar to the South Branch study.

As can be seen in Figure 10-2, the pre- and post-developed peak runoff rates from the development site are indeed equal. However, further examination of the hydrograph shows several key dissimilarities. First, the two peak runoff rates occur at different times, with the pre-developed hydrograph peaking at Hour 12.5, approximately 0.5 hours before the post-developed one. This is due, at least in part, to the timing effects of the site's new drainage system and on-site detention basin. Second, due in part to this peak time difference, there is a period of approximately 0.9 hours (between approximately Hours 11.9 and 12.8) when the post-developed runoff rates are less than those for

Figure 10-2: Comparison of Pre- and Post-Developed Site Hydrographs with Peak Rate Control



pre-developed conditions. However, starting at Hour 12.8 and continuing beyond the end of the x-axis at Hour 24.0, the post-developed condition runoff rates exceed the pre-developed ones by amounts that range from approximately 25 percent to over 200 percent. From the detailed discussion in Chapter 2 regarding the quantitative impacts of urbanization, it can be seen that these increased runoff rates are the result of the increased runoff volume caused by the site development. This increased runoff volume (represented by the area beneath the hydrograph plot) requires the non-peak post-developed runoff rates to generally exceed those for pre-developed conditions.

In addition, these higher non-peak runoff rates are the primary reason why the developed peak rates with peak rate controls at Points 4 and 8 in the South Branch Rockaway Creek Stormwater Management Study exceeded those for existing or pre-developed conditions. Looking again at Figure 10-2 and assuming that it represents the runoff at Point 1, the following can be seen: if the peak runoff rates downstream at Points 4 and 8 were produced, in part, by the runoff at Hour 12.8 or later at Point 1, then the post-developed peak rates at these locations would exceed the pre-developed peak. That is because, as described above, the post-developed flows at Point 1 starting at Hour 12.8 will be greater than the pre-developed rates, even though the peak rate controls produced the same peak rate at Point 1. For example, if the pre-developed peak runoff rate at Point 4 is produced, in part, by the pre-developed rate at Point 1 at Hour 14, then the post-developed rate at Point 4 can be expected to increase (despite the same peak rate at Point 1) because, as shown on Figure 10-2, the post-developed rate at Point 1 will be approximately 70 percent greater than the pre-developed one.

The above findings further highlight the value of maintaining both existing runoff peaks and runoff volumes following land development or redevelopment. However, recognizing how difficult it may be to achieve the site infiltration rates required to do this, how else may the problem of peak runoff rate increases downstream of development sites be addressed? Fortunately, additional watershed-level studies similar to the South Branch Rockaway Creek Stormwater Management Study offer some answers. These answers are illustrated below in a discussion of two such studies performed in New Jersey during the same time period as the South Branch study.

The two studies in question were performed to both confirm the peak runoff rate increase problem highlighted by the South Branch study and to develop answers to it. The studies were based in the Middle Brook watershed in Somerset County and the Devils Brook watershed in neighboring Middlesex County. Details of both studies are presented in the reference section of this chapter. Both watersheds are located in the central portion of New Jersey. The 16 square mile Middle Brook watershed is characterized by forest cover and steep ground slopes formed by the Watchung Mountains, while the 22 square mile Devils Brook watershed has significantly flatter ground slopes and a combination of forest cover along with existing residential and agricultural land uses.

In the Middle Brook Watershed study, the pre-developed land use was assumed to be entirely forest, while the Devils Brook Watershed study was based upon existing development levels at the time of the study. As a result of these factors, the two studies taken together are felt to be generally representative of the range of potential topographic, geologic, and land development conditions that may be encountered in an urban runoff management program. Finally, it should be noted that, due to generally fair to poor soil permeability in both watersheds and, in the case of the Middle Brook, shallow depth to bedrock due to the mountainous terrain, maintaining pre-developed runoff volumes was not considered feasible. This was particularly the case for the large runoff increases created during the 10- and 100-year storms whose control was vital to preventing existing erosion and flooding problems in both watersheds from worsening with future development.

Included in each study's scope was the investigation of an idea developed through the analysis of pre- and post-developed runoff hydrographs similar to those in Figure 10-2. The idea was this: Could the downstream runoff increases described above be avoided by reducing the post-developed peak runoff rate from a land development site to less than the existing rates? While such a control would not decrease the post-developed runoff volume, could a reduction in post-developed peak runoff to rates less than pre-developed conditions cause sufficient flattening of the post-developed hydrograph to maintain pre-developed peak rate not only at but downstream of land developments? The results of these two studies are summarized below.

First, both studies confirmed the problem of increased post-developed peak runoff rates highlighted by the South Branch Rockaway Creek Stormwater Management Study. For example, the Devils Brook watershed study demonstrated an increase in downstream peak runoff rates despite a peak rate control that maintained pre-developed peaks at each development site in the watershed. These results are illustrated in Figures 10-3 through 10-5. They depict for the 2-, 10-, and 100-year storms, respectively, the post-developed peak rates with the peak rate control described above at various points of analysis in the watershed expressed as a percentage of the pre-developed peaks at those locations. The points of analysis used in the study are numbered consecutively and increase in value in a downstream direction. As can be seen in Figure 10-3, the 2-year peak post-developed rates generally increase

Figure 10-3: Comparison of 2-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed

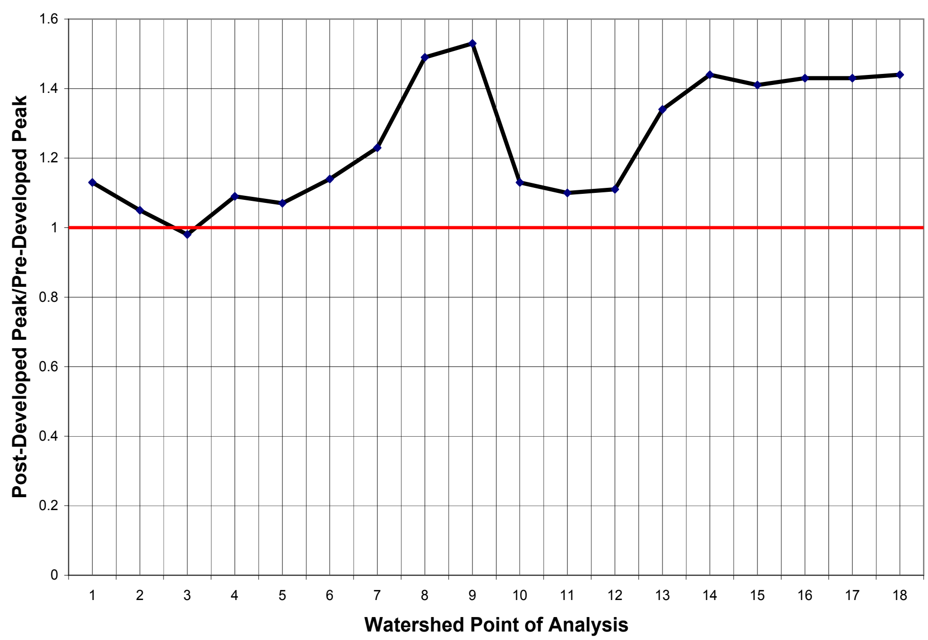


Figure 10-4: Comparison of 10-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed

in a downstream direction from the upper to lower portions of the watershed (i.e., the displayed values exceed 1.0) despite the peak post-developed runoff from each watershed subarea (again assumed to represent a future development site) being equal to the pre-developed rate. At Point of Analysis 18 at the mouth of the watershed, the post-developed peak runoff rate is approximately 42 percent greater than the pre-developed rate. Similar results are shown for the 10- and 100-year storms in Figures 10-4 and 10-5, with post-developed peak rates ranging as high as 20 percent to 45 percent greater than pre-developed. It should be noted that qualitatively similar results were also encountered in the Middle Brook Watershed study.

Having confirmed the peak rate increase problem caused by setting post-developed peak runoff rates equal to pre-developed rates, both studies then investigated the effectiveness of reducing post-developed peak runoff rates from land development sites to rates less than, rather than equal to, predeveloped peak rates. As noted above, existing erosion and flooding problems in both watersheds made effective runoff quantity control a vital requirement in both watersheds. Because of this need and the inability to control runoff volume increases through infiltration, it was necessary to develop effective peak rate controls that avoided the peak rate increase problems highlighted by the South Branch Rockaway Creek study. After several iterations, the peak runoff rate control criteria presented in Table 10-4 were selected for each development site in the watersheds. As shown in the table, the Middle Brook Watershed study investigated the downstream effectiveness of reducing the post-developed peak runoff rates from development sites for the 2-, 10-, and 100-year storms to 50 percent, 65 percent, and 80 percent, respectively, of the pre-developed peak site rates. In the Devils Brook Watershed study, the percentages of the pre-developed 2-, 10-, and 100-year peak runoff rates were 40 percent, 65 percent, and 65 percent, respectively.

The results of applying the Devils Brook peak rate reduction factors shown in Table 10-4 are illustrated in Figures 10-6 to 10-8. As shown in Figure 10-6,

Figure 10-5: Comparison of 100-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed

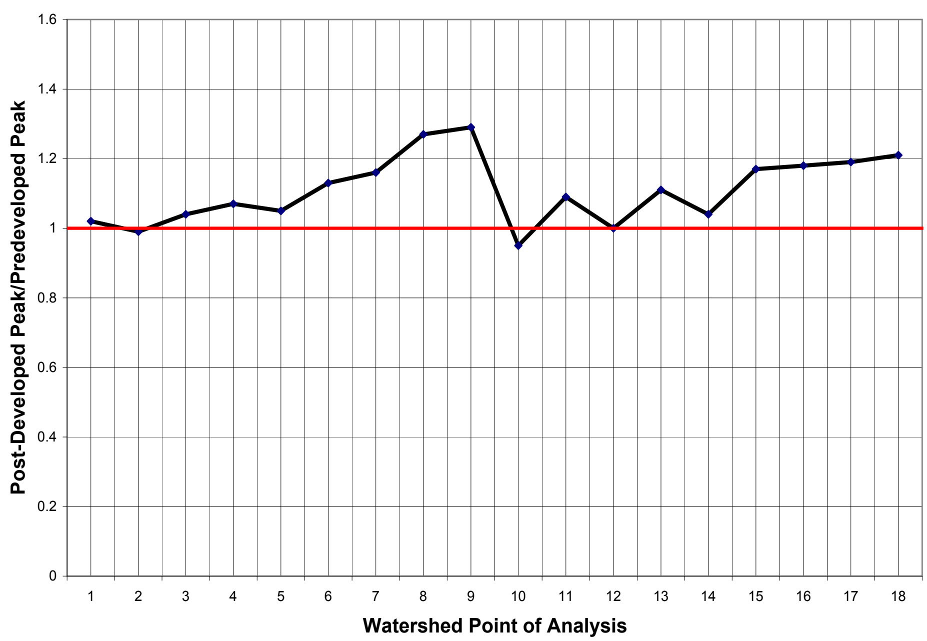


Table 10-4: Final Peak Runoff Rate Reduction Factors, Middle Brook and Devils Brook Watershed Studies

Watershed	Required Post-Developed Peak Site Outflow Rates Expressed as Percentage of Pre-Developed Peak Rate		
	2-Year Storm	10-Year Storm	100-Year Storm
Middle Brook	50%	65%	80%
Devils Brook	40%	65%	65%

requiring future development sites in the Devils Brook watershed to reduce their post-developed 2-year peak runoff rates to 40 percent of their pre-developed rates achieved the desired goal, namely, to not allow ultimate watershed development to increase pre-developed 2-year runoff rates anywhere in the watershed. Unlike the 2-year storm results shown in Figure 10-3, where post-developed peak site rates were allowed to equal pre-developed ones, Figure 10-6 shows that, by reducing peak site rates to 40 percent of pre-developed peaks, the post-developed 2-year peak runoff rates throughout Devils Brook remain at or below pre-developed levels.

Figure 10-6: Comparison of 2-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed

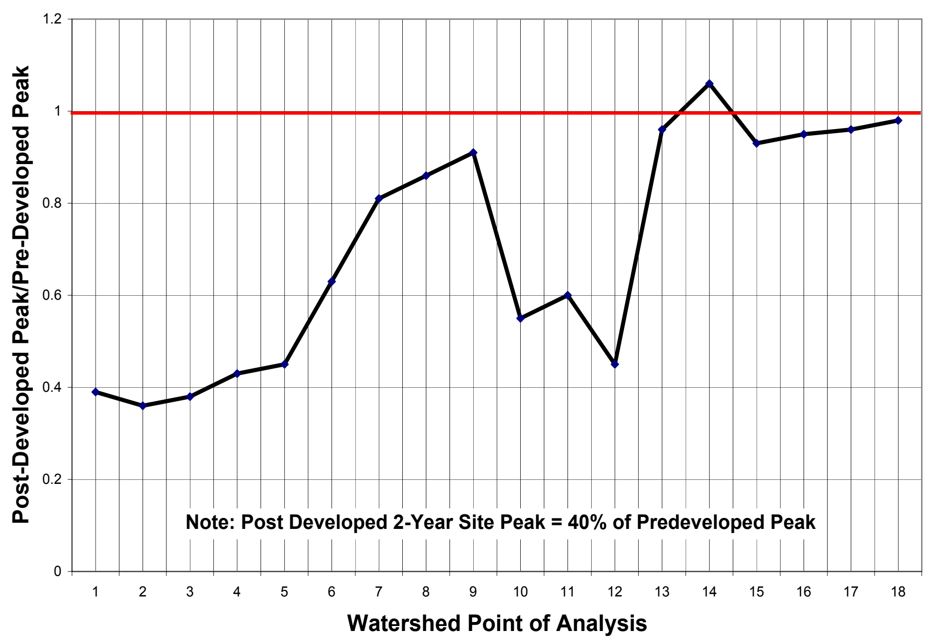
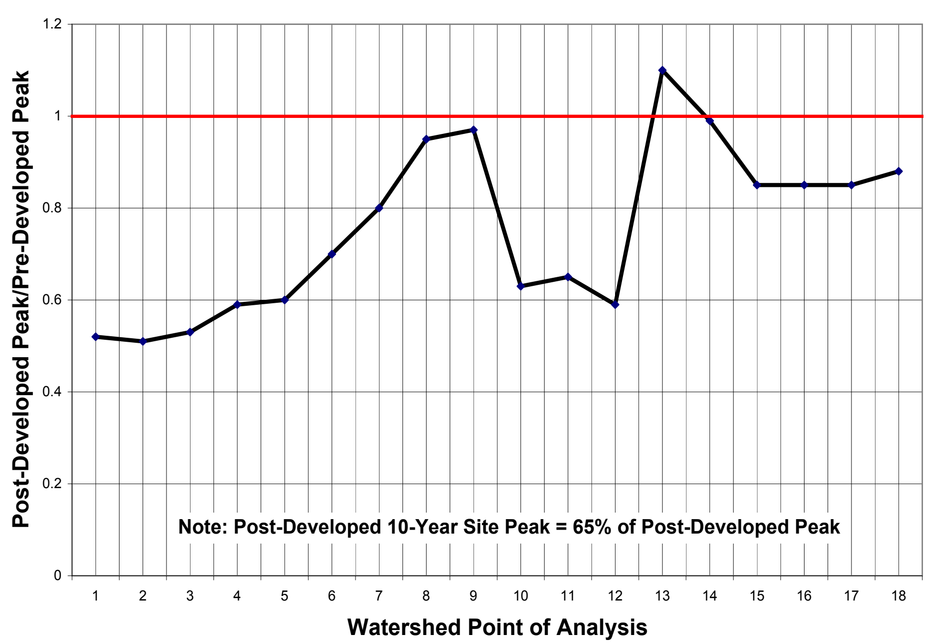


Figure 10-7: Comparison of 10-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed



A single exception to these results occurs at Point of Analysis 14 where the post-developed peak 2-year rate exceeds the pre-developed one by approximately 5 percent. Similar results are shown for the 10-year rates in Figure 10-7 with, again, a small exceedance at Point of Analysis 13. For the 100-year storm, Figure 10-8 shows that the post-developed 100-year peak runoff rates remain without exception at or below pre-developed levels throughout Devils Brook. This contrasts with the results shown in Figure 10-5, where allowing post-developed peak site runoff rates to equal pre-developed conditions resulted in post-developed peak increases

Figure 10-8: Comparison of 100-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Devils Brook Watershed

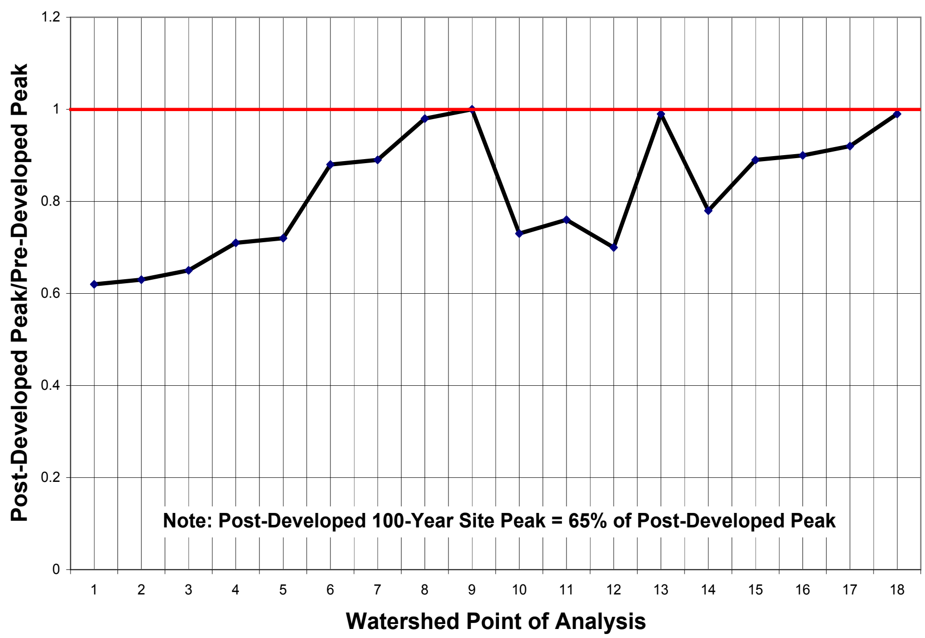
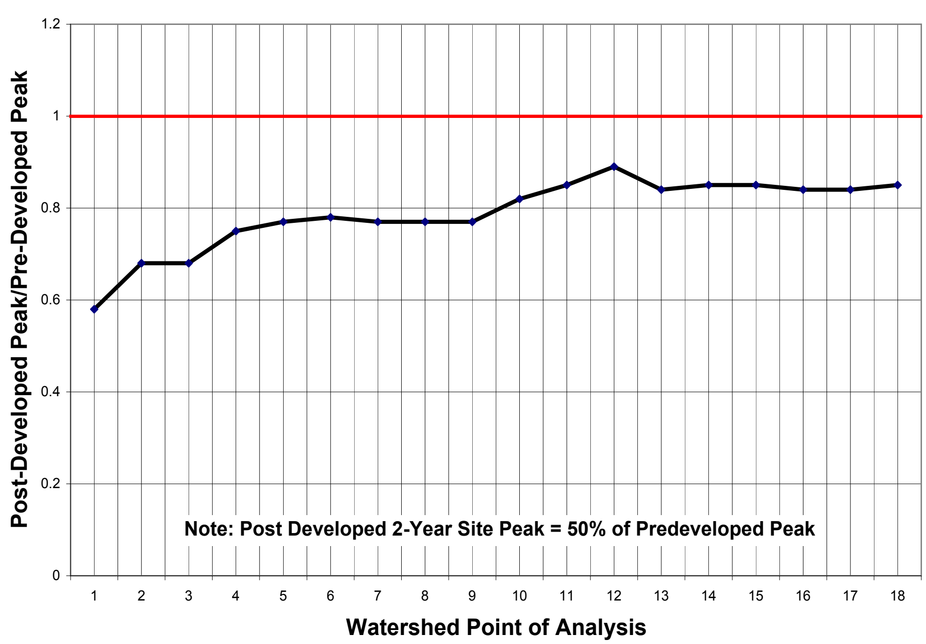


Figure 10-9: Comparison of 2-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Middle Brook Watershed



throughout most of the waterway, ranging as high as 30 percent at Point of Analysis 9.

The results of applying the Middle Brook peak rate reduction factors shown in Table 10-4 are illustrated in Figures 10-9 to 10-11. As shown in Figure 10-9, requiring future development sites in the Middle Brook watershed to reduce their post-developed 2-year peak runoff rates to 50 percent of their pre-developed rates also achieved the goal of not increasing pre-developed 2-year peak runoff rates anywhere in the watershed following ultimate development. Figure 10-6 shows that the post-developed 2-year peak runoff rates throughout

Figure 10-10: Comparison of 10-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Middle Brook Watershed

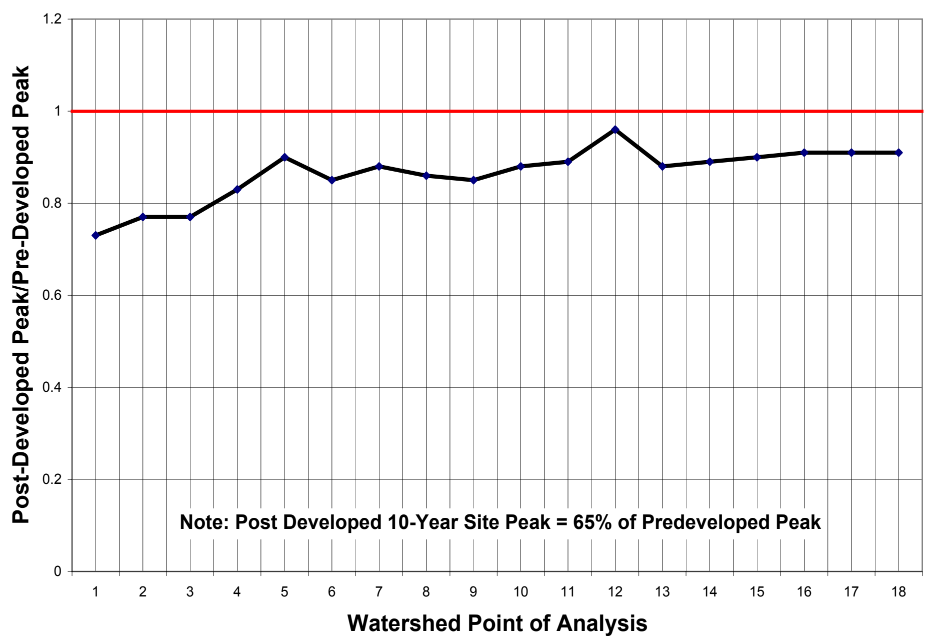
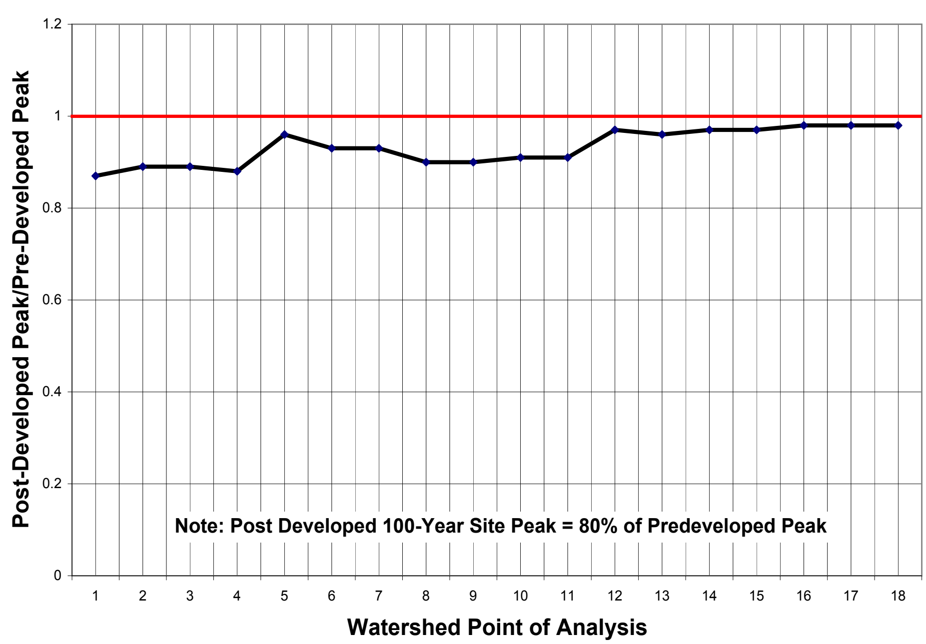


Figure 10-11: Comparison of 100-Year Pre- and Post-Developed Peak Rates with Peak Rate Controls, Middle Brook Watershed

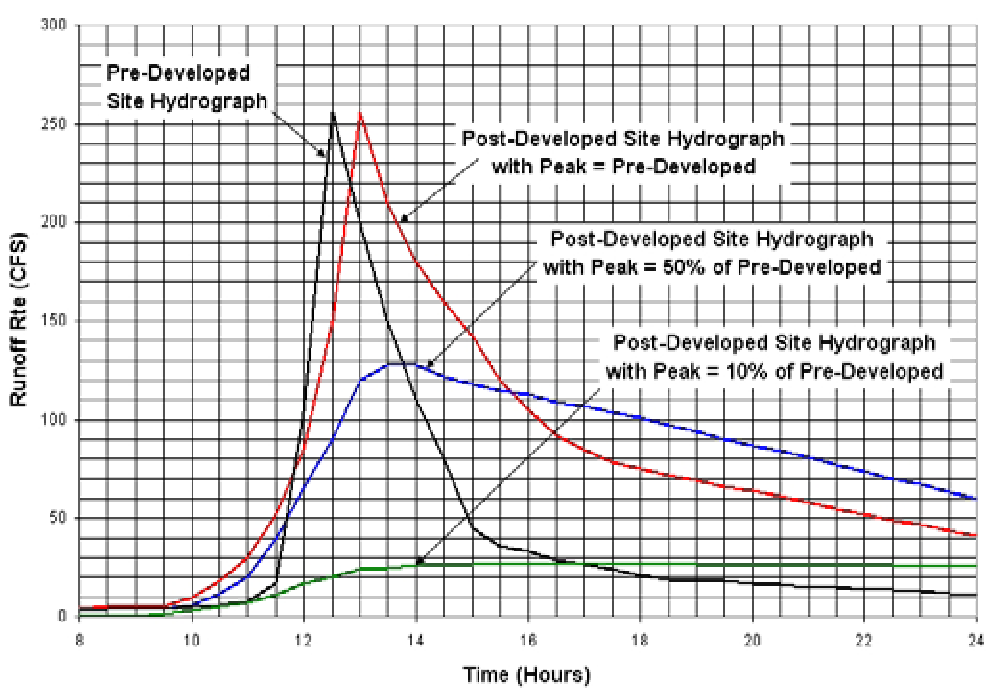


Middle Brook remain at or below pre-developed levels. Similar results are shown for the 10- and 100-year storms in Figures 10-10 and 10-11, where post-developed 10- and 100-year peak site runoff rates are required to be 65 percent and 80 percent, respectively, of the pre-developed peak site rates.

A simplified explanation of the reasons why the peak rate reduction factors used in the Middle Brook and Devils Brook watershed studies were effective in preventing downstream peak runoff rate increases is presented in Figure 10-12. The figure once again depicts the pre- and post-developed site hydrographs shown in Figure 10-2, with the peak of the post-developed hydrograph equal to the pre-developed peak. As noted in the discussion of Figure 10-2, the post-developed runoff rates for this level of peak rate control are less than pre-developed for only approximately 0.9 hours or from Hour 11.9 to 12.8. However, also shown in Figure 10-12 is a post-developed site hydrograph with a peak runoff rate equal to 50 percent of the pre-developed rate as analyzed in the Middle Brook study. An examination of this hydrograph shows that, under this level of peak rate control, post-developed runoff rates are less than pre-developed for approximately 2 hours (from approximately Hour 11.8 to 13.8), or more than twice as long as the post-developed site hydrograph with a peak equal to pre-developed. This increased time period offers greater opportunity for this and other post-developed site hydrographs with similar levels of control to combine downstream in such a way as to produce a total downstream peak that is no greater than the pre-developed peak at that location.

From the above, it becomes apparent that, at least for the study watersheds described above, simply requiring land development and redevelopment sites to match their pre-developed peak runoff rates will not prevent increases in peak downstream runoff rates. As such, the development-induced runoff quantity impacts of flooding, erosion, and habitat damage described in detail in Chapter 2 will not be avoided even with the imposition of runoff quantity requirements in the watersheds' runoff management programs. Based upon the results of the studies described above, the State of New Jersey has adopted statewide 2-, 10-, and 100-year peak rate reduction factors of 50 percent, 75 percent, and 80 percent, respectively, for all land development and redevelopment projects that disturb at least an acre of ground surface. These requirements are contained in the state's Stormwater Management Rules (N.J.A.C 7:8) promulgated by the New Jersey Department of Environmental Protection (NJDEP).

Figure 10-12: Comparison of Pre- and Post-Developed Site Hydrographs with Various Peak Rate Controls



At this point, it is important to note more recent research into the effectiveness of peak reduction factors to control runoff volume increases. From a flood prevention standpoint, limiting downstream post-developed peak discharge rates on a waterway to levels no greater than pre-developed (as achieved through the use of development site peak rate reduction factors described above) should be effective in preventing increases in existing waterway flood depths and limits. However, research conducted since the Middle Brook and Devils Brook studies, including McRae in 1997 and Bledsoe in 2002, indicates that limiting downstream waterway discharges to pre-developed levels may not be sufficient to prevent development-induced erosion damage along the waterway. This is due, once again, to the increase in total site runoff volume caused by the development (and highlights once again the desirability of preventing runoff volume increases where possible through site design and post-development runoff infiltration).

As shown in Figure 10-12, the two post-development site hydrographs with peaks equal to 100 percent and 50 percent of predeveloped will both have runoff rates that exceed pre-developed rates for an extended period of time. As noted above, this is due to the greater runoff volume under post-developed conditions, represented by the area under the hydrographs, that causes a longer overall duration of runoff than under pre-developed conditions. This longer runoff duration means a similarly longer period when the runoff will create shear and other forces along the downstream waterway's bed and banks. If these forces are greater at times than certain critical levels, which are based upon various waterway properties, erosion will occur even if the runoff rate is less than pre-developed. In fact, it can be seen that, as the peak post-developed site runoff rate becomes a smaller percentage of the pre-developed rate, the duration of post-developed runoff can be expected to increase. If this increased time includes longer periods of excessive channel forces, reducing post-developed site peak runoff rates may actually cause greater erosion than higher post-developed peaks.

In response to this research, some jurisdictions, including the State of Maryland, have adopted significantly lower peak rate reduction factors for erosion control than the 2-year 50 percent reduction factor required by New Jersey. As described in Chapter 2 of the October 2000 Maryland Stormwater Design Manual, the rationale for this greatly increased control of frequent storm events typically associated with waterway erosion is that "runoff will be stored and released in such a gradual manner that critical erosive velocities during bankfull and near-bankfull events will seldom be exceeded in downstream channels." Or, as one might add, at least not exceeded for durations longer than under pre-developed conditions.

This rationale is illustrated with the final post-developed site hydrograph shown in Figure 10-12. With a peak runoff rate equal to only 10 percent of the pre-developed peak, it can be seen that there is not only a considerably longer period when post-developed runoff rates are less than pre-developed (between approximately Hours 10 and 17 in Figure 10-12) but that, even beyond Hour 17, the post-developed runoff rates only exceed the pre-developed rates by a small percentage and have absolute values considerably less than the pre-developed peak.

In conclusion, it is important to note that all of the studies and research discussed above are highly complex and that their specific findings pertain to particular watersheds and water bodies. Nevertheless, while conducting similar studies of the watersheds and water bodies under the jurisdiction of a new urban runoff management program would be the most accurate way of determining appropriate runoff volume controls for that program, the results discussed above can be (and have been) used effectively to formulate general quantity requirements.

In summary, the above subsection on runoff quantity control design storms presented the following ideas and information:

- Runoff quantity controls are necessary in order to prevent the adverse flooding, erosion, and habitat impacts that can be caused by land development and redevelopment.
- In order to prevent this wide range of impacts, the maximum design storm for runoff quantity control is typically set at the 100-year storm frequency.
- The most effective runoff quantity controls are those that limit both post-developed site runoff volumes and peak rates to levels no greater than pre-developed site conditions.
- Limiting post-developed runoff volumes to pre-developed amounts may be difficult or impossible to achieve at certain development sites due to insufficient soil permeability, soil thickness, and other site factors.
- Where post-developed runoff volumes cannot be maintained at pre-developed amounts, requiring

post-developed peak site runoff rates to be less than pre-developed rates has been shown to be more effective than allowing post-developed peaks to equal pre-developed peaks.

- The required percent reduction below pre-developed peak rates can range from approximately 10 percent to 50 percent for frequent storm events such as the 1- or 2-year storm to 60 percent to 80 percent of the 100-year storm.
- Exact determination of required peak rate reduction factors requires detailed watershed and water body data and analyses.

Runoff Quality Control

As noted at the start of this section on design events, it is necessary for an effective urban runoff management program to address both the quantitative and qualitative impacts of urbanization. That is because the land development and redevelopment activities associated with urbanization can have adverse safety and environmental impacts due to changes in both stormwater runoff quantity and quality. And similar to runoff quantity controls, the establishment of effective runoff quality controls includes the determination of a maximum rainfall or runoff amount that such controls will apply.

In seeking information on appropriate runoff quality design event limits, one encounters problems similar to those described above for runoff quantity design events. A review of the EPA's Stormwater Phase II Final Rule once again does not yield any specific requirements (for reasons explained in the final Rule text). Therefore, a review of the requirements of established urban runoff management programs is the next logical avenue of pursuit. However, a review of these programs shows that the concept of a runoff quality design event has been the source of some confusion and misunderstanding, which has diminished the success of certain programs. As a result, this section will attempt to address these misunderstandings and promote the selection of effective and understandable design event levels.

As with runoff quantity, a runoff quality design event represents the maximum event depth, measured either by rainfall or runoff, that must be met by a runoff management program's stormwater quality requirements. As such, it represents an upper limit on the performance of structural facilities designed and constructed to prevent the adverse runoff quality impacts of land development. This immediately raises three questions regarding the maximum runoff quality design event:

1. Should the maximum design event be based upon rainfall or runoff?
2. What should the maximum design event depth be?
3. Is a total event depth sufficient, or is a temporal distribution also required?

Each of these questions is addressed below.

Rainfall or Runoff

In addressing the first question, some additional and intriguing questions are raised. First, since the goal of the urban runoff management program is to prevent adverse impacts to stormwater runoff, shouldn't the runoff quality design event be based upon the runoff from a development, rather than the rainfall that produces it? At first look, an affirmative answer to this question would appear logical. And in fact, many existing urban runoff management programs have chosen to do that. In simplified form, these programs require that a certain level of quality treatment be provided to a fixed amount of runoff. And while the level of treatment may vary, depending on the type of development or pollutant and/or the proximity and value of a downstream water body or other resource, the amount of runoff to be treated, typically expressed as a depth over the development site's area, usually remains the same.

For example, many municipal, county, and state runoff management programs currently require treatment of the first inch of runoff, averaged over the total site area, from a proposed development site. This requirement is imposed regardless of the type of development. In some programs, this is done to simplify the computations needed to meet the requirement, while in others, it is considered the best or at least a suitable maximum runoff quality design event.

However, the use of runoff in general and a constant or fixed amount in particular as the runoff quality design event can cause significant disparities in the levels of water quality control provided by different types of developments. In addition, these disparities unfortunately result in certain development types normally associated with greater pollutant loadings than others being allowed to provide lower overall levels of quality treatment. The example presented below illustrates this problem.

For example, let's apply the 1 inch runoff requirement to both a single family residential and a commercial development site, each one acre in area. The development characteristics of each site are summarized in Table 10-5. Both sites are assumed to have soils belonging to Hydrologic Group C as defined by the Natural Resource Conservation Service's (NRCS) Technical Release 55 – Urban Hydrology for Small Watersheds (TR-55), which has become something of a standard for computing runoff volumes and rates from land development sites. As shown in Table 10-5, 20 percent of the proposed residential site and 85 percent of the proposed commercial site will be covered with impervious surfaces. The remainder of both sites will be turf grass that will be assumed to be in good hydrologic condition as defined in TR-55. According to the parameters defined by the TR-55 methodology, the impervious surfaces at both sites will have a Runoff Curve Number (CN), which is a measure of a surface's runoff potential, of 98, while the turf grass surfaces will have a CN of 74. (See Table 2-2a in TR-55 for more details of these various CNs.) It should be noted that the proposed impervious coverages for each development type in Table 10-5 are identical to those specified for these land uses in Table 2-2a. As such, they can be considered typical of these two types of development.

Utilizing the runoff volume computation methodology contained in TR-55, which is based upon the NRCS Runoff Equation, for an average runoff depth of 1 inch from both the proposed residential and commercial sites, we can compute the rainfall depths that would be required at each site. These results are summarized in Table 10-6. As shown in the table, it will be necessary for approximately 2.6 inches of rain to fall on the proposed residential site to produce the required 1 inch of runoff that must receive runoff quality treatment. However, at the proposed commercial site, only 1.4 inches of rain will be necessary to produce the same 1 inch runoff treatment volume. When we remember from Chapter 2 that different rainfall depths of equal duration, which we must assume are occurring at our two proposed development sites, are associated with different probabilities or recurrence intervals, we can see that this disparity in required rainfall depth means that the two sites are not provided equal levels of runoff quality treatment.

This disparity can be illustrated by analyzing the recurrence interval of each required rainfall depth. To do so, we will need to locate our proposed sites somewhere in the country, since the recurrence interval of a certain rainfall will depend upon its geographic location. We will also need to assume a duration of the rainfall that produced the 1-inch design event volume (although it is interesting to note that many programs do not specify a duration, an omission that prevents any determination of design event probability). Assuming that both sites are located in central New Jersey and assuming that both rainfalls have a 24-hour duration, we can quickly perform a simplified statistical analysis of 24-hour rainfall records for this part of the state in order to produce estimates of each rainfall's recurrence intervals. The results of our simplified analysis, based upon rainfall data developed by the Hydrometeorological Design Studies Center of the National Weather Service, are shown in Table 10-6 and illustrated in Figure 10-13.

From the above results, it can be seen that by requiring both the proposed residential and commercial sites to treat up to an equal runoff depth of 1 inch, the residential site will provide the required treatment for all storms up to a 2.6 inch rainfall, which for a 24-hour storm duration will have a recurrence interval of approximately 0.8 years or 10 months. However, the same

Table 10-5: Residential and Commercial Site Characteristics, Runoff Quality Design Event Example

Development Type	Area (Acres)	% Impervious Cover	% Pervious Cover	Pervious Cover Type	Impervious Cover CN	Pervious Cover CN
Residential	1.0	20%	80%	Grass	98	74
Commercial	1.0	85%	15%	Grass	98	74

Table 10-6: Required Rainfall Depths and Recurrence Intervals, Runoff Quality Design Event Example

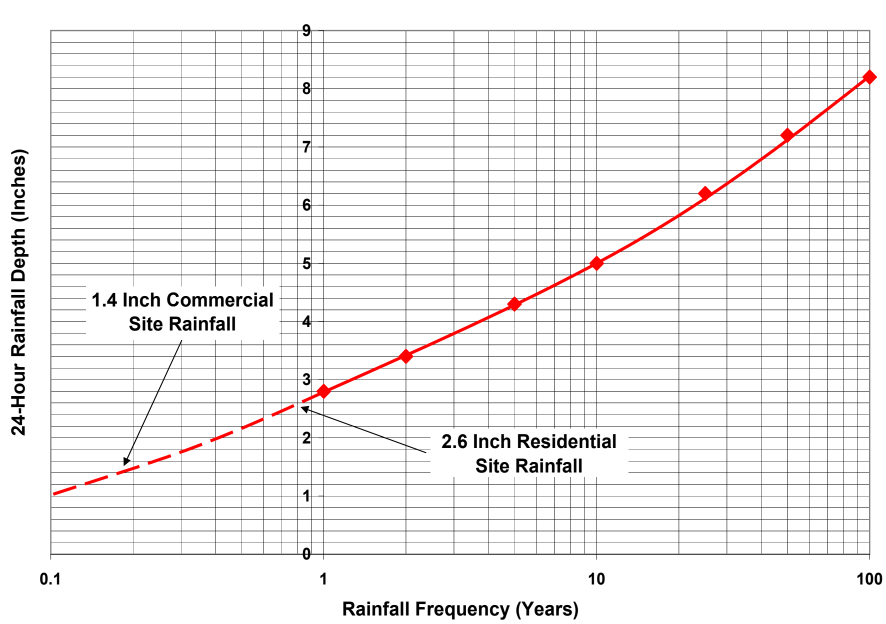
Development Type	Average Site Runoff (Inches)	Required Site Rainfall (Inches)	Rainfall Recurrence Interval (Months)
Residential	1.0	2.6	10
Commercial	1.0	1.4	2

size commercial site will only be required to treat the runoff from all storms up to a 1.4 inch rainfall, which will have a recurrence interval of approximately 0.2 years or 2 months. Stated another way, the proposed commercial site will only be required to provide one fifth the level of runoff quality protection of the residential site, measured on a recurrence interval or probability basis. And when the higher expected pollutant loadings from the commercial site are taken into consideration, this disparity in treatment or protection levels becomes even more detrimental to the goals of the urban runoff management program.

The above illustrates that rainfall and not runoff should be used to specify the maximum water quality design storm. That is because, unlike runoff, a rainfall depth is independent of development type, surface cover, and other site features. And as a series of random events with sufficient records of past occurrences, rainfall can be statistically analyzed to produce depth-probability or depth-recurrence interval relationships. These relationships can be used, in turn, to select the correct rainfall depth for a desired or required level of protection that can be applied uniformly to all developments, regardless of type. However, as shown in the above example, using a fixed runoff depth instead of rainfall allows the probability or recurrence interval of the design event to vary with the development site's rainfall-runoff characteristics. And since each type of development will generally have different characteristics, the resultant runoff depth and, more importantly, level of water quality protection will vary with the development type, and in a direction that allows those levels to decrease with increasing imperviousness.

One additional point should be noted. Many programs that have adopted a runoff depth as the basis of their quality design storm have done so on the assumption that only the impervious surfaces at a proposed development site will produce runoff under such storm conditions. As a result, there will be a relatively constant relationship between rainfall and runoff that will avoid the variable recurrence interval or level of protection problem discussed above. Other programs do use a rainfall depth, but then only compute the resultant runoff from the site's impervious surfaces. However, a review of the runoff computations for both the residential and commercial sites in the above example shows that this will only be true at sites with highly permeable soils with large initial abstractions or surface storage volumes. In the above example, the pervious portions of the residential site that were assumed to have grass cover were the source of more than half the total average 1-inch site runoff depth that required runoff quality treatment. Even the grass areas at the largely impervious commercial site in the example contributed approximately to the total 1 inch of runoff from the site. Therefore, those programs that

Figure 10-13: Estimated Design Event Rainfall Depths and Recurrence Intervals



do not include pervious surface runoff must consider whether such runoff can be safely ignored in the design of structural runoff quality treatment facilities.

Maximum Event Depth

Regarding the question of maximum design storm depth, one must consider the overall level of runoff quality protection that is being sought by the urban runoff management program. Since nonpoint source pollutant loadings and associated runoff quality impacts are discussed or otherwise addressed on an annual basis in many research studies and most stormwater regulations, including the EPA's Stormwater Phase II Final Rule, most existing urban runoff management programs use the same one-year period as the time basis for their runoff quality requirements. For example, many programs require specified levels of pollutant removal or treatment measured on an average annual basis. Therefore, if the time basis of the runoff quality requirement is one year, the largest likely storm that may occur in that one-year period would be the logical choice for a maximum design storm. As a result, many runoff management programs either use or formerly used a maximum annual rainfall as the maximum runoff quality design storm depth.

However, it should be noted that for most, if not all, structural stormwater facilities, some degree of runoff quality treatment is provided even for rain events that exceed the maximum design storm. This will occur primarily during those storm periods when the actual runoff volume and/or rate is less than the maximum design level, although some degree of treatment (albeit at a reduced rate) may occur during periods of overflow when the actual volume and/or rate exceeds the maximum design level. This is true for both online and offline structural facilities. As a result, the use of a maximum design storm depth less than the maximum annual storm appears justifiable.

Presently, the use of a rainfall depth that, in combination with all smaller storms, will on average produce 90 percent of the average annual rainfall (or impervious surface runoff) at a land development site appears to be a standard technique for selecting a maximum runoff quality control design storm depth in many current urban runoff management programs. Other programs use a variation of this technique by specifying some percentage of the 1- or 2-year rainfall as the maximum design storm depth, again based upon the finding that the resultant rainfall will represent approximately 80 to 90 percent of annual average rainfall or impervious surface runoff at the development site. Determination of the required depth typically requires a suitably long rainfall record with adequate areal coverage that can then be statistically analyzed to determine average annual rainfall patterns and depths. Depending upon the variation in average annual rainfall throughout a program's jurisdiction, one or more design depths may need to be specified. Examples of various maximum runoff quality design storm depths in current state stormwater management programs are summarized in Table 10-7.

Design Storm Distribution

The decision of whether a maximum design storm depth alone is sufficient or whether a temporal distribution of that design depth is also needed will depend in large measure on the types of structural facilities that might be used to meet the urban runoff management programs' runoff quality control requirements. If all of the potential structural facilities will provide the required level of runoff quality control primarily through some form of storage and slow, regulated release that results in the peak outflow rate being a small percentage of the peak inflow rate, then a maximum design storm depth may be sufficient. This is due to the fact that in such facilities, the total volume of inflow is the domi-

Table 10-7: Maximum Runoff Quality Design Storm Depths Examples

State Program	Maximum Quality Storm Depth
Connecticut	1.00 Inches
New Jersey	1.25 Inches in 2 Hours
New York	0.8 to 1.3 Inches Depending Upon Location
Maryland	0.9 to 1.0 Inches Depending Upon Location
Washington	72% of 2-Year, 24-Hour Storm

nant design factor, and the various rates at which the inflow may occur have little influence on overall facility size or details. Typical examples of such volume-based structural facilities include infiltration basins, retention basins (also known as wet ponds), wetlands, and even extended detention basins where the peak outflow is a small percentage of peak inflow.

However, if the potential structural facility types include those that do not rely to any great extent on runoff storage but instead treat runoff at flow rates essentially unchanged from their inflow rates, then it will be necessary to specify both a maximum design storm depth and the manner in which that depth will occur over time. This is because the dominant design factor for such facilities is the maximum inflow rate, not the total inflow volume, and it will therefore be necessary to determine that maximum runoff rate. This will require knowledge of the total duration of the design storm and the manner in which the rain fell during that time period. A design storm distribution will also be required for volume-based facilities if the peak outflow rate is sensitive not only to total inflow volume but also to inflow rates. This can generally be assessed by comparing the peak inflow and resultant outflow rates. If the peak outflow rate is greater than approximately 10 percent of the peak inflow rate, then such inflow rate sensitivity probably exists, or at the least needs to be checked, and a design storm distribution will also be required. Typical examples of peak rate-based structural facilities include swales, filter strips, and hydro-mechanical devices including the growing number of manufactured runoff treatment devices.

Having determined the need for a design storm distribution to accompany a program's maximum design storm depth, the logical next question is: what type of distribution? And the answer can be found in the same discussion of recurrence interval or level of runoff quality protection that was presented in the section above on design storm depths. Presumably, a runoff management program's design storm depth, which, when converted to runoff over a drainage area, will determine the total runoff volume of the design storm, will be based upon some measure of probability or recurrence interval, such as a 1- or 2-year, 24-hour storm or a 90 percent rainfall depth. As such, this probability or recurrence interval will establish the level of runoff quality protection or control that will be provided by volume-based structural stormwater management facilities designed to treat the associated runoff volume. However, as noted above, there are certain types of structural facilities such as swales, filter strips, and especially hydro-dynamic devices that do not provide volume-based treatment and whose designs are not based upon a total runoff volume but upon a peak design storm runoff rate. To ensure that these peak rate-based facilities provide the same level of runoff quality control or treatment as volume-based ones, it will be necessary to use a design storm distribution that produces peak runoff rates that have the same probability or recurrence interval as the design storm's total rainfall depth.

To do so, this design storm distribution must have certain important characteristics. First, since it will be used to design stormwater management facilities for a range of development sites and Times of Concentration (see Chapter 2), the design storm distribution must be able to produce peak runoff rates with the same recurrence interval for all of them. As a result, the design storm distribution must consist of varying rainfall rates throughout its duration, with the maximum rainfall intensities for each time period up to the total design storm duration all having the same probability or recurrence interval; namely, the recurrence interval of the storm's total rainfall depth. For an urban runoff management program that utilizes a 24-hour design storm depth, an appropriate design storm distribution could be one of the dimensionless rainfall distributions developed by the Natural Resources Conservation Service (NRCS) for Technical Release 55 – Urban Hydrology for Small Watersheds (TR-55). Each of these distributions meets the maximum rainfall intensity criteria described above; namely, each contains a range of intensities and durations with the same probability or recurrence interval and, as such, can be used to produce the peak runoff rates for a range of Times of Concentrations.

However, it is important to emphasize at this point that the use of any of the NRCS design storm distributions in TR-55 requires the design storm depth to have been based upon 24-hour rainfall data and be assumed to fall over a 24-hour period. Using any of the TR-55 distributions to compute a peak runoff rate or entire hydrograph for a rainfall with a total duration other than 24 hours will result in incorrect runoff rates and inconsistent recurrence intervals between the total design storm runoff volume and peak runoff rate. As described above, this will lead to inconsistent levels of protection between volume-based and peak rate-based structural facilities. This also applies to the use of the NRCS design storm distributions for a design event runoff volume of unspecified duration.

For these reasons, an urban runoff management program that uses a design storm duration other than 24 hours must also develop an appropriate design storm distribution. For example, New Jersey's Stormwater Management Rules specify the use of a 1.25-inch, 2-hour runoff quality design storm. Based upon a statistical analysis of New Jersey rainfall data, such a rainfall has a recurrence interval of approximately 0.8 years or 10 months. Therefore, to insure that peak rate-based runoff treatment facilities will provide the same level of protection as volume-based facilities, it was necessary to vary the intensity of the 1.25 inches of rain over the total 2-hour duration in such a way that the resultant peak runoff rate for any Time of Concentration up to 2 hours would have the same 10 month recurrence interval.

The resultant New Jersey stormwater quality design storm distribution is shown in Figure 10-14 and Table 10-8. Figure 10-14 depicts a nonlinear rainfall distribution with the maximum intensity (indicated by the slope of the line) occurring in the middle of the 2-hour storm duration. This general shape is similar to several other design storm distributions, including the various TR-55 design storms, with the centrally located maximum rainfall intensity and overall symmetric shape considered to have average runoff potential. In addition, a review of Table 10-8 shows how, for a range of rainfall periods up to the total 2-hour duration, the maximum rainfall intensities occurring in those periods have the same 10-month recurrence interval as the overall 1.25 inch, 2-hour design storm. These various intensities were determined from the same statistical analysis of New Jersey rainfall data used to determine the recurrence

Figure 10-14: New Jersey 1.25-Inch/2-Hour Stormwater Quality Design Storm Distribution

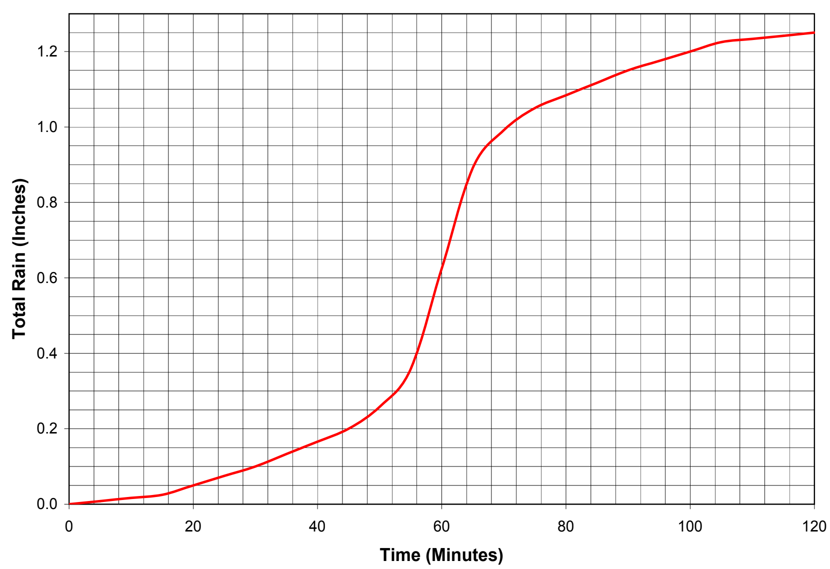


Table 10-8: New Jersey 1.25-Inch/2-Hour Stormwater Quality Design Storm, Maximum Rainfalls and Probabilities for Various Time Periods

Time Period (Minutes)	Maximum Rainfall (Inches)	Maximum Rainfall Intensity (Inches/Hour)	Average Recurrence Interval
10	0.53	3.2	10 Months
20	0.73	2.2	10 Months
30	0.85	1.7	10 Months
60	1.05	1.05	10 Months
90	1.20	0.80	10 Months
120	1.25	0.625	10 Months

interval of the overall 1.25 inch, 2-hour design storm. As a result of these specific rainfall intensities and durations within the overall 2-hour duration, both volume- and peak rate-based structural stormwater management facilities design in accordance with this design storm will provide the same level of runoff quality protection and control.

In summary, the above subsection on runoff quality control design storms presented the following ideas and information:

- Runoff quality controls are necessary in order to prevent the adverse impacts on water quality, aquatic organisms, and habitat that can be caused by land development and redevelopment.
- In order to prevent these impacts, the maximum design event for runoff quality control is typically based upon a percentage of the maximum annual rainfall or runoff.
- The selection of a runoff quality design event depth should be based upon a standard rainfall depth and duration with an appropriate probability or average recurrence interval. This allows a consistent level of runoff quality treatment or control to be achieved for all land development and redevelopment sites.
- Use of a standard runoff depth instead of rainfall causes inconsistencies in the level of runoff quality treatment or control between different development types, with the sites with the greatest amount of impervious cover providing the lowest level of treatment.
- In order to achieve consistent levels of runoff quality treatment or control between volume-based structural facilities such as basins and wetlands and peak rate-based facilities such as swales and hydro-dynamic devices, it is necessary to also specify a temporal distribution of the design storm rainfall.
- This distribution must be capable of producing peak design storm runoff rates that are identical to the probability or average recurrence interval of the total design storm depth. As such, a nonlinear distribution based upon site- or area-specific rainfall intensity-duration-frequency data is required.

Long-Term Rainfall-Runoff Simulation

To complete our discussion of design events, it is important to revisit a topic discussed in detail in Chapter 2; namely, the use of continuous rainfall-runoff simulation based upon long-term rainfall records to determine suitable levels of runoff quantity and, in particular, quality control at a land development site. Depending upon the extent and detail of the available rainfall data and the suitability of the simulation techniques for the project in question, the use of such analytic tools and data can produce results that are superior to the single, maximum design approach described above. This is true for several reasons, most notably the increased accuracy or certainty of results achieved by analyzing the actual rainfall events that occurred at a land development site over an extended time period over the use of a single design storm that is presumed to represent that long-term rainfall. Other advantages include the ability of continuous simulation to include the variability of rainfall depths and occurrences and the interaction between sequential events into the analysis. Since the required rainfall record length should be several multiples of the desired design storm level, continuous simulation is currently more appropriate for runoff quality than quantity control due to the typically lower treatment or control levels required for quality control. Nevertheless, depending upon the rainfall record length and desired level of protection, runoff quantity control analysis and design based upon long-term simulation can also provide superior results to a single design storm approach.

As discussed in Chapter 2, factors that may complicate or even prevent the use of long-term simulation typically include the lack of adequate rainfall data, either in overall length, time increment, or proximity to the development site. Other factors include lack of adequate calibration and verification data, increased analysis time and costs, and the lack of an appropriate simulation model. Despite these complications, an urban runoff management program should always include provisions that allow such an approach to be utilized, particularly to address runoff quality impacts. Programs should also devote some portion of their overall runoff management efforts to developing such analytic tools for eventual future use. For example, the Washington Department of Ecology has sponsored the development of the Western Washington Hydrology Model (WWHM), a continuous runoff simulation model based upon the computer program HSPF (Hydrological Simulation Program – Fortran). According to the department, its plans for the WWHM include improving the model so that it can eventually provide users with the appropriate runoff quality flow rate for land development sites in the western portion of the state, rather than relying on the design storm approach currently utilized by the department. As rainfall data and continuous simulation models become more available, use of this approach is expected to increase.

In summary, the above section on design storms presented the following ideas and information:

- Effective urban runoff management requires control of both the runoff quantity and quality impacts of land development and redevelopment.
- Levels of quality and quantity control can be established through the selection of appropriate design events.
- In order to ensure uniform levels of control, design events should be based upon rainfall rather than runoff amounts and should also have a specified duration that will allow recurrence interval determination.
- A design storm will also require a rainfall distribution over its duration if both volume-based and peak rate-based structural stormwater management facilities are included in the urban runoff management program.
- Such distributions must produce peak runoff rates with the same probability or recurrence interval as the total design storm depth.
- Where available and feasible, continuous rainfall-runoff simulation using long-term rainfall records can produce superior results to a single design storm approach.
- Urban runoff management programs should both monitor and promote the development of continuous simulation techniques and data.

Treatment Levels

In developing a list of suitable structural stormwater management facilities to address the runoff quality requirements of an urban runoff management program, two questions are immediately raised:

1. What runoff pollutants should be treated?
2. What treatment levels should be provided?

These questions are both complex and inter-related. Their answers depend primarily on the conditions within the geographical boundaries of an urban runoff management program. These conditions include, among many others, present runoff quality and quantity levels; the presence, extent, and severity of any existing water body impairments; existing and future land uses, development levels, and water body uses; and related needs such as water supply, sewage treatment, and recreation. It can be seen that such issues pertain to the entire scope of an urban runoff management program and not just to its structural facility component.

A review of the EPA's Stormwater Phase II Final Rule illustrates both the area-dependent and complex nature of these two issues. As discussed above, the final Phase II Rule does not contain specific requirements for either the types of runoff pollutants that must be addressed or the levels of treatment that must be provided by the various owners of small municipal separate storm sewer systems (MS4s). According to the final Rule, this is intentional, for it allows the individual MS4 owner to evaluate and select the pollutants and treatment levels necessary to both achieve the program's goals and comply with NPDES requirements. In addition, the lack of specificity in the final Rule also affords the EPA, designated NPDES permitting authorities, and Phase II permittees the time and opportunity to further investigate stormwater runoff processes, pollutants, and impacts and to introduce more specific requirements into the Rule as problems require and future solutions allow. This process will be supplemented by the ongoing process of determining total maximum daily loads (TMDLs) of specific pollutants for impaired water bodies.

Nevertheless, before a comprehensive list of effective structural stormwater management facilities can be compiled for an urban runoff management program, or a specific facility selected for a land development proposal, both questions must be addressed. This is due to the fact that not all structural stormwater management facilities can effectively treat the same runoff pollutants or provide similar levels of treatment. Therefore, the program's runoff treatment and control goals must be understood before structural facility selection can begin. Information regarding the identification of pollutants of concern and the establishment of required treatment or control levels can be found in Chapters 3 through 6.

The remainder of this section will focus on a third question that logically follows from the first two discussed above; namely, once key runoff pollutants and required treatment or removal rates have been identified, how does an urban runoff management program go about specifying what structural facilities can be used to meet these treatment requirements? Without such information, the program's runoff treatment goals can obviously only be stated, but never met with any certainty.

However, analysis of this question raises more complex ones, including what runoff pollutants can each type of structural facility effectively treat and, more importantly, what specific level of treatment can each type provide. Specific answers to these questions, the second in particular, depend upon several highly variable factors, including the concentration and total load of the pollutant, the volume and various rates of the runoff that transports it, antecedent rainfall and runoff conditions, and even the season or time of year. The variability of both applicable pollutants and levels of treatment can be seen by reviewing the sampling results of actual structural facilities taken over a number of storm events. Depending upon the pollutant, the reduction in pollutant load or mean concentration achieved by the structural facility can vary considerably from event to event, with even negative reductions achieved at times, particularly for nutrients. Such variability makes it extremely difficult to determine a structural facility's exact pollutant removal rate and illustrates why pollutant removal criteria are typically based upon average annual conditions.

In light of these questions and complexities, a review of current urban runoff management programs indicates that there are two general approaches to the task of specifying an appropriate set of structural stormwater management facilities to meet a program's runoff quality goals. A discussion of each approach is provided below. Solely for the purposes of these discussions, the two approaches have been assigned the following names:

1. Specified Facility Approach
2. Specified Treatment Approach

The discussions presented below are intended to illustrate the distinguishing features of each approach and identify some of the advantages and disadvantages of each. Nothing presented in these discussions should be considered to favor one approach over the other.

Specified Facility Approach

Following the selection of the program's pollutants of concern and their required level of treatment or control, a list of structural stormwater management facilities considered capable of providing such treatment must be selected. The selection process is typically based upon both model studies and field sampling of each structure type over a range of conditions. Typically, such activities were previously conducted by others with the results taken from a literature search or from one of the growing number of pollutant removal performance databases. At times, this data is supplemented by research conducted or sponsored by the program itself. From this information, a list of structural facilities considered capable of providing the program's required level of runoff quality treatment can be developed using the specified facility approach.

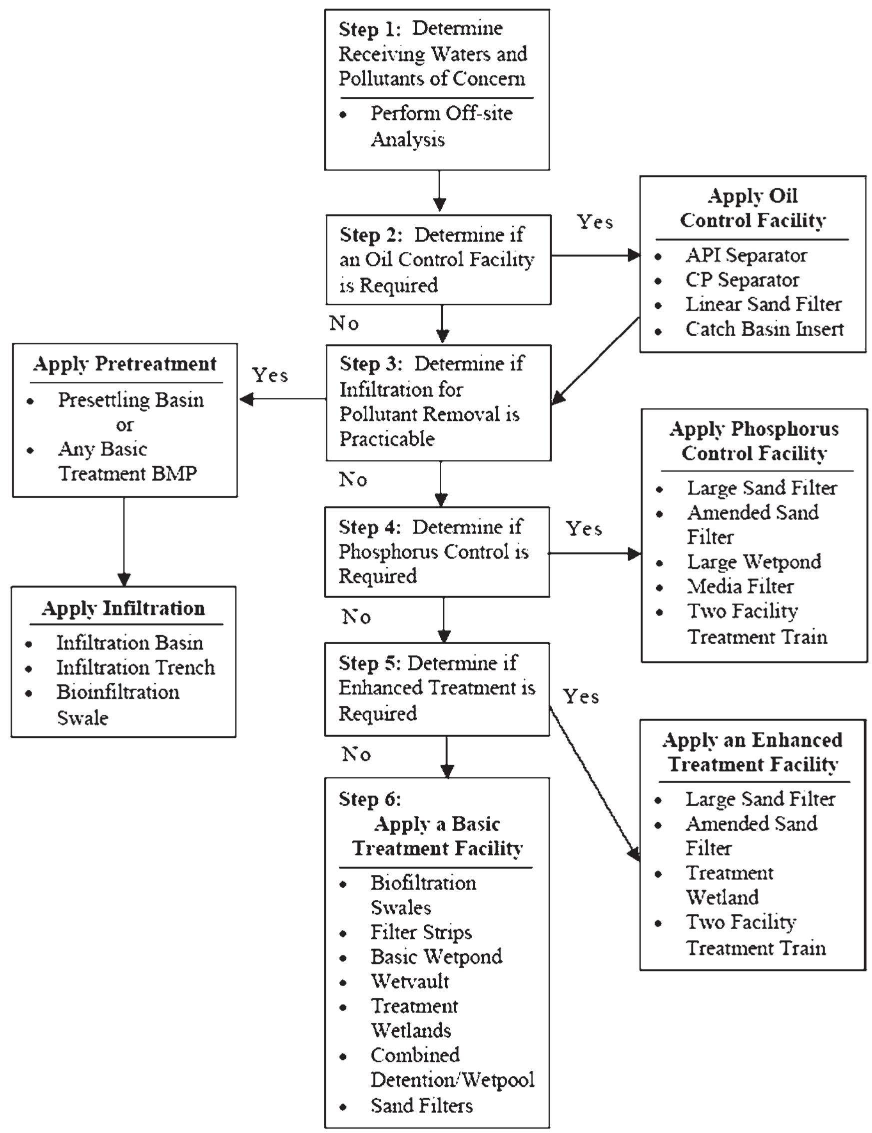
In the specified facility approach, exact pollutant removal performance values for the structural facilities on the program list do not have to be determined. Instead, the list identifies those structural facilities considered capable of meeting the program's treatment requirements. As a result, the specific question "Exactly how much pollutant reduction can a facility achieve?" is replaced by the relative and, therefore, more easily answered question "Can the facility achieve enough?" In this approach, it is also not even necessary to assign a numerical value to the required level of treatment. The program only needs to match up a required level of treatment with a list of capable or acceptable structural facilities.

An example of the specified facility approach can be found in the Stormwater Management Manual for Western Washington, developed by the Washington State Department of Ecology. In Chapter 4 of Volume I of this manual, the department defines four levels of required treatment in the western portion of the state, depending upon a range of factors, including the type of proposed land development, the intensity of traffic or other site uses, and the presence of impacted water bodies downstream. However, quantitative values for these four levels of treatment, expressed either as pollutant removal rates or other standard performance measures, are not specified. Next, the manual presents a list of structural facilities that are considered capable of providing each level of required treatment. Subsequent volumes of the manual then provide details on the hydrologic, hydraulic, and structural design of the various facilities.

The overall selection process is illustrated in the chart shown in Figure 10-15. In the figure, which is based upon Figure 4.1 in the Western Washington Manual, the various structural facilities considered acceptable for each required level of runoff treatment can be seen in various boxes around the chart's perimeter following the designation of each required treatment level. What should be noted is the lack of any specific treatment levels, either in the form of required program levels or actual facility performance values.

A variation of the specified facility approach can be found in the Maryland Stormwater Design Manual published by the state's Department of the Environment. In Chapter 1 of that manual, the required levels of post-development runoff treatment are stated. They include 80 percent removal of post-development total suspended solids (TSS) load and 40 percent of post-development total phosphorus load. In Chapter 2, the manual provides a detailed listing of acceptable

Figure 10-15: Structural Stormwater Management Facility Selection Chart from the Stormwater Management Manual for Western Washington



structural facilities considered capable of meeting these required treatment levels. Detailed sizing and structural criteria for these facilities are provided in subsequent chapters and appendices. However, similar to the Western Washington Manual, the Maryland Manual does not provide specific pollutant removal performance values for the selected facilities. Instead, it states in Chapter 1 that any of the structural facilities can be assumed to meet the program's required TSS and phosphorus removal levels if it is sized to capture the required runoff quality control volume, designed in accordance with the specific structural criteria contained in the manual, and both constructed and maintained properly.

From the above examples, it can be seen that the specified facility approach is in keeping with the minimum requirements for post-construction stormwater management contained in the EPA's Stormwater Phase II Final Rule. As stated in the final Rule, owners of small municipal separate storm sewer systems (MS4s) are required to "develop and implement strategies which include a combination of structural and/or nonstructural best management practices (BMPs) appropriate for the community." Such BMPs should also "minimize water quality impacts and attempt to maintain pre-development runoff conditions." As in the two examples above, no structural facility performance values are specified, only the need to achieve suitable protection levels with acceptable structural facilities.

As noted above, the advantage of the specified facility approach is the avoidance of any need to provide specific performance data for structural stormwater management facilities. In light of the uncertainties regarding such data and the difficulties in selecting (and possibly defending) representative values, this advantage can be significant. However, there are certain disadvantages that urban runoff management program developers should be aware of. These include the need to identify all acceptable structural facilities for each level of runoff quality control. In the case of the Western Washington Manual, this required the development of four sets of acceptable structures, one for each of the state's required treatment levels. As the number of required treatment levels increases, this can become cumbersome to administer and use, particularly when different portions of a proposed development site fall under different treatment requirements. It should be noted that this problem is addressed very effectively in the Western Washington Manual by the structural facility selection chart shown in Figure 10-15.

A second disadvantage of the specified facility approach is that is promotes, to a certain degree, the site design philosophy that a single structural facility can be used to meet all of the program's runoff treatment requirements. In providing a list of equally acceptable facilities, it is left to the site designer to simply select a single one best suited to site conditions in order to meet program requirements. Providing such a list can, albeit unintentionally, limit site design creativity and lead to the repetitive use of one or a few structural facilities that can earn program approval. It should be noted that both the Western Washington and Maryland Manuals address this issue by continually promoting site design creativity and the incorporation of structural facilities and nonstructural measures. For example, the introduction to the Maryland Manual states:

> *It is hoped that the design standards and environmental incentives provided below will produce better methods and advance the science of managing stormwater by relying less on single BMPs for all development projects and more on mimicking existing hydrology through total site design policies.*

A final disadvantage may occur in programs with multiple pollutant removal requirements. Since not all structural facilities are equally effective at treating certain pollutants, it is not uncommon to provide a series of facilities in a treatment train approach in order to meet the removal requirements for all of the required pollutants. In such cases, one facility may be selected to provide the required levels of treatment for pollutant A, followed by a second facility for pollutant B. However, since there are no specific performance values for the selected facilities for each pollutant and, therefore, no way to incorporate these values into the facility designs, each must be designed to meet all of the requirements specified for that facility by the program. This can result in overdesign, which may not be detrimental from a runoff protection standpoint but from cost, land disturbance, safety, and/or maintenance standpoints. Programs may respond to this problem by preparing alternative design standards for structural facilities used in series or treatment trains. However, this can increase the effort needed to create, administer, and design under the program.

Specified Treatment Approach

The specified treatment approach is the second method of determining acceptable structural stormwater management facilities for an urban runoff management program. Under this approach, both required treatment levels for each pollutant of concern and a list of structural facilities are developed and specified by the program. However, in contrast to the specified facility approach, the list of structural facilities includes specific pollutant removal rates or other performance measures for each facility type. This allows a site designer to select a single or combination of facilities that meet the program's treatment levels based upon their respective performance values.

An example of the specified treatment approach can be found in the New Jersey Stormwater Best Management Practices Manual published by the state's Department of Environmental Protection (NJDEP). In Chapter 4 of the manual, the NJDEP establishes a required TSS removal rate of 80 percent for land development and redevelopment projects, as well as the removal of nutrients to the maximum extent feasible. The chapter also provides a list of approved structural facilities that includes adopted TSS, total phosphorus, and total nitrogen removal rates for each. These removal rates can then be used to select appropriate facilities to meet the required treatment levels. Subsequent chapters in the manual provide design details for each facility type.

The various structural facilities and adopted removal rates are summarized in Table 10-9. It should be noted that the range of TSS removal rates shown for extended detention basins, vegetative filters, and wet ponds reflects varying facility designs. For example, for extended detention basins, the 40 percent to 60 percent TSS removal range pertains to extended detention times varying from 12 to 24 hours. For wet ponds, the 50 percent to 90 percent TSS removal range is based upon both the relative size of the pond's permanent pool and the use of extended detention above the permanent pool level. The 60 percent to 80 percent range of TSS removal rates for vegetative filters is based upon the type of vegetation used in the filter, which can range from turf grass to indigenous woods.

The above shows that one of the advantages of the specified treatment approach is that in providing designers and reviewers with quantitative measures of facility effectiveness, it also provides a demonstrable method of using structural stormwater management facilities either individually or in series to meet the program's pollutant treatment requirements. This allows designers a greater degree of freedom in selecting and locating structural facilities on a land development site. It also allows certain structural facilities such as extended detention basins or turf grass filters, that would not be sufficient by themselves, to be included in a site design. Under the specified facility approach, such facilities, which are relatively easy and inexpensive to design, construct, and maintain, may not be allowed. In addition, by providing a range of removal rates, the specified treatment approach allows designers to adjust a facilities design to optimize its size and features to meet the program's required treatment rates.

As stated above, the primary disadvantage of the specified treatment approach is the required development of appropriate removal rates for the program's pollutants of concern. This requires a considerable amount

Table 10-9: Approved Structural Facilities and Adopted Pollutant Removal Rates, New Jersey Stormwater Best Management Practices Manual

Structural Facility Type	Adopted TSS Removal Rate (%)	Adopted Total Phosphorous Removal Rate (%)	Adopted Total Nitrogen Removal Rate (%)
Bioretention Basin	90	60	30
Constructed Wetland	90	50	30
Extended Detention Basin	40 – 60	20	20
Infiltration Basin	80	60	50
Manufactured Treatment Device	Subject to NJDEP Verification		
Pervious Paving	80	60	50
Sand Filter	80	50	35
Vegetative Filter	60 – 80	30	30
Wet Pond	50 – 90	50	30

of research and review in order to develop appropriate removal rates that are accurate not only for a specific facility type but also relative to other types. This must be done for each pollutant of concern in the program. Once this process is completed, the existence of specific, quantitative removal rates may inadvertently imply an accuracy that does not exist. The New Jersey Manual addresses this concern with the following text:

> *It is important to note that the TSS removal rates shown in [Chapter 4] have been based upon several sources of BMP research and monitoring data as well as consultation with numerous stormwater management experts. As demonstrated by that research, actual TSS removals at specific BMPs during specific storm events will depend upon a number of site factors and can be highly variable. As such, the TSS removal rates presented in Table 4-1 are considered representative values that recognize this variability and the state's need to develop and implement a statewide stormwater management program.*

Another disadvantage to the specified treatment approach is the need to address structural facilities in series or a treatment train. While this was also required with the specified facility approach, the specified treatment approach requires a methodology by which the total pollutant removal rate of the structure series can be determined. In the New Jersey Manual, a simplified equation is presented that allows the determination of the total pollutant removal rate of two separate structural facilities operating in series. The equation is presented below:

$$R = A + B - [(A \times B)/100]$$

where:

R = Total Pollutant Removal Rate

A = Pollutant Removal Rate of the Upstream BMP

B = Pollutant Removal Rate of the Downstream BMP

The equation assumes that the removal rates shown in Table 10-9 for a specific facility remain the same regardless of the facility's location in the series. Recognizing the limitations of this assumption, the manual also provides the following guidelines for arranging the various facilities in the most effective order:

1. Arrange the BMPs from upstream to downstream in ascending order of TSS removal rate. In this arrangement, the BMP with the lowest TSS removal rate would be located at the upstream end of the treatment train. Downstream BMPs should have progressively higher TSS removal rates.
2. Arrange the BMPs from upstream to downstream in ascending order of nutrient removal rate. Similar to 1 above, the BMP with the lowest nutrient removal rate would be located at the upstream end of the treatment train in this arrangement. Downstream BMPs should have progressively higher nutrient removal rates.
3. Arrange the BMPs from upstream to downstream by their relative ease of sediment and debris removal. In this arrangement, the BMP from which it is easiest to remove collected sediment and debris would be located at the upstream end of the treatment train. In downstream BMPs, it should be progressively more difficult to remove sediment and debris.
4. These guidelines should generally be applied in the order presented above. As such, a series of BMPs would be preliminarily arranged in accordance with their relative TSS removal rates (Guideline 1). This preliminary arrangement would then be refined by the BMPs' relative nutrient removal rate (Guideline 2) and then their ease of sediment and debris removal (Guideline 3). Two or more iterations may be necessary to select the optimum arrangement, which should also include consideration for site conditions and the abilities and equipment of the party responsible for the BMPs' maintenance.

The Future

In addition to the two general approaches discussed above, a third approach to designating acceptable structural stormwater management facilities for runoff quality control holds great promise for the future. Under this approach, loadings of a program's pollutants of concern would be estimated from a continuous rainfall-runoff simulation based upon long-term records for actual rain events. These loadings would then be introduced to one or a series of structural facilities that would achieve load reductions on an event-by-event basis based upon both the event and facility characteristics. Upon completion, a comparison could be made over a selected time period between pre- and post-development pollutant loadings.

Utilizing this approach, an urban runoff management program could require specific pollutant treatment or removal levels based upon site location, development type, and/or type and condition of the downstream water body. Such treatment levels could range, for example, from no increase in existing pollutant loadings for low-intensity developments or unimpaired downstream water bodies to a specific decrease in existing loadings for high-intensity developments or impaired downstream water bodies. Reductions in existing loadings may also be appropriate for redevelopment projects in highly urbanized areas with highly impaired water bodies in order to restore such water bodies and address environmental justice issues in such areas.

While the data, models, and other analytical tools are already available and being used for specific development proposals, the application of this approach to a program-wide basis remains in the future. As noted above, the development to date of the Western Washington Hydrology Model (WWHM) by the Washington Department of Ecology represents an initial step toward such an approach. Further movement is expected with the continued development and application of TMDLs for impaired water bodies. A comparison of the two structural facility designation approaches discussed earlier in this section indicates that the specified treatment approach appears initially somewhat better suited to advance toward this more detailed program type.

In summary, the above section on pollutant treatment levels and structural facility performance presented the following ideas and information:

- After identifying pollutants of concern and determining appropriate treatment or removal levels, a list of acceptable structural stormwater management facilities capable of achieving these treatment levels needs to be specified.
- At present, there are two general approaches used to specify such a list.
- The specified facility approach is based upon specifying a list of acceptable structural facilities that will meet the program's runoff quality requirements without specifying facility pollutant treatment performance values.
- The specified treatment approach also specifies a list of acceptable structural facilities along with specific pollutant treatment performance values for each.
- Both approaches have advantages and disadvantages. In general, the specified facility approach requires less initial development. However, the specified treatment approach offers more options in the selection of structural facilities and appears more suited for future development.
- Future urban runoff management programs are expected to be based upon long-term, continuous simulation of runoff and both pollutant generation and treatment.

Structural Facility Selection and Design Criteria

Similar to the 1994 *Fundamentals of Urban Runoff Management*, this section of the structural stormwater management facilities chapter will present information on the selection, siting, and design of structural facilities. However, in the period between the 1994 edition and the present book, the amount of detailed, reliable, and readily available information on these topics has grown at what seems like an exponential rate. The 1994 *Fundamentals* book contained more than 35 pages of structural facility selection criteria, performance data, and design details for seven types of structural facilities, consisting of wet ponds, extended detention basins, oil separators, wetlands, infiltration practices, swales, and sand and leaf compost filters. Much of the information contained in those pages had previously received limited distribution or had been unpublished. However, at the time of this chapter's writing, a single internet search of "structural stormwater management facilities" yields 224,000 sites, while "best management practices manual" yields an additional 6,690,000.

In light of this almost incomprehensible wealth of structural stormwater management facility information, this chapter will take a different approach to providing structural facility selection, siting, and design information. Instead of adding to this vast body of information (and raising the number of structural facility sites to 224,001), it recommends in Table 10-10 those information sources, available through the internet, that it considers exceptional and worthy of note by urban runoff management program developers and administrators. It should be noted that there are many other outstanding sources not mentioned below and that the order of those that are should not be taken as an indication of relative quality or value.

Table 10-10: Recommended Structural Stormwater Management Facility Selection, Siting, and Design Information Sources	
Source:	The Stormwater Manager's Resource Center
By:	The Center for Watershed Protection
Site:	http://www.stormwatercenter.net/SMRC_home_test.htm)
Comments:	Outstanding internet site with slideshows, fact sheets, and even a stormwater manual builder.
Source:	NPDES Stormwater Home Page
By:	U.S. Environmental Protection Agency
Site:	http://cfpub.epa.gov/npdes/home.cfm?program_id=6
Comments:	Nothing like going to the Source for information. And there's plenty, including the Stormwater Phase II Final Rule, fact sheets, and outreach materials.
Source:	Guide for Best Management Practice Selection in Urban Developed Areas
By:	American Society of Civil Engineers
Site:	https://www.asce.org/bookstore/book.cfm?book=4058
Comments:	Concise guide that covers all of the pertinent facility selection criteria.
Source:	Stormwater Management Manual for Western Washington
By:	Washington Department of Ecology
Site:	http://www.ecy.wa.gov/programs/wq/stormwater/manual.html
Comments:	One of the most comprehensive stormwater management manuals available. Includes extensive structural facility design criteria and explanations.
Source:	Maryland Stormwater Design Manual
By:	Maryland Department of the Environment
Site:	http://www.mde.state.md.us/Programs/WaterPrograms/SedimentandStormwater/stormwater_design/index.asp
Comments:	Outstanding stormwater management manual with numerous structural facility design examples and sample computations.
Source:	New Jersey Stormwater Best Management Practices Manual
By:	New Jersey Department of Environmental Protection
Site:	http://www.njstormwater.org/bmp_manual2.htm
Comment:	Comprehensive stormwater manual that includes extensive structural facility design, construction, and maintenance criteria, including adopted pollutant removal performance rates.
Source:	Wisconsin Stormwater Manual
By:	Wisconsin Department of Natural Resources
Site:	http://www.dnr.state.wi.us/org/water/wm/nps/stormwater/publications.htm#uwex
Comment:	Extensively researched stormwater management manual with concise presentation of structural facility design criteria.
Source:	Stormwater Treatment Devices: Design Guideline Manual
By:	Auckland Regional Council
Site:	http://www.arc.govt.nz/arc/environment/water/stormwater-tp10.cfm
Comment:	Includes well-researched chapters on numerous structural stormwater management facilities. Proves that both runoff and runoff management expertise know no borders.

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U.S. Environmental Protection Agency, "National Pollution Discharge Elimination System – Regulations for the Revision of the Water Pollution Control Program Addressing Storm Water Discharges," Federal Register Volume 64 Number 235, December 8, 1999.

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Wisconsin Department of Natural Resources, "Wisconsin Stormwater Manual," 2000.

CHAPTER 11

Low Streamflow Augmentation and Groundwater Recharge

CHAPTER 11

Low Streamflow Augmentation and Groundwater Recharge

This is a difficult chapter to write. There is considerable discussion about issues related to low streamflow and the impacts of imperviousness on streamflow, but reality is a bit more elusive. Some researchers have noted that urbanization decreases low flows, others have found that baseflows have increased as urbanization has occurred, and some studies have not been able to support a conclusion either way.

As a result, this chapter will attempt to discuss the issue objectively and make recommendations where appropriate. Clearly, it is a difficult issue to quantify, as very few studies have been conducted over a time scale that is long enough to assess trends or establish very basic issues such as flow change resulting from increased impervious surfaces. Several case studies will be presented that argue both sides from a trend perception.

Stormwater professionals are increasingly aware both of the impacts that stormwater volumes have on sizing of stormwater practices and the possible impacts of additional volumes on receiving system health. A number of practices are available to reduce total stormwater runoff, including water reuse and practices that increase retention of water in the soil mantle, but reducing surface runoff is not the same as maintaining groundwater recharge. There are many situations where both types of practice (volume reduction and recharge) must be used in conjunction if downstream receiving system protection is to be provided.

If an effective policy is to be developed, there must be an understanding of groundwater/surface water interaction, recharge/discharge zones, and hydrogeology. It is also important to be aware of the ambiguity that surrounds the issue so that an intelligent discussion and a better understanding of possible variables can result.

It is important to note that the following discussion does not address the impacts caused by the use of groundwater as a water supply source. While aquifer drawdown by water supply wells can result in the loss of groundwater resources, the issue is beyond the scope of this manual.

General Understanding

First, it is important to define what low streamflow is.

> *Low streamflow is that flow that occurs during periods of little rain, typically in mid-late summer and can be highly variable over time primarily due to watershed geology, climate, and topography. It can also be affected by many other factors.*

When it is not raining and runoff into streams through the drainage system has ended, the flow in the stream is derived from groundwater. If groundwater levels are reduced so that they decline below the stream bed invert, the stream loses its ability to have perennial flow. It then becomes an ephemeral stream and loses the potentially rich biologic atributes that a perennial stream can have.

Baseflow is influenced by groundwater gradient, hydrogeologic properties (eg. permeability) of the aquifer materials, and the properties of the materials at the surface water/groundwater interface. With urbanization, the main change is in groundwater gradient. Areas with steeper gradients are likely affected differently than areas of low hydraulic gradients. Recharge areas will be affected more than discharge areas. Examples of two types of aquifers are shown in Figure 11-1.

Our study of stormwater over the years has focused on imperviousness as an effects indicator. Impervious cover does significantly increase peak rates of discharge and volumes. The perception is that impervious cover will reduce groundwater recharge and cause water levels in urban streams to decline during dry periods. The groundwater table is not replenished, as surface runoff during storms will carry water away that would otherwise infiltrate into the ground.

From a stormwater management perspective, we initially controlled runoff by detention practices that might mitigate downstream flood increases, but detention ponds, and even retention ponds (due to bottom sealing) have not resupplied water to groundwater. Infiltration practices are among the few urban practices that provide groundwater recharge at least as a byproduct.

Attempts to detect the effect of impervious surfaces on stream baseflow are very difficult due to the need for long-term data from watersheds where the hydrological record has existed prior to watershed development and extended throughout the development period. Too often, gauging stations are discontinued or are on watersheds so large that development on a subwatershed level will not show significant change. In addition, groundwater supply may come from an area outside of a subwatershed or from an area that has not been developed.

As a result, most of our understanding stems from a rational expectation that imperviousness reduces groundwater recharge, which then translates into reduced stream baseflow.

Various Influences on Stream Base Flow

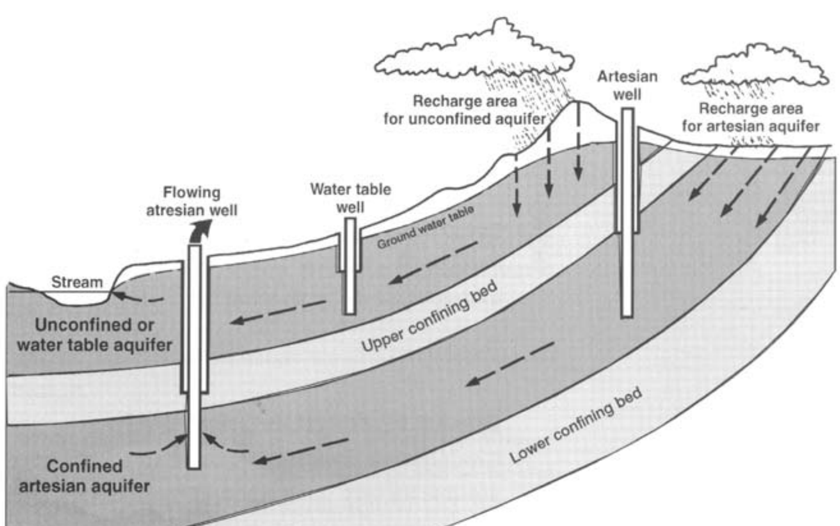
Transpiration, Evaporation, and Evapotranspiration

Transpiration, the process by which water from plants is discharged into the atmosphere as vapor, depends essentially on the same factors as those which control evaporation, namely air temperature, wind velocity, and solar radiation. Transpiration also varies with the species and density of plants and to a certain extent with the moisture content of the soil, in that a certain minimum amount of water must be available to the plant roots.

Evaporation is the process whereby liquid water becomes water vapor. It includes vaporization from water surfaces, land surfaces, and snow fields, but not from leaf surfaces.

Evapotranspiration, the combination of evaporation and transpiration, is the consumptive use of plants, or

Figure 11-1: Two Common Types of Aquifers



Source: Arc, 2003

the total amount of water absorbed by vegetation for transpiration or building of plant tissue, plus evaporation from the soil. Evapotranspiration has a significant effect on water yield from a watershed. It has to be considered on a seasonal as well as an annual basis to allow a good understanding of fluctuation and total amounts.

Timing of Groundwater Response

The effect of land use change on watershed yield is not instantaneous. While the evaporated and runoff components may respond relatively quickly to a land use change, increased recharge to a groundwater system may not express itself as a corresponding increase in surface discharge for many years. The timing of effects of a large-scale land use change on watershed yield will be different throughout the range of groundwater systems within a watershed.

Timing is effected by both shape and profile of a groundwater system, and these two factors have significant impacts on where surface discharge may occur in different land use scenarios. Another timing factor is the depth of unsaturated zone above the aquifer. This does not affect the equilibrium discharge conditions, but it will affect the timing of the response to recharge change, since aquifer heads will need to increase to a higher level before surface discharge can occur.

Predictors of groundwater response times are an essential part of predicting likely effects of land use change on low streamflow.

Leakage

Storm drains and sanitary sewers can function in two different ways. On the one hand, they may intercept groundwater, convey it downstream, and thus reduce groundwater levels in the vicinity where the water was intercepted.

On the other hand, sanitary sewer pipes may augment groundwater by leaking into adjacent soil and elevating groundwater levels through continuous flow.

The same could occur if potable water lines leaked. Water supply systems may have significant influence on groundwater levels, because leakage from a pressurized water main would occur all of the time, not just during storm events.

Both of these situations are common in urban utilities and may to some degree offset impervious surface impacts.

Compaction

The issue of compaction is extremely important when considering infiltration and subsequent impact on base stream flow.

Infiltration of rainfall into pervious surfaces is controlled by three mechanisms:

- The rate of entry of water through the soil/plant surface;
- The rate of movement of water through the vadose (unsaturated) zone; and
- The rate of drainage from the vadose zone into the saturated zone.

During periods of rainfall excess, long-term infiltration is generally the least of these three rates. The runoff rate after depression storage is filled is the amount by which the rainfall intensity exceeds the infiltration rate. The infiltration rate typically decreases during periods of rainfall excess. Storage capacity is recovered when the drainage from the vadose zone is faster than the infiltration rate.

The surface entry rate of water may be affected by the presence of a thin layer of silts and clay particles at the surface of the soil and vegetation. These particles may cause a surface seal that would decrease a normally high infiltration rate. The movement of water through the soil depends on the characteristics of the underlying soil. Once the surface layer is saturated, water cannot enter soil faster than it is being transmitted away, so this transmission rate affects the infiltration rate during longer events. The depletion of available storage in the soil affects the transmission and drainage rates, and the storage capacity of soils depends in turn on the soil thickness, porosity, and the soil-water content. The effective porosity of the soil is again affected by many factors, such as soil texture, root development, soil insect and animal bore holes, structure, and presence of organic matter.

The infiltration of water into the surface soil is responsible for the largest abstraction of rainwater in natural areas. The infiltration capacity of most soils allows low-intensity rainfall to totally infiltrate, unless the soil voids become saturated or the underlain soil is much more compact than the top layer. High-intensity rainfalls generate substantial runoff because the infiltration capacity at the upper soil surface is surpassed, even though the underlain soil might still be very dry.]

Urban development increases runoff due to a number of reasons, including impervious surfaces, but urban soils may also be significantly compacted during the urban development phase of land conversion. Soil compaction due to construction significantly reduces infiltration capacity of urban soils. For one, it will reduce the organic content of the surface layers, generally seen as a reduction of topsoil depth in a post-, compared to a pre-development condition. Also, urban land development generally involves massive site clearance of vegetation and movement of dirt with large areas of cut and fill, all of which require heavy earth-moving equipment that compacts soils. These areas, even vegetated, have significantly reduced ability to infiltrate rain into the ground, which further reduces groundwater recharge. In addition, the use of exotic plants over native vegetation may result in roots being shallower. Urban soils are very shallow to begin with, and the lack of penetration into the ground reduces infiltration rates further. A pre-development condition for a site may be forest with deep-rooted woody vegetation or meadow with shrubby plants that also have deeper roots. Prairie grasses have exceptionally deep roots that improve soil infiltration rates.

Percentage of Baseflow Versus Streamflow

A number of studies have discussed percentages of baseflow versus total streamflow. In all studies reviewed, the percentage of baseflow versus stormflow decreases as watershed imperviousness increases. That is not a surprising result, as impervious surfaces have profound effects on overland flow rates and volumes. But at the same time, a reduction in percentage of baseflow versus total streamflow does not necessarily mean that baseflows in a given stream are decreasing as a result of watershed development. It only means that a greater percentage of flow is now "quickflow" or stormflow.

The Auckland Regional Council study on the Oteha stream found the baseflow ratio reduced from 0.79 to 0.4 between 1982 and 2002. But baseflow is highly variable and shows no real trend. Rainfall variation in this study had a bigger influence than urbanization. There may be a lag effect where effects may be more rapid in high permeable materials such as sands or gravels.

Recognizing that watershed hydrology is determined by rainfall, land cover (which also affects evapotranspiration), soils, slopes, and conveyance, a reduction in the percentage of flow that is baseflow may not necessarily mean that baseflow itself is decreasing at all. There may be a reduction in evapotranspiration that allows a percentage increase in total streamflow as less is sent back to the atmosphere.

Trends would indicate the potential for decreasing groundwater recharge, but the effect has not been clearly defined. We know what level of recharge is necessary to maintain stream baseflow. What we do not know are the compounding factors or sources of other influences on groundwater recharge. Studies around the U.S. and New Zealand would indicate that there are situations where post-development baseflows are higher than pre-development, which presents a problem with establishing recharge requirements.

Case Studies

There are a number of case studies that may provide a greater understanding of the baseflow issue.

North Carolina (Evett et al., 1994)

Baseflow and precipitation trends at U.S.G.S. gage sites were studied in four urban centers and surrounding rural areas. The flows in these areas decreased in recent years. While the results tend not to support the development/reduced baseflow discussion, they did show that trends in precipitation alone cannot account for the decreased flow in urban and rural streams. Regional land use effects could be exerting some negative effect on the rural streams as well.

Explanations given in the study were the following:

- The urbanization effect on baseflow exists but may be too small to show up in the statistics.
- Some substrate types are less vulnerable to reduced groundwater recharge than others.
- The streams studied were large and of mixed land use. Factors outside the station area may have exerted an effect at the measuring point.

It was concluded from the study that there is some support for the theory that urbanization causes a decrease in low streamflows over time, but statistically, the results are inconclusive. It appears more likely that most small streams, both urban and rural, are experiencing decreasing low flows over time to a greater degree than would result from decreasing trends in precipitation alone.

Long Island, New York (Simmons and Reynolds, 1992)

This study and an earlier one (Simmons and Reynolds, 1982) attributed the alteration of flow components to the installation of sanitary sewers for the conveyance of treated wastewater to tidewater, the routing of stormwater directly to streams, and an increase in impervious surfaces throughout the watersheds. They reported as much as a 70 percent reduction in baseflow in streams draining Long Island between 1948 and 1985.

Upper Chattahoochee River Basin, Georgia (Calhoon, Frick, and Buell, 2003)

The baseflow component of total streamflow in Peachtree Creek has declined from approximately 50 to 30 percent since continuous streamflow measurements began in 1958. Equilibrium in the baseflow decline does not appear to have been reached in this watershed, and it can be extrapolated that without any additional increase in urbanization, further declines in baseflow will occur. Conversely, baseflow components of total streamflow in Snake Creek and the Chestatee River, watersheds that had little to no urban development, have been more consistent over time.

An extrapolation done in the study shows that for each percent increase in impervious area, there is a corresponding decrease in baseflow of approximately 2 percent.

It must be pointed out that the study did not actually look at baseflow level reduction but rather considered it as a percentage of total streamflow. The relative contribution of baseflow to total flow was reduced, but no conclusions can be stated regarding actual reductions in stream baseflow as a result of urbanization.

Upper River Rouge Watershed (Richards and Brabec, undated)

Analysis of discharge data from the Upper River Rouge at Detroit shows a gradual increase in baseflow and streamflow since 1932. The trend was consistent even in the late 1990s when the precipitation decreased over a three-year period. It is possible that changes in seasonal precipitation distribution account for some of this increase in baseflow. Most of the recharge in a watershed occurs in the dormant season, particularly in the late fall, when evapotranspiration fluxes are low. Climate changes that increase the proportion of precipitation falling in the dormant season could increase baseflow with no apparent increase in annual precipitation. Thus, if a drop in evapotranspiration is the cause of the trend, its decrease has to be related to either the presence of imperviousness or changes in the type and aerial extent of vegetation which are reducing the efficiency of evapotranspiration.

Another interesting possibility discussed is that climate changes have decreased the driving force for evapotranspiration over the study period. A decrease in evapotranspiration will increase the amount of water available for recharge. Annual potential evapotranspiration (estimated using the Thornwaite method) over the period of interest suggests that the driving force for evapotranspiration has not changed significantly. Total streamflow and surface runoff have increased significantly over the time period.

Australia (Zhang, L., et al., June 2003)

There have been a number of studies in Australia that estimate response of groundwater systems to changes in recharge that arise from land use changes. A primary emphasis of some of these studies is concern over expansion of saline land surfaces and rising river salinities that occur in many parts of Australia.

In addition, Australia is a very dry climate, and other studies have considered the effects that large-scale afforestation has on the volume of streamflow and the associated water allocations. The impact of afforestation on mean annual flow is well known, but efforts have been underway to better understand its impact on seasonal flow or flow regime. It was found that blue gum plantations would significantly reduce low flow and hence increase flow variability. Results indicated that the maximum reduction in mean annual flow would be 8 percent for Lake Eidon and 14 percent for Goulburn Weir if all suitable areas were planted.

New Zealand (Herald, 1989)

Monthly streamflow yields and flow duration curves for watersheds of pastoral, urban construction, and fully urbanized land covers were compared. Due to missing data, it was not possible to compare annual water budgets for the study watersheds. However, comparison of the area-specific yields for the periods for which data were available provided a useful index for the magnitudes of change likely to occur as urban development proceeds.

Groundwater recharge was markedly reduced as a result of urban development. An index of monthly groundwater recharge showed substantial recharge in the pastoral watersheds during four months of the study period, but only limited recharge in the urban construction watershed during three months, and no recharge in the urban watershed. An intuitive assumption commonly suggested in the literature is that the reduced groundwater recharge subsequent to development leads to a reduction of low flows. However, although results of the study show an increase in total discharge and a decrease in groundwater recharge, low flows were also seen to increase in both frequency and magnitude. These findings may suggest that more sustained low flows result from the urban watershed responding more rapidly to lower-intensity and shorter-duration rainfall than pastoral watersheds.

General Discussion

There is probably truth in both the assumption that infiltration to groundwater is reduced by urban impervious surfaces and that evaporative losses may be reduced since both interception storage and depression storage are often reduced by urban development. The net effect of these changes is commonly an increase in total runoff. When considering low streamflow or baseflow, the issue is one of prediction from a trend perspective.

This is an important issue, as designing for infiltration on a watershed basis may result in groundwater levels potentially being higher than during the pre-urbanization period. At the same time, disregard for potential drop in groundwater levels due to urbanization may result in loss of perennial streams and their associated aquatic ecology.

The important point to recognize is that the two issues are not necessarily at the opposite ends of the "hydrological spectrum." Through watershed-wide approaches, we can consider the variables that have the greatest effect on stream baseflows.

The historical approach of predominantly using wet or dry stormwater management ponds cannot address the issue. Any analysis of extending pond outflow durations will not address overall changes to groundwater recharge or discharge, because no stormwater ponds, no matter how large, can delay wintertime rainfall sufficiently for it to become summertime runoff. Yet exactly this magnitude of delay does occur under pre-development conditions, because far more of the precipitation is stored as groundwater than can ever be stored in stormwater ponds. This stored precipitation is also released from the groundwater much more slowly than from a pond. Therefore, we have to rethink our traditional approaches to stormwater management if we consider stream baseflow protection an important program issue.

The specific issues that seem to relate to variations of stream baseflow are:

- Watershed imperviousness;
- Compaction of soils;
- Loss of watershed evapotranspiration;
- Existence of significant natural recharge areas; and
- Leakage of water and sewer pipes.

With all that said and done, we need to recognize that there are adverse impacts to the urbanization of previously undeveloped land. If we are going to continue with traditional development approaches, we will lose many of those resources that attract us to a given location. If we can implement a stormwater program that addresses these items, stream baseflow impacts can be better predicted and designed for.

Recommendations

When considering recommendations for an approach to low streamflow maintenance, we need to address the items mentioned in the *General Discussion* section of this chapter. In addition, there are obvious overlaps with other chapters, especially Chapter 8, Impact Avoidance. We have to recognize the value of natural site features, including existing vegetation, and protect those features.

Reduce the Impact of Watershed Imperviousness

Urban land use will continue to create and maintain impervious surfaces. Roads must shed water for safety reasons, and structures cannot leak. With that said, we can reduce the impact of those impervious surfaces through a number of different actions, including the following:

Disconnection of Impervious Surfaces from the Historic Drainage System

This approach would mean disconnecting roof runoff from the historic drainage system. The basic approach here is to reduce the efficiency of the stormwater conveyance system by slowing the water down and allowing greater contact with vegetation and soil.

Green Roofs

Green roofs are being considered much more as a mainstream practice than they have been historically. In the early 1980s, rooftop storage was a measure considered for detention of stormwater to reduce potential increases in peak flow. At that time, it was eliminated as a practice because roof leakage was a problem. Using an impermeable membrane in conjunction with better site control will reduce leakage concerns. A good design approach for green roofs can provide significant hydrological benefits for smaller storms and improve evapotranspiration potential.

Water Reuse

Water reuse is a good practice even if stream baseflow is not an issue. Use of roof runoff for water reuse reduces not only the total volume of stormwater runoff but also demand for public supply. Unlike evapotranspiration, which is very seasonal in benefit, water reuse provides benefits all year, as long as there is water use in a residential, commercial, or industrial property. Water reuse can be very beneficial on industrial properties where water is essential for day-to-day operations.

Bioretention Practices

Bioretention practices include rain gardens, filtration systems that use organic materials, swales, and filter strips. Studies have detailed volume reductions for these types of practices in the range of 20 to 35 percent, and a recent study in Melbourne detailed a 54 percent annual reduction of total runoff. That is a significant reduction in runoff volume. Bioretention practices can be used in very urban environments such as parking areas, or along roads as swales.

Infiltration Practices

In an urban environment, putting water into infiltration will help maintain groundwater recharge and take out some of the stormflow and pollutants, which provides multiple benefits. Infiltration practices are especially appropriate on small sites where total drainage to the practice is fairly small. Having more practices serving smaller drainage areas, rather than fewer practices serving larger ones, is desirable, as clogging may reduce overall effectiveness and total clogging of one or more systems is not as critical if there are more practices serving the same property. In addition, when imple-

Melbourne bioretention practice where total volume study was done

Example of an infiltration practice used for groundwater recharge menting infiltration practices, it is important to put a greater effort into providing infiltration in recharge areas and less effort into discharge areas. There is still concern about long-term performance of infiltration practices, and an aggressive program of site inspection and maintenance is necessary to ensure that proper maintenance is accomplished.

It is important that all of the above practices be used in a "treatment train" approach to protecting stream baseflow. Possibilities for a reduction of "effective" imperviousness exist on any new development and can often be considered cost-effectively as a retrofit. In Auckland, a number of industrial sites have instituted water reuse on sites that are completely impervious, and benefits include reduced water charges for plant operation in addition to reduced stormwater quantity and quality concerns.

Revegetation of steep slopes and footprinting house locations in a development to minimize surface runoff

Example of a porous block parking area

Minimize the Impact of Soil Compaction

The easiest way to reduce soil compaction is to keep construction equipment off site areas that are to be left in a natural state. That is often not possible due to maximum development for profit margins. Still, there are a number of ways in which this concern can be addressed and soil permeability improved:

1. Where cuts or fills of at least two feet are intended to facilitate site development, the expected permeability of the soil may be reduced. Stormwater management computations that detail post-construction hydrology should use a modified approach to soil classifications.
2. In areas of significant site disturbance, and where there is less than two feet of cut or fill, soil classifications are not modified, but the approved site permit should contain a construction requirement that significantly disturbed soils in areas where those soils remain pervious should be chisel-plowed. Chisel-plowing will break the surface crust of the disturbed soil and allow for a greater infiltration rate. This would provide a good foundation for the placement of topsoil and prevent slippage of the topsoil on slopes that become saturated.
3. Use of soil amendments (compost, polyacrylamides) or otherwise modifying soil structure and chemical characteristics is becoming more popular. At present, there is little information to quantify benefits or problems with their application.
4. Rather than using a minimum depth of topsoil that is stockpiled and disposing of the rest off-site, the depth of topsoil that is retained on site should be maximized. This topsoil can act as a sponge during rainfall. One good requirement would be to maintain, to the extent possible, the same volume of topsoil on a property post-development that existed prior to development.
5. Woody vegetation should be planted in open space areas to improve root depth penetration. There will be a period of time when compacted soils reduce permeability, but long-term benefits will be obtained.

Maintain Watershed Evapotranspiration

In a number of places in this manual the value of vegetation has been stressed. Maintaining evapotranspiration in a watershed is an effective balance to prevent increases in groundwater levels. For years, people have been planting willow trees in areas of high water table to reduce groundwater levels. Native vegetation, deeper roots, protection of existing woody vegetation, or planting more vegetation all provide a wealth of benefits, including seasonal drawdown of groundwater and maintaining a balance for groundwater recharge. The above-mentioned Australian study on afforestation shows that woody vegetation can have a significant impact on baseflow. Those study results can be considered in an urban context where woody vegetation has been removed and groundwater levels could increase.

Leave Areas of Significant Recharge Natural

In every watershed, there are areas where significant groundwater recharge occurs. In general, these areas are away from stream channels but not situated on steep slopes. They are sandy soils and sandy loam soils that have high infiltration rates. In existing wooded areas, these soils act as a sponge for any rainfall that lands on them. Before development occurs in a given watershed, watershed planning should detail environmentally sensitive areas that include areas of significant groundwater recharge. Those areas should be targeted for limited growth and site disturbance. Watershed areas with limited recharge capability would be more suitable for higher levels of site disturbance and development.

Prevent Water and Sewer Pipe Leakage

Older water supply and wastewater pipe systems certainly leak. Design criteria for both also have a factor of safety for pipe sizing that accounts for infiltration or exfiltration. Newer construction techniques can minimize that historic leakage problem, but this is only done on an emergency basis or where pipe systems must be upgraded to account for increased demand.

Wastewater systems are in the unenviable position of having exfiltration concerns when the pipe is above the water table and infiltration concerns when it is adjacent to a stream and in the water table. As a result, there is both the potential to augment groundwater flows and to reduce groundwater levels.

The actual impact of this is not expected to be significant. If you assume 10 houses per hectare with each house using 200 m^3 of water per year, and there is a 20 percent leakage of water into the ground, monthly increases are expected to be only 3 mm over those generated from inputs of rainfall. This is a negligible increase that would be expected to have a limited effect on groundwater levels.

Concluding Comments

One very important point in considering stream baseflow is the need for planning from a watershed perspective. The protection of, for example, a sensitive trout or salmon stream can only be achieved if it is considered entirely from a watershed perspective. Many program priorities can be addressed by a standardized approach that incorporates impact avoidance principles. Protecting stream baseflows will rely to some degree on these principles, but it will also require fairly detailed land use considerations if the goal of stream baseflow maintenance is to be achieved.

The bottom line of the discussions presented in this chapter is that we need more information in more areas to be able to make predictions for the impact of development on stream baseflow. We do not even have a clear understanding of whether stream baseflow will increase or decrease. There is certainly an expectation that stream baseflows will decrease with increasing watershed imperviousness, but existing studies cannot verify whether that is true.

We cannot expect people to pay thousands of dollars to implement practices that may provide a stream lowflow benefit but may also have little value or even increase groundwater levels above the pre-development one.

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CHAPTER 12

Watershed Management Approaches

CHAPTER 12

Watershed Management Approaches

The complexity of urban runoff quality and quantity problems has been documented and discussed throughout this book. This chapter discusses the benefits, challenges, and technical requirements of using a watershed-based approach to address these problems and to manage urban runoff in a truly comprehensive manner. To accomplish this, the chapter presents the following topics:

- What is a watershed?
- What is watershed management?
- Why use a watershed management approach to manage urban runoff?
- Which aspects of urban runoff can watershed management address?
- What are the technical and program requirements of watershed management?
- What challenges and difficulties can be expected when utilizing a watershed-based approach?

The chapter also includes information on the development of total maximum daily loads (TMDL), a federally mandated, watershed-based approach to addressing water quality problems along specific water bodies. The chapter concludes with a look at what future watershed management efforts may encounter and address.

As with many of the topics presented in the original 1994 *Fundamentals of Urban Runoff Management*, much has been written regarding watershed management over the last 11 years. This growth is expected to continue and even accelerate in the future due, in part, to the almost limitless range of problems and solutions that could be addressed through a watershed-based approach. A recent Internet search for information on "watershed planning" returned over one million hits.

Not surprisingly, EPA has a number of guidance documents available on watershed management, including the website www.epa.gov/watertrain, which provides online training in a variety of watershed management issues. The site includes a number of topics including watershed change, analysis and planning, watershed ecology, and watershed management practices and community issues. It serves as an excellent starting point to acquire a greater understanding of watershed management issues.

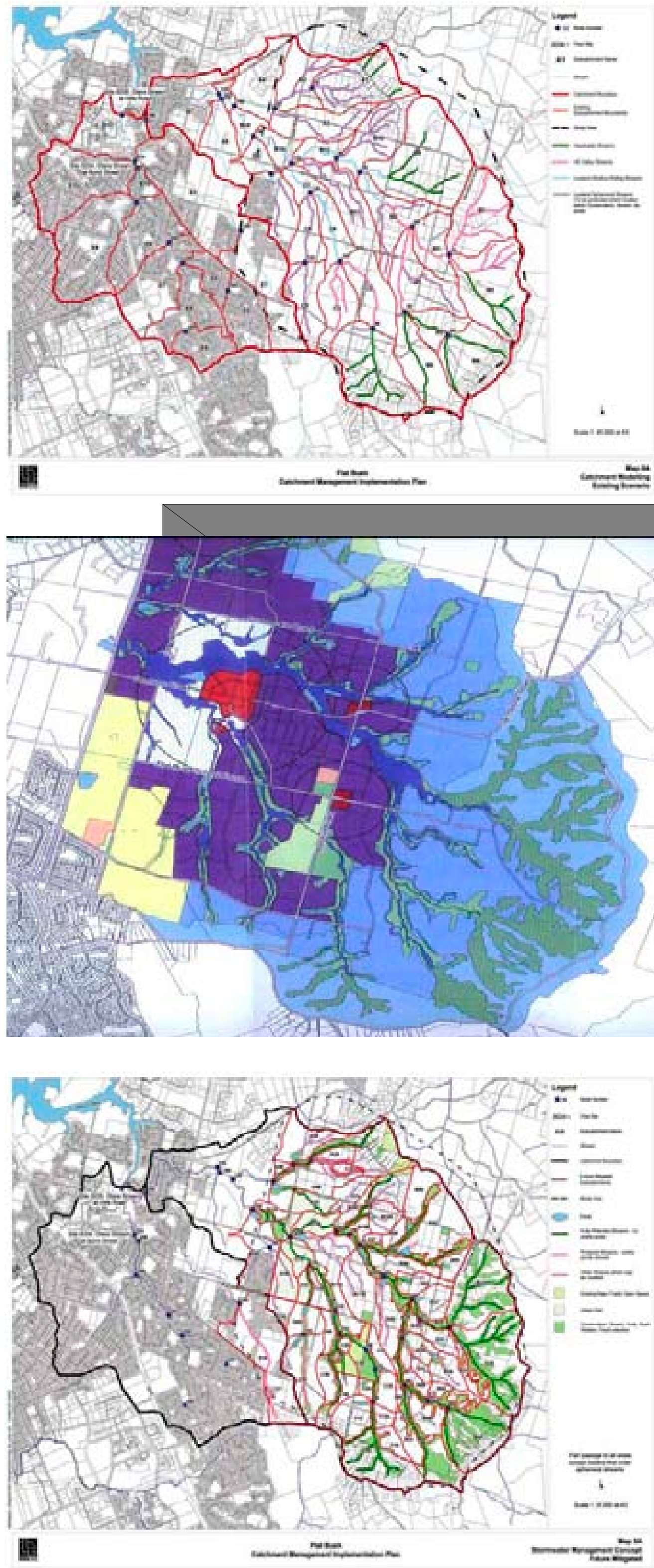
In light of this vastly greater amount of information, it is not the purpose of this new chapter on watershed management to simply repeat or cite information that is readily available from others. Instead, the chapter will discuss those watershed management issues that, in general, have not been considered or addressed by either researchers or program managers. These discussions will seek to identify not only the benefits of watershed-based runoff management planning but also the short- and long-term commitments that must be made to develop an effective watershed plan that will not diminish in value or effectiveness over time. Clearly, watershed-based approaches have proven to be essential components of successful stormwater management and resource protection programs. Identifying not only the benefits but the challenges associated with watershed management in this chapter should increase the number of successful watershed management efforts.

What is a Watershed?

A watershed is a geographical area from which stormwater runoff, and the pollutants and other materials borne by that runoff, drain to a down gradient central collector such as a stream, lake, or estuary. As such, a watershed is often named for the water body that conveys this runoff at the outlet. The term watershed means just that – an area of land that, from rainfall, produces or "sheds" water in the form of runoff and delivers it to a specific point. Figure 12-1 shows a series of watershed maps: the first details ephemeral and perennial streams, the second shows intended land use, and the third shows riparian cover and locations of stormwater wetlands.

A watershed's size is in part defined by the topography of the surrounding land and upon the location chosen as its outlet. Often the latter reflects a point where runoff rates and/or sediment or other pollutant loadings need to be computed. Depending upon the resultant size, a watershed may also be described as a drainage area or catchment, although these terms normally apply to areas smaller than those typically considered watersheds. As the larger of these areas, watersheds can be considered to be comprised of a collection of drainage areas, catchments, or subwatersheds. A watershed may alternately be referred to as a basin, particularly by federal agencies such as the U.S. Army Corps of Engineers. Whatever term is used, the general concept remains the same: When a raindrop falls within a watershed and produces runoff, that runoff will ultimately pass through the watershed's outlet.

Figure 12-1: Watershed Maps Showing (From Top) Stream Networks, Land Use and Riparian Corridors, and Stormwater Practices



What is Watershed Management and Why Should it be Done?

If the goal of an urban stormwater runoff management program is to address a runoff-related problem or protect a water resource at a particular location, and a watershed represents the entire area of land that contributes runoff to that location, using a plan that addresses the problem or protect the resource that is based upon managing runoff over the entire watershed is a technically sound strategy. By basing the plan on the entire watershed rather than a single location or portion within it, all of the relevant factors contributing to the problem can be included in the planning process. In addition, this approach increases the number of potential solutions to the problem or threat.

This logic can be used to both describe what watershed management is and why it should be pursued. Basing urban runoff management decisions on the entire watershed provides a flexible framework for considering and integrating all pertinent factors and resources into both the analysis of runoff-related problems and threats and the development of their solutions. Watershed management also allows multiple problems to be prioritized and multiple solutions, including their development, implementation, and funding, to be sequenced in the most efficient and effective manner. And where alternative solutions exist, watershed management provides the framework by which they can be comparatively evaluated using the broadest set of factors.

Similarly, if a regulatory, policy, or program approach to a problem or threat is considered to be the most effective solution, watershed management offers the most comprehensive and effective means by which such solutions can be identified, developed, and implemented. Such results cannot be achieved using a localized, piecemeal, or site-specific approach. As previously discussed in Chapters 1 and 2, this is particularly true of the EPA's TMDL approach which defines the pollutants of greatest concern and then uses regulations and discharge permits to achieve required pollutant load reductions from dischargers. Such a result can be achieved only through a watershed-based analysis and implementation effort.

In light of all the above information, it may be more intriguing to ask why watershed management should not be done rather than why it should be.

Watershed-based urban stormwater management continues to expand, a trend that is expected to continue in the future as more runoff-related issues are identified and our watershed management skills and databases increase. The next section of this chapter will discuss those urban runoff- and water resource-related areas that either should or could be addressed from a watershed management perspective.

The Watershed Management Universe

Runoff Quantity Considerations

Runoff quantity impacts have been addressed with a watershed management approach for several decades. Watershed managment was initially used to control or reduce flooding, but now is commonly employed to control development-induced impacts caused by increases in pollutant loading, peak runoff rates and volumes. Such controls have been achieved through structural, nonstructural, and regulatory measures. The various watershed management plans described in detail in Chapter 10 are all examples of this proven use of watershed management.

More recently, watershed management tools have been increasingly used to investigate how to protect stream channels from erosion due to the increased runoff volumes resulting from watershed development. Many questions regarding the mechanisms by which such erosion occurs and the runoff controls required to prevent it, remain unanswered and further research and analysis are needed. Nevertheless, there is growing consensus that a stream channel's physical structure can only be protected if watershed development is designed to occur without causing significant change in watershed runoff rates and volumes. Such results are the fundamental goal of the newer low impact design approaches discussed in detail in Chapter 8. Developing the site design strategies, measures, and requirements necessary to achieve these results can clearly best be done on a watershed-wide basis. Attempting to develop and implement them on a site-by-site basis will be far less successful, and in many cases will not work.

Runoff Quality Considerations

Water quality problems and threats are also best addressed on a watershed basis. This approach can not only best identify the overall extent and severity of the problem or threat, but it also has the greatest potential to identify all of the relevant sources or causes. For example, if downstream water resource protection damage is the problem, such as in the Chesapeake Bay and Puget Sound runoff management programs, watershed-based studies are necessary to determine all causes of degradation. Once such a determination has been made, the same watershed-based framework can be utilized to develop a comprehensive range of structural and nonstructural solutions, oversee their implementation, and administer their financing, operation, maintenance, and/or enforcement.

Addressing the causes or sources of runoff quality problems can be used to further illustrate the power of watershed management and its advantages over a limited, site-by-site or piecemeal approach. For example, watershed-based studies of the Upper Waitemata Harbour in New Zealand (NIWA, 2004) demonstrated how a single sub-watershed discharging to the harbor estuary accounted for virtually all of the zinc that was damaging this important aquatic resource. This knowledge will allow the development of a focused zinc control program in the subject watershed only, thereby avoiding the considerable time, effort, and other resources that would have been wasted developing and implementing similar controls throughout all of the estuary's watersheds. Furthermore, the framework established by this watershed management approach can further be used to determine the optimum combination of structural, nonstructural, and regulatory solutions to address the zinc problem. Once again, such results can only be achieved by utilizing a watershed-based approach.

Wastewater Considerations

Watershed management principles and capabilities can be extended to address both wastewater and water supply issues. This applies to both combined sewer overflows (CSOs) and combined sewer systems (systems where both sanitary and stormwater are conveyed to a plant for treatment). In both cases, a watershed-based approach is essential to addressing runoff quality or water body impacts. In the case of a CSO, segregating stormwater from wastewater would eliminate the discharge of untreated wastewater into the receiving system. As for combined sewer systems, separation of wastewater from stormwater would then allow smaller stormwater flows to discharge untreated where they would have gone to a wastewater treatment plant. Very clearly, prioritization needs to be made on water quality issues when combined sewers are considered. In addition, it is absolutely necessary to include consideration of combined wastewater systems in a watershed-based context if the goal is downstream water resource protection. If funding is limited to a given level, a greater return on the expenditure may be realized by prioritizing separation of combined systems ahead of implementing nonpoint source controls on urban area runoff. Such decisions can only be made using a watershed-based approach.

Similarly, the use of separate wastewater systems is important when considering watershed management. The frequency and volume of overflows from a combined sewer system may have a significant effect on receiving water quality. If the combined wastewater and stormwater system is so undersized due to both system age and ongoing watershed urbanization that damaging overflows occur on a frequent basis, upgrading the combined sewer system to reduce the frequency of such overflows may in fact be the best approach to improving receiving water quality. New development or redevelopment in such a watershed may still have stormwater requirements placed on them, but the expenditure of public funds would target the combined sewer system upgrade. Once again, such solutions can only be identified and developed through watershed management.

Water Supply Considerations

There are also situations where continuing watershed urbanization can have significant water supply impacts and where a watershed management approach can best be used to address them. For example, if current public water supplies are inadequate and the expansion of the system is either infeasible or too expensive, the new development will need to rely on on-site groundwater sources of potable water. At such developments, both groundwater recharge to increase the supply of water and water reuse to reduce its demand will help to ensure the success of this water supply strategy.

In addition, both water reuse and groundwater recharge can not only be a valuable water supply tool but they can also reduce development site stormwater runoff volumes. For example, approximately 60 percent of average annual residential water use is for toilet, laundry, and outdoor use. If the water needed for these activities could be obtained from roof runoff that was captured and stored, it would be removed from the stormwater drainage system. Such reuse then becomes a volume reduction practice in addition to reducing reliance on potable water. Runoff volume reductions can also be achieved if a portion of a development's site runoff can be recharged into the groundwater.

The effectiveness of such practices can best be evaluated through a watershed management approach that includes consideration of both public water supply and stormwater management needs. Such a watershed-based approach could then include cost considerations to determine whether such on-site measures could effectively replace the need to expand off-site public sources. Benefits of water reuse can become even greater for commercial or industrial sites which traditionally use significant amounts of water. The historic provision of cheap, high-quality water has limited consideration of water reuse in the past, but the additional need to consider volume reduction practices on a stormwater program should lead to greater consideration of water reuse in the future. This is only possible if the issue is considered from a watershed management perspective.

Watershed Management as a Means to Managing Growth

Historically, stormwater runoff impacts have often been considered as simply the effects of land development and watershed growth that can be addressed through management practices after such growth has been planned. This is based on the expectation that stormwater management practices can be relied on to adequately mitigate the effects of almost any type and degree of watershed development. This expectation has led many planners to exclude considerations of stormwater management capabilities and limitations from land development and growth decisions and to include them in the planning process only as a response to those decisions. However, recent studies, including those detailed in earlier chapters of this book, have highlighted the limitations of stormwater management and have demonstrated that adverse, long-term impacts on receiving waters due to watershed development can occur despite the level of stormwater management controls applied to that development. In such cases, the abilities and limitations of stormwater management practices must be considered an integral part of land development and watershed growth decisions and not simply turned to as a response after the fact.

In addition to the specific examples described in the research, there are general examples that highlight this problem. For example, the headwaters of a watershed have very steep slopes that significantly reduce the range of effective stormwater management practices that may be used to address the impacts of development. In other portions, clay soils may preclude the use of infiltration or other runoff volume management practices that would ordinarily be expected to prevent increases in runoff volume. Or conversely, highly permeable sandy soils in any area may achieve significant aquifer recharge and should be left undisturbed rather than built upon and mitigated by stormwater management measures that cannot achieve the same long-term recharge as the existing, natural systems. If such stormwater management factors are not identified and considered during the development of master plans, zoning ordinances, and other land use planning processes, the final development types and levels authorized by those processes may have adverse impacts that exceed the capabilities of the available stormwater management practices. However, if such factors are identified and included in the land use planning process, such impacts can be avoided through the selection of development types, levels, and locations that can be controlled by available stormwater management measures. Unfortunately, land use and development decisions are often reached without regard for stormwater management possibilities and constraints. This can lead to land use decisions that will have severe and even irreversible impacts that could have been avoided.

During land use planning efforts for a given watershed, decisions should be made as to what level or degree of development, if any, can be allowed to occur with a reasonable expectation that the available stormwater management practices in the watershed will be able to manage the resultant adverse stormwater impacts. Only those development levels and types that are controllable by available stormwater management practices should be allowed in the watershed and included in the watershed's land use plans and regulations. Such land use planning and regulation decisions can only be made through a watershed management approach.

Finally, the watershed management approach provides planners and regulators with both the opportunity and framework to combine stormwater management considerations with other traditional land use planning factors, such as:

- Wetlands;
- Floodplains;
- Existing vegetation;
- Soils;
- Slopes;
- Riparian corridors;
- Historic or cultural sites; and
- Terrestrial ecology and landscape form.

Using New Development to Address Retrofit Needs

Typically, the selection of required stormwater management practices for a proposed land development is based only on preventing the adverse runoff impacts of the development itself. While this may prevent a worsening of existing runoff quantity and quality problems, it does not address already existing quantity or quality problems in the watershed and its water bodies. Addressing these problems typically requires retrofits in existing developed areas. However, due to a number of factors, including lack of available space, higher property values, and greater design and construction constraints, stormwater retrofits can be very difficult, expensive, and disruptive to implement.

However, watershed management planning can provide both land developers and runoff program managers with the tools and framework to incorporate retrofit requirements into the design of a proposed development's on-site stormwater management practices. While this will typically require a larger sized practice or a greater number of practices to be incorporated into the proposed development's design, many of the design and construction complexities normally encountered in retrofits can be avoided. In addition, design and construction of a single, larger on-site stormwater management practice can usually be achieved with less cost and required land than a standard on-site practice combined with a separate, off-site retrofit practice. While the program would be responsible for compensating the developer for the extra design, construction, and land costs of the larger practice, the watershed management approach could also serve to establish a watershed stormwater utility or other assessment program that could be used to generate the required compensatory funds.

While combining retrofit measures with those required for new land developments can appear to be a logical and effective way to address existing runoff and water resource problems, the decision to do so can be very complex and require consideration of many on-site, off-site, and program-related factors. However, it can be seen that such decisions can be much better supported by a watershed-based urban runoff management program than one based upon individual development sites.

In conclusion, an effective urban runoff management program must be pragmatic and based on technically sound data and definitive objectives. The program not only needs to be aware of the required improvements, but where are how those improvements can best be accomplished. This is especially true in an existing urban environment where retrofit options are limited and expensive. These need-to-know answers can best be obtained through a watershed management approach.

Watershed Management Decisions and Considerations

While extolling the capabilities, benefits, and even virtues of watershed management may not require a significant commitment of time, money, and effort, developing and operating an urban runoff management program based upon a watershed management approach certainly can. This is not unusual, since the aspects that yield the most comprehensive and beneficial outcomes, whether they are programs, structures, products, services, or relationships, are usually those that require and receive the most input. Nevertheless, it would be helpful at this point to review some of the important decisions and commitments that must be made to achieve an effective and efficient watershed management program. These range from issues that must be addressed both prior to and following program startup. Unfortunately, there have been far too many instances where a watershed study was completed and then put on a shelf and never touched again. There are a number of issues that need to be considered before going down the watershed management path.

Watershed Model Selection

As discussed in detail in Chapter 2, the actual process of converting rainfall into runoff is extremely complex. This complexity increases when one begins to include not only the rate or volume of runoff but also pollutant loadings and impacts to water resources. In light of these complexities, the actual physical processes are replaced with mathematical equations and models that are used to predict the results or outcomes of the real processes under various conditions, assumptions, and constraints. Such models may be based upon a single real or hypothetical rainfall-runoff event or a long, continuous series of actual events based upon a similarly long event data record. Since the equations and algorithms that a model is based upon are only approximations of the actual rainfall-runoff processes, the results of actual runoff events are typically needed to adjust and check or, as modelers say, calibrate and verify the model's predictions in order for it to be reliable.

Typically, the complexity of the actual processes also limits the scope or predictive capabilities of rainfall-runoff models. While large advances in model theory and computing power have been made in recent years, along with advances in the range and precision of available databases, there are still relatively few computer models that even attempt to simulate more than a few rainfall-runoff processes or parameters. Those that do sometimes suffer from the effort to predict only a limited range of parameters. For every broad-based computer model that can predict a large range of parameters, there are easily a half dozen more narrowly focused models that can more accurately predict a specific parameter in that range.

Due to the amount of data that must be processed and the number of equations that must be solved, virtually all rainfall-runoff models are run or exercised (another modeling term) on computers. In addition, virtually all watershed management efforts require the use of one or, at times, multiple computer models in order to accurately and efficiently analyze all of the pertinent factors, processes, and conditions. As a result, selection of the appropriate computer model or models is one of the most important decisions associated with watershed-based runoff management. And since model selection typically occurs in the earlier phases of the process, there is generally a limited amount of watershed, resource, or problem information available to base model selection on. As a result, successful watershed modeling efforts are typically performed by those with prior knowledge of a particular model's capabilities, requirements, and limitations and extensive experience in its use.

In selecting a computer model for a watershed management plan or program, the selection process should be relatively straightforward. Based upon a desire to produce the best possible plan, the selected model should be the one that produces the best results. The difficulties, however, come in defining what constitutes 'best', a quality that can be measured from several different reference frames. Some of these include:

- **Applicability and accuracy of predictions** – The selected model must be able to predict the answers or outcomes required by the watershed plan with the required level of accuracy. If, for example, the goal of the plan is to reduce annual TSS and nutrient loadings in runoff, the model must be able to predict these parameters in this time frame.
- **Soundness of model theory and equations** – While the accuracy of model predictions can be checked to some extent against real event data, such checks are usually limited to a relatively narrow range of parameter values and input conditions. Therefore, they cannot be solely relied upon in judging a model's accuracy. The model's theoretical basis, assumptions, equations, and algorithms must all be scientifically sound, reliable, and defensible.
- **Extent, availability, and cost of required input data** – In watershed modeling, data needs can be measured both in terms of the cost required to obtain it and the relative value of the results it produces. With regards to required data, model selection must begin with consideration for overall data acquisition costs in order to ensure that such costs are compatible with the overall watershed management plan budget. Next, the relative value of such an expenditure must be evaluated to determine if the value of the results produced by the acquired data is worth the cost of acquisition. At this point in the model selection process, model input data can be considered an investment in the model's output. Is the cost of the investment in obtaining the required data worth the value of the answers returned by that investment? Unfortunately,

many well-intentioned modeling efforts are thwarted by excessively high data acquisition costs or diminished by the lack of value produced by that expenditure of program or plan funds.

- **Model familiarity and ease of use** – Of these two model selection factors, model familiarity may be the most important, since a modeler's familiarity with a particular model is usually reflected in the ease with which they use it. However, model familiarity does not only pertain to data acquisition and input requirements, model operating commands, and output options and review procedures. It also includes knowledge of a model's capabilities, limitations, computer requirements, operating bugs, accuracy, precision, and flexibility, as well as the ability, effort, and techniques required to efficiently calibrate and verify it. In other words, model familiarity may be described as the ability to know whether a model's predictions are acceptably accurate and, when they are not, to know how to improve them. Ease of use should also not solely be considered in terms of the number and simplicity of operating commands. Since most watershed-based rainfall-runoff models require a significant amount of geographic data such as subwatershed sizes, land uses and land covers, soil characteristics, slopes, and pollutant loadings, data input can involve considerable effort unless it can be automated. Similarly, model output analysis can be cumbersome and costly unless the model includes sufficient analytical tools, or results can be easily exported to other analytic software.

From the above, it can be seen that the best model for a watershed management plan or program is the optimum combination of capability, accuracy, data needs, ease of use, and past experience. In more general terms, the best model can be seen to be the one that meets output needs without being overly complicated or data-intensive.

Finally, it should be noted that, while model selection typically occurs near the start of a watershed management effort, it should not be the first activity. Too many watershed studies or management plans have begun with model selection, followed by a determination of the study's goals or required answers. In such cases, the answers sought by the study end up being determined by the model's capabilities. Instead, the goals, objectives, and desired answers should be determined first, followed by the selection of the best or most appropriate model capable of achieving them.

Data Needs

Closely linked to model selection is the data required to drive the model or to achieve the level of accuracy that is needed for a required or desired output. As noted above, data acquisition can be an extremely time-consuming and expensive component of the overall watershed planning effort. It is essential to know the data needs of a specific model before initiating the watershed modeling effort. Questions need to be answered regarding the general availability of required data and how time-consuming and costly its acquisition will be.

Typically, types of required watershed model data include the following:

- Rainfall;
- Topography;
- Watershed boundaries;
- Soil and subsurface characteristics;
- Existing and future land use and land cover;
- Runoff conveyance systems and outfalls;
- Wastewater overflow locations and details;
- Existing stormwater management structures;
- Existing water quality data;
- Groundwater levels; and
- Receiving water conditions and characteristics.

In performing the watershed study or analysis, it may be necessary to link watershed conditions with the receiving water responses to determine the effectiveness or benefits of various stormwater management or treatment options. This can be a complex process that may require significant receiving water data from which to predict results. For example, in Figure 12-2, samples of sediments in estuarine areas of the Waitemata Harbour and Manukau Harbours in Auckland, New Zealand were first obtained to provide historic rates of accumulation of metals over time. Then toxicity levels were determined along with trends toward increasing toxicity over time. Figure 12-3 shows changes in zinc and lead over time. Both figures demonstrate the concern that must be given to reducing zinc and copper levels in urban stormwater. Failure to implement a zinc and copper reduction program will cause environmental problems in the future. The second figure also shows the reduction in lead in urban stormwater, which probably relates to elimination of lead in gasoline. Based on this information, watershed modeling of the pollutant inputs lead, zinc, and copper was used to both predict future impacts and assess the ability of various stormwater management practices, including source control and runoff treatment measures, to prevent them. This modeling effort was then used to determine the level of pollutant reduction needed to alter the rate of pollutant accumulation in the harbors. While this was a very expensive modeling approach, it was justified first by the need to reduce or reverse pollutant concentrations in bottom sediments and second by the fact that the model could also be used to identify the necessary pollutant control approaches and implement them in a cost-effective manner based upon derived benefits.

Data Accuracy

Modeling in general and watershed modeling in particular are only as good as the data used in the model. Input data errors can cause significant errors in model output. Since these model outputs could result in the expenditure of millions of dollars for implementation and additional millions for subsequent operation

Figure 12-2: Sediment Sampling Program for Waitamata and Manukau Harbours

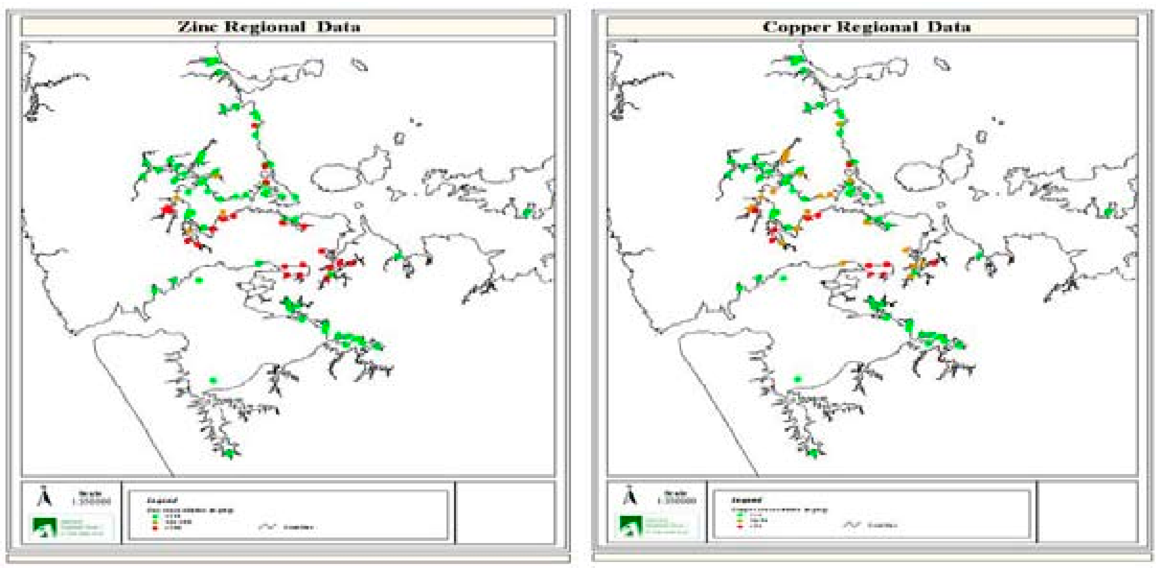
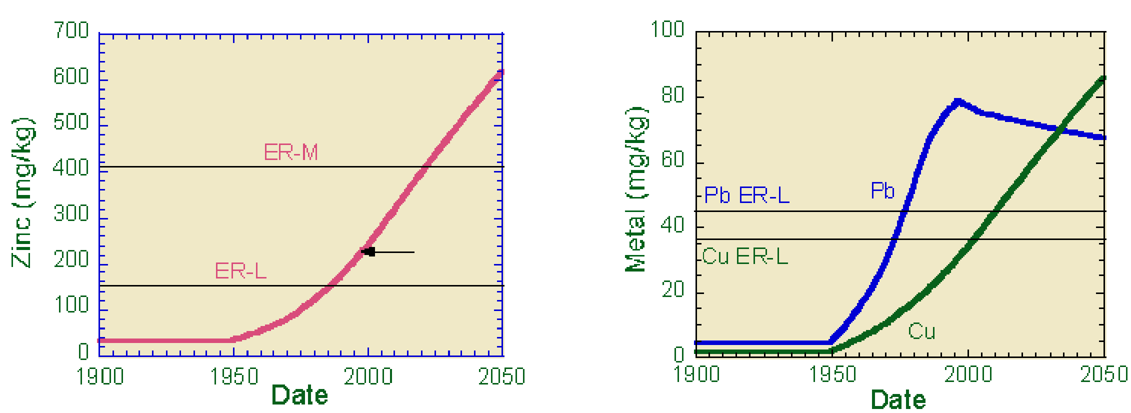


Figure 12-3: Consideration of Zinc, Lead and Copper in a Receiving Environment Showing Concentration Changes Over Time



and maintenance, input data must be accurate if such amounts are to be well spent. Required input data accuracy can be determined by the extent to which its variation will affect model results.

From a review of past watershed modeling efforts, certain types of model inputs that require improved accuracy have been identified. These include unit runoff pollutant loadings for various land uses and pollutant removal performance data for various stormwater treatment practices. Available data for these two key model input parameters are highly variable and, as a result, can usually only be relied upon to make planning level decisions. Land use loading data is particularly variable and can be read interpreted differently by different people. Experience is the best guide in determining unit loadings for various land uses, so it is important to involve experienced individuals in this component.

In addition, the lack of common input data collection and reporting protocols has meant that much data is not transferable from one watershed study to another. While this situation has been improving recently, it should still be considered indicative of general watershed study conditions. The American Society of Civil Engineers (ASCE) Stormwater Database (www.bmpdatabase.org) is a good start in terms of a consistent protocol for collection of performance data for various practices. At the present time, its data should be considered preliminary, and practices such as constructed wetlands will have variable performance data depending on location.

Another area where input data is extremely limited is the combined effect of runoff treatment practices arranged in series and the performance benefits that can be gained by that approach. Many stormwater management programs, publications, and experts have promoted this stormwater "treatment train" approach for many years. However, we still have only a general understanding of the performance of such an approach, and while we believe in its quality, we need more quantitative information to gain a better understanding of it, particularly for modeling purposes.

A Stepwise Approach Toward Comprehensive Results

It is unrealistic to expect to solve all runoff and water body quantity and quality problems through a watershed management process in the short term. Improvements to actual runoff and receiving water conditions will generally require an iterative process. However, this reality does not prevent significant progress being made in the short to medium term as priorities are identified and addressed through watershed management efforts. Additionally, a long-term vision for watershed-based stormwater management needs to be identified.

Furthermore, since large-scale land development activities have occurred for at least the last century in many urbanized areas, it may similarly have taken a hundred years for receiving waters to reach their present impacted condition. Therefore, it appears reasonable to expect that it will take considerable time for these waters to show significant recovery. For example, many older urbanized areas may still have galvanized metal roofs that, until replaced with a more runoff-neutral material, will continue to be a significant source of zinc to a receiving water and its aquatic environment. Sources of such pollutants need to be identified and strategies developed to address them. Some strategies may be placed in the 'too hard' basket for now and dealt with in the future, while others can be addressed immediately, depending on their relative importance and the availability of funding. Once an overall watershed management plan has been developed, an implementation strategy must be developed with public input to determine the degree to which improvements can be made and when.

Throughout this overall development and implementation period, small steps can be taken to make improvements or reduce the rate of system decline. While such steps are being taken, more information on both impacts and planned solutions will become available, along with new tools or approaches that may augment already identified actions. In other words, it is important to continually take manageable steps toward comprehensive watershed management and not delay the entire process by placing things in the 'too hard' basket. An initial, limited watershed management effort can provide information on the magnitude of a runoff or receiving water problem as well as on the next steps necessary to address it. While certain components of an overall watershed management planning effort may appear too complex or impractical to be implemented in short term, they should nevertheless be developed whenever possible. Their implementation may become more feasible over time as practical, technical, institutional, and social obstacles are overcome by increased research, knowledge, interest, and funding.

The Need to Update

Development of a watershed management plan and the implementation of its recommendations should only be considered the first step toward receiving water protection or restoration. A second, more difficult and less recognized step is the periodic update of model inputs and outputs to evaluate potential and necessary changes to the plan. Regardless of the accuracy of the original land use data, it is reasonable to expect a different and perhaps greater degree of land development than initially considered.

In addition, initial assumptions about impervious surface coverage will probably have to be adjusted upward over time. Experience has shown that residents can be expected to erect sheds, widen driveways, construct house additions, and create an impervious surface creep above the levels initially used in the watershed management plan development. Furthermore, land that was expected to remain rural may have experienced urbanization sooner or at a rate faster than initially anticipated, and model updates will have to be performed in order to take the effects of these land use changes into account. Without such updates, an initial model can be completed and its results published and acted on, only to become outdated along with the plan it yielded in a couple of years' time.

Therefore, it is important that sufficient funding also be allocated for future updates to ensure that both the model and the plan remain current and effective. If such funding cannot be provided either in the initial project budget or in subsequent annual plan operating budgets, the model's accuracy and the plan's effectiveness will be diminished.

Gaining Acceptance

Once a watershed management model is developed and a range of implementation options identified, it is important to gain plan acceptance from those who will be impacted by such implementation. This could involve residents, farmers, industries, transportation agencies, and local, county, and regional governments. To accomplish this, there will have to be significant public education and public input activities throughout the watershed management plan's development.

This acceptance is necessary due to the nature of the required implementation measures. While the plan may target a specific industry or sector of the watershed in which change can be achieved through the regulatory process, in many stormwater management situations it will be necessary to change human behavior in order to achieve plan goals. And even where plan implementation can be achieved solely through regulation, it will still be necessary to have public funds allocated to developing, administering, and enforcing such regulations. Approval of such allocation and expenditure will proceed more smoothly and with a greater chance of success if those with an interest or stake in the plan's outcome are informed and involved. If the plan's implementation is to proceed as effectively and efficiently as possible, fundamental questions such as the following must be addressed during its development stage:

- Why is there a problem?
- What is causing it?
- What steps are necessary to correct it?
- How much will it cost?
- What will the outcome be?
- How will I be affected?

The advantage of public involvement throughout plan development is that community expectations can be a significant motivator in getting plan recommendations funded or regulations implemented. Seeking out and involving the plan's stakeholders is the key to simplifying plan approval and implementation.

Total Maximum Daily Load (TMDL)

Total maximum daily loads or TMDLs are tools for implementing state water quality standards and management/restoration goals for a specific water body throughout its watershed. A TMDL is an implementation plan that identifies the allowable loading of a specific pollutant a water body can receive from both point and nonpoint sources without violating state water quality standards. Selection of appropriate TMDLs for a water body is based on the relationship between pollutants from both point and nonpoint sources in the watershed and instream water quality conditions. Unlike technology-based stormwater best management practices (BMPs), which ordinarily do not have specific numeric requirements or performance values, the use of TMDLs provides the basis for state governments to establish specific numeric pollution controls for a water body. And since the development of TMDLs requires an understanding of pollutant sources and loadings throughout the water body's watershed, they also provide states with both the technical and regulatory basis to undertake watershed-based stormwater management.

The TMDL approach has four major features:

- Targeting of priority problems and pollutants;
- Endorsement and ecouragment of high levels of stakeholder involvement;
- Development of integrated solutions that make use of the expertise and authority of multiple agencies; and
- Measurement of success through monitoring and other data gathering.

The TMDL process represents a view of water quality protection that considers watersheds to be the fundamental unit by which to manage water quality.

The TMDL approach to stormwater management has existed for a number of years, having originally been identified in the original 1972 Clean Water Act. For a number of reasons, including the 1999 promulgation of the EPA's Stormwater Phase II Final Rule, the TMDL approach has recently achieved much higher priority in both national and state water quality programs than it had in the past. There are a number of TMDLs that have been established for a range of water bodies around the country that are now serving as templates or models for future ones. The TMDL approach is evolving fairly rapidly, with new guidance information becoming available on a regular basis.

TMDLs are established for impaired water bodies where standard, technology-based stormwater management measures (i.e., BMPs) are not considered capable of correcting the impairment and restoring the water body to required levels. As contained in Section 303(d)(1) of the Clean Water Act and 40 CFR Section 130.7, where technology-based limits or other pollution control requirements (i.e., stormwater BMPs) are not sufficient to achieve compliance with water quality standards, a TMDL must be established.

TMDL determination must also include a margin of safety that, as described in 40 CFR Section 130.7, is intended to address "any lack of knowledge concerning the relationship between effluent limitations and water quality." This margin of safety may be provided in two ways:

- By using conservative assumptions in calculating the loading capacity and wasteload allocations; or
- By establishing wasteload allocations that are lower than the defined loading capacity.

When evaluating the need for a TMDL for a water body, the first step is to determine whether a technology-based approach will be adequate to ensure that water quality standards are met. This determination will typically be based on available data regarding pollutant levels in the water body and a determination of which pollutants exceed water quality standards. Water quality standards can be considered to represent the water body's assimilative capacity, or the amount of pollutant a water body can assimilate without causing or contributing to a violation of water quality standards.

The next step is to allocate the water body's total assimilative capacity for a particular pollutant between point and nonpoint sources of that pollutant in the watershed. This allocation process must take into account natural background loadings and, as discussed above, include a margin of safety to account for any uncertainties. The resultant TMDL for that pollutant is the sum of the nonpoint, point, and background loadings, and a margin of safety as illustrated in the following equation:

$$TMDL = LC = \sum WLA + \sum LA + MOS$$

where:

LC = Loading capacity

WLA = Wasteload allocation (for point sources)

LA = Load allocation (for non-point sources)

MOS = Margin of safety.

TMDLs can be expressed in terms of mass per time, toxicity, or other appropriate measures. The final TMDL is then used to develop numeric discharge permit limitations for point dischargers and pollutant discharge standards for nonpoint sources. The nonpoint discharge standards based upon the TMDL will typically be technology-based, although numeric limitations may be justified in certain instances. The relative pollutant contributions from point and nonpoint sources are a key factor in TMDL development, and their determinations may require a significant data collection and analysis effort.

As with all watershed-based management efforts, the availability of resources is a major factor in TMDL development. The EPA has estimated the costs to implement required TMDLs to range from approximately $1 billion to $4.3 billion per year, depending on the efficiency of TMDLs. Additional EPA cost estimates are summarized below:

- The cost of measures to implement TMDLs for presently identified impaired waters is estimated to be between $900 million and $2.3 billion per year if the problem is approached through the implementation of TMDLs that strictly seek the lowest cost alternatives among all sources of the impairments.
- If the TMDL program was implemented based on an assessment of the reduction needed for the water body and an allocation that includes all sources of impairment, without strict attention to the most cost-effective allocations, these costs would be expected to rise to between $1 billion and $3.4 billion per year.
- In the event that the impaired waters were addressed using a least flexible TMDL scenario, these costs might rise to as high as between $1.9 billion and $4.3 billion per year. In this unlikely scenario, states would simply tighten discharge permits and other national requirements, regardless of the individual contributions of different sources, through a uniform and inflexible approach. This scenario would not benefit from the site-specific tailoring to local conditions that should result from development of a more careful allocation.
- When a moderately cost-effective TMDL program that looks for readily available, cost-effective solutions is used to allocate pollution reduction responsibilities, the costs for both point and nonpoint sources are reduced.
- The nonpoint pollution control measures expected to be implemented under each option would generate some partly offsetting cost savings (e.g., by reducing the frequency of application and the amount of fertilizer used), but these specific savings could not be calculated.

It must be clearly stated that, while it is certainly expensive to address our runoff pollution and water body impairment problems through watershed-based planning and management programs, it will be even more expensive not to follow this approach. The long-term health and well-being of our water resources and, therefore, our society depend on making intelligent decisions and taking effective watershed-based action today.

Regulatory Framework

Once the watershed modeling has been completed and appropriate regulations and requirements developed, a framework for administering these findings must be developed. Presented below are brief discussions of a number of possible regulatory approaches.

Voluntary Compliance

This approach focuses primarily on educating the watershed's population to encourage them to modify behaviors or practices that are causing, contributing to or exacerbating the identified stormwater problems. It may also include cost-sharing assistance if such funding is available. This approach has historically been used to reduce runoff pollution from agricultural lands through changes in farming practices and materials. It has also been used in urban and suburban areas, where residents are asked to reduce their individual pollution contributions by modifying such activities as vehicle washing, hazardous material disposal, lawn care, waste recycling, and litter disposal. Educational measures can include brochures, videos, seminars, demonstrations, group meetings, and other outreach measures and activities.

Permit Requirements for Point Source Discharges

This is a traditional regulatory approach for wastewater discharges that can be adapted to a watershed-based runoff management program. The TMDL program discussed above is an example of how permit limitations for individual point dischargers can be part of a watershed- based approach to runoff management and water resource protection.

This is also an area where pollution trading can provide significant benefits. Where one industry may have great difficulty meeting their discharge requirements, they may trade with another one in the watershed that has excess compliance capacity. This type of cooperation is only possible in a watershed management situation in which benefits of such an approach can be identified and quantified.

Requirements for Land Development

Many states and local authorities have watershed-wide stormwater management requirements for proposed land developments. These requirements can be considered a baseline for development in general, but those general requirements may not be adequate to protect a given resource or watershed receiving system. A general requirement such as an 80 percent reduction in total suspended solids may not provide sufficient protection for a particularly sensitive receiving environment. It may also not provide protection if pollutants other than sediments are a particular concern. For example, a BMP that focuses on capturing sediments may not capture a sufficient level of metals or remove an adequate amount of dissolved nutrients to protect or improve downstream receiving systems.

Since permit requirements based on a watershed management plan are clearly based on a cause/effect approach, they provide a greater certainty that the program goals may be attained. This makes them defensible to those impacted by them.

Source Controls

Another technique that can be used to fulfil the pollutant reduction requirements of a watershed management program is the elimination of the pollutant at its source. An excellent example of this is the banning of phosphorus in laundry detergents in areas tributary to the Chesapeake Bay. Another example from Auckland, New Zealand is based on a roof materials runoff study that identified soluble zinc as a significant pollutant from various roof types. A policy requiring treatment of runoff from such roofs at new development sites quickly led to a shift away from those roof types to more benign roofing materials. In a related way, reducing the extent of new impervious surfaces such as streets, sidewalks, and parking lots can be an effective source control approach to reducing downstream flooding and related runoff quantity impacts.

Development Fees

Under this approach, fees from development and redevelopment projects are collected to provide a funding source for projects and activities identified in the watershed management plan. The approach has been used for many years and still remains a viable implementation option, particularly in watersheds with extensive existing urbanization and associated runoff impacts. It necessitates taking the water quality impairment study to a more refined level where specific projects are identified and await funding. It is important to clearly identify those projects for which the fees will be spent in the watershed management plan to avoid the possibility of the fees being used for other, non-stormwater purposes.

General Discussion

In addition to the approaches discussed above, there are certainly other techniques and approaches that can be used to implement the requirements of a TMDL or watershed management plan. Some or even all of them can be used in combination in a specific watershed. Furthermore, approaches such as source controls may be used at a number of levels that involve regulation of specific products, general population education, and industrial site pollution reduction practices.

If the watershed is already highly urbanized, developer levies may be a significant means of funding existing stormwater system improvements. Retrofit and regulations for redevelopment may include treatment or source controls based on the watershed management plan that help restore site runoff quantity and quality.

Regulation of new development in a relatively undeveloped watershed presents a good opportunity to use the results of a watershed management plan to prevent problems from occurring in the first place. Determining and prioritizing where urban growth can best occur, in conjunction with the protection of existing natural features and aquatic resources such as streams and wetlands, is key to downstream resource protection. This approach is neither pro- nor anti-development; it is based on the concept that better balance between development and environmental interests may be achieved if watershed-specific issues are considered in conjunction with development approaches.

The Future

Clearly, we are following a fairly steep learning curve in developing new approaches to stormwater and aquatic resources issues. We have evolved from a position where runoff was considered the common enemy to the widespread use of runoff treatment practices, source controls, and regional facilities. However, there are still many unanswered questions, and we must maintain our desire and ability to continually strive for improvement. Fortunately, as discussed below, there are a number of recent advances and improvements in our ability to effectively manage urban runoff that bode well for the future.

Availability of More and Better Tools

Computer-based watershed modeling of both runoff quantity and quality is evolving rapidly with excellent recent advances. More programs are now capable of considering stormwater practices in series or performing continuous, long-term rainfall-runoff simulation. For example, recent work at the Cooperative Research Centre for Catchment Hydrology in Australia has produced the Model for Urban Stormwater Improvement Conceptualization or MUSIC. This model provides a flexible tool for watershed modeling, considers stormwater practices in series, and is now being updated to consider whole or life costs for stormwater management practices.

Another computer model that has evolved significantly over the past five years is the Source Loading and Management Model (SLAMM) developed by John Voorhees and Robert Pitt and maintained by USGS. This model was originally developed in the 1970s to gain a better understanding of the relationship between sources of pollutants and runoff quality. It has been continually expanded since and now includes a wide variety of source area, conveyance system, and outfall control practices.

Both of these models are supported by their developers, which is an important consideration in model selection. At this time, both are recommended primarily for planning purposes, but their accuracy and output detail are expected to increase through continued research and development.

Accuracy and Consistency of Data

In addition to improved computer models, it is vital that our store of available data continue to increase and improve. Too often, money can be spent on plan implementation but not on data collection. Every year, more streamflow gages are discontinued, even though the data they provided was extremely valuable. The same applies to water quality and aquatic resource monitoring.

It is also important to have improved consistency in the data that is collected. As noted earlier, the ASCE has made an attempt to provide a protocol for the collection of data related to stormwater practice performance, but similar protocols need to be used for water quality characterization and receiving water evaluation. We will only be able to maximize the value of the data that is collected if we increase its consistency and reliability.

One way to achieve this is to not cut corners on data collection for a specific watershed study. Both rainfall and runoff data must be representative of the entire watershed and not merely portions of it. In data collection, you get what you pay for, and data collected for a specific watershed study must be accurate enough to ensure confidence in the results. There may come a time when data collection needs may be considerably reduced as past experience and new understandings combine to produce new modeling techniques that are less data-dependent. Until that time, however, we have to continue to pursue necessary data collection so that answers may be provided with an acceptable level of confidence.

Linkage of Cause and Effect

If we aggressively implement source control throughout a watershed, as well as all of the stormwater practices that we want to, what will be the impact on downstream water resources? While there are some situations where we know the answer, there are many others where it is unclear and we have to assume we are following the right course.

To a general public that is being asked to fund many different activities, there has to be greater certainty that their taxes will result in a given benefit. Stormwater management has historically been based on an assumption of benefit, but that is not going to be good enough in the future. We have to use case studies of implementation on a watershed basis to evaluate the effectiveness of our activities. This can only be done if implementation throughout a watershed allows us to monitor the results, but we have to recognize that evaluation process as an integral component of the overall watershed management planning process.

It is important that the goals of a watershed management plan be measurable in specific terms. In addition, the measurable improvements should address quantity, quality, ecological and user related improvement. These include such improvements as reductions in sediment load, return of sea grasses, reduction in anoxic zones, increase in abundance and diversity of certain aquatic species, greater recreational opportunities, reduction in flood damages, and/or increased property values. In addition, the achievement of these goals and the terms they are expressed in must have meaning and value, not only to the plan developers and administrators but also to the watershed stakeholders, government leaders, and the general public.

The cost of watershed management plan development and implementation has reached sufficiently high levels to now register on the economic radar screen, and as a result, program developers and administrators are going to be held much more accountable in the future.

Greater Community Recognition

In light of the complexities of watershed management plan development, we are often tempted to operate in relative isolation without much consideration for public input or involvement. However, once a watershed management project is initiated, it is vital that focus groups be established that represent all elements of the community, especially those who will be impacted by the plan's results. As mentioned above, people will support stormwater initiatives in many situations if they understand the purpose of the initiative and the benefits of successful implementation. Public outreach is an essential component of watershed management, and there are numerous guidance documents available on many different websites.

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CHAPTER 13

Maintenance of Stormwater Management Practices

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An important component of any successful urban runoff management program is the effective and efficient maintenance of the stormwater management practices it creates. This chapter presents the key elements of a comprehensive maintenance program for such practices, including both structural facilities and nonstructural measures. Program elements include regulatory aspects, pre-construction planning and design considerations, post-construction inspection and maintenance activities.

As described throughout this book, an effective urban runoff management program requires the successful execution of several steps during a land development project. These steps include:

- Comprehensive project planning to analyze site conditions and identify potential adverse environmental impacts of the project;
- Intelligent and informed design of stormwater management practices that will prevent or minimize these adverse impacts without excessive operation or maintenance demands;
- Competent review of facility and measure designs to ensure compliance with the requirements of the urban runoff management program;
- Proper facility construction and measure implementation according to approved plans and applicable permit conditions; and
- Proper and effective maintenance of facilities and measures following their construction to ensure long-term operation and safety.

Although maintenance is listed as a separate step at the end of the above list, both research and experience have shown that to be truly effective, maintenance considerations must be included in all project steps, starting with the development of the urban runoff program, continuing through the project's design and review phases, and ending with the actual maintenance activities (N.J. Department of Environmental Protection, 1989). As the range of stormwater management practices expands beyond structural facilities to also include a wide variety of nonstructural measures (and, consequently, a similarly expanded range of owners, designers, and maintainers), the need for maintenance awareness throughout the entire land development process has become more important than ever.

Despite the importance of comprehensive stormwater management practice maintenance, several factors can complicate or hinder its performance. One is the authority to perform inspections and enforce maintenance requirements. A second factor is operation and maintenance costs. As we attempt to address a wider range of environmental impacts with stormwater management practices, their complexity grows, resulting in greater and more specialized operation and maintenance demands. A third factor is the inherent institutional difficulties of adequately managing the wide range of available practices through their respective planning, design, construction, and operation phases through a regulatory program.

However complex, the benefits of a comprehensive stormwater management practice maintenance program are substantial. Therefore, the goal of this chapter is to provide information that highlights these benefits and helps overcome the complications. The chapter begins with an overview of key maintenance program elements. It also discusses the particular operation and maintenance challenges posed by nonstructural stormwater management measures, which are becoming an increasingly important component of many urban runoff management programs. The chapter then explores the interrelationship between effective stormwater management practice maintenance and the practice's planning, design, permitting, and construction phases. Finally, it presents various options for funding stormwater management practice maintenance by public entities.

Operation and Maintenance Program Overview

An effective maintenance program for stormwater management practices has a number of key elements. These include:

- Regulations that help ensure that maintenance is addressed from the practice's pre- to post-construction phases;
- Pre-construction planning and design standards that help reduce and facilitate post-construction maintenance;
- A design review and approval process that helps ensure proper application of the program's maintenance-based planning and design standards;
- Construction inspection activities that ensure proper construction in accordance with the practice design;
- Post-construction monitoring and enforcement of maintenance obligations;
- Responsible ownership that recognizes the importance of regular and thorough maintenance;
- Adequate funding of inspection and maintenance activities; and
- Effective and efficient performance of maintenance activities.

Details of each of these elements are discussed below. This discussion highlights the strong interrelationship between all of the elements and the important role they play individually and jointly in achieving safe and effective practice operation and thorough and efficient maintenance.

Regulatory Aspects

A successful urban stormwater management program must contain strong, effective requirements that ensure that the stormwater management practices it creates are adequately maintained. These requirements must have a sound legal basis and pertain to both structural stormwater management facilities and nonstructural stormwater management measures. They must consider all aspects of a facility's or measure's creation, from planning and design through construction to post-construction operation. In doing so, they must address practice owners, designers, construction inspectors, and maintenance personnel. They must also ensure adequate inspection and maintenance funding, effective enforcement, and efficient record keeping. Details of each of these program components are discussed below.

Legal Authority

In order for an urban stormwater management program to effectively address the maintenance of stormwater management practices, it must include written requirements for such maintenance. For example, the Stormwater Management Rules of the New Jersey Department of Environmental Protection (NJDEP), as published in Section 7:8 of the New Jersey Administrative Code (NJAC), require the designers of structural stormwater management practices to consider several maintenance aspects in their design. As stated at NJAC 7:8-5.7-a-2:

> *Structural stormwater management measures shall be designed to minimize maintenance, facilitate maintenance and repairs, and ensure proper functioning.*

NJAC 7:8-5.7-a-3 further states:

> *Structural stormwater management measures shall be designed, constructed, and installed to be strong, durable, and corrosion resistant.*

Finally, at NJAC 7:8-5.8, the following is required:

> *The design engineer shall prepare a maintenance plan for the stormwater management measures incorporated into the design of a major development. The maintenance plan shall contain specific preventative maintenance tasks*

and schedules, cost estimates including the cost of sediment, debris, or trash removal, and the name, address, and telephone number of the person or persons responsible for preventative and corrective maintenance (including replacement).

It is important to note that all program maintenance requirements must receive thorough legal review prior to promulgation. Perhaps the most critical aspect of this review are the legal implications of a program that establishes planning and design standards and, in many instances, oversees construction inspections of stormwater management practices. It must be clear to all involved in the program that, unless otherwise declared, the ultimate responsibility for the safe and proper design, construction, and performance of a stormwater management practice rests with the design and construction professionals who participated in its creation and not with program reviewers and inspectors.

An example of this approach can be found in the NJDEP's Dam Safety Standards as published in Section 7:20 of the state's Administrative Code (NJAC 7:20). As stated at NJAC 7:20-1.4-f:

No action shall be brought against the State or the Department or is agents or employees for the recovery of damages caused by the partial or total failure of any dam or reservoir or through the operation of any dam or reservoir upon the grounds that the Department is liable by virtue of any of the following:

1. The approval of the dam or reservoir, or approval of flood handling plans during construction.

2. The issuance or enforcement of orders relative to maintenance or operation of the dam or reservoir.

3. Control, regulation, and inspection of the dam or reservoir.

4. Measures taken to protect against failure during an emergency.

Structural and Nonstructural Practices

The need for thorough maintenance of structural stormwater management facilities should be self-evident, particularly of those intended to reduce nonpoint source pollution and improve runoff quality through the removal of trash, debris, suspended solids, and harmful chemical and biological agents. It is clear that removing these from the stormwater runoff that passes through a structural facility means that they will consequently be deposited in the facility and that failure to remove them in a timely way can result in outlet blockage, loss of detention storage, and excessive structural loads. Each of these consequences can lead to reduced facility performance and, ultimately, facility failure. The physical character of the structural facility itself, in combination with the consequences of poor maintenance described above, illustrates the importance of thorough maintenance: the visualization of a trash-laden outlet structure or a sediment-filled pond makes it easy to appreciate the importance of an effective maintenance program.

However, as the range of stormwater management practices expands beyond traditional structural facilities to include new nonstructural measures, the importance of maintenance can become less apparent. This is because the reduced physical character of nonstructural stormwater management practices may result in a similarly diminished appreciation of the importance of their maintenance. While it may be easy to visualize how such nonstructural practices as open space preservation, protection of indigenous vegetation, steep slope avoidance, and impervious surface limitations can directly impact the quantity and quality of runoff, the lack of tangible physical attributes of such nonstructural measures may weaken the connection between practice and maintenance that is so readily discernible at structural facilities. As a result, there is the chance that an urban stormwater management program will fail to recognize and impose adequate maintenance requirements upon the program's nonstructural measures. This can significantly diminish the program's overall effectiveness, because in spite of their lack of physical characteristics, nonstructural stormwater management measures also require regular, thorough maintenance, albeit through somewhat nontraditional requirements that reflect their nonstructural character.

Therefore, maintenance of nonstructural practices may require new ways of visualizing stormwater management practice operation and new definitions of maintenance actions. For example, in the case of preserved open space, steep slopes, or groundwater recharge areas, maintenance of these nonstructural practices may mean literally that – maintaining the existence of these areas by preventing their elimination, modification, or abuse. Similarly, the movement of runoff and the filtering and deposition of solids in a vegetated buffer or filter strip may not be as easy to visualize and understand as a wet pond, wetland, or similar structural facility. Nevertheless, these processes do occur, and the resulting accumulation of solids must be addressed through regular, thorough maintenance. While a plot of turf or meadow grasses or a stand of indigenous trees may not have the distinct structural features of a wet pond's permanent pool or a sand filter's sand bed, these nonstructural measures are nonetheless performing pollutant removal functions similar to their structural counterparts and consequently deserve similar maintenance.

There is another important difference between structural facilities and nonstructural practices that can have disturbing consequences for an urban stormwater management program. In addition to the general lack of readily discernable physical features and the need for somewhat nontraditional maintenance actions, nonstructural practices may also differ from structural facilities in both their total number and their location on a land development or redevelopment site. In general, structural facilities are typically located at a centralized location that receives runoff from a significant portion of the development site. This is normally done to minimize construction mobilization costs and to take advantage of the construction efficiency and economy inherent in large facility size. Stated in other terms, it normally requires considerably less land and money to construct a single, somewhat larger stormwater management facility to serve a particular drainage area than to do so with two or more smaller facilities. In addition, a single structural facility typically requires less overall maintenance effort and expense, and since it is a readily visible and recognized stormwater management practice, it is easier to monitor its performance and condition and enforce required maintenance activity.

Furthermore, due to the limited number, regional effectiveness, and centralized location of structural stormwater management facilities, their maintenance is typically the responsibility of a limited number of public or private entities such as municipal Public Works Departments or property owners' associations. Such entities typically have sufficient legal, financial, and organizational authority to allow them to not only accept and perform required facility maintenance but to allow others to effectively bring enforcement actions against them if they fail to meet their maintenance obligations.

However, it is not uncommon for numerous nonstructural stormwater management measures to be distributed throughout a development site, with each one receiving and treating runoff from only small portions of the overall site. This happens for a number of reasons, the most notable being the fundamental intent or goal of nonstructural stormwater management. As described in detail in Chapter 8, Impact Avoidance, the intent of nonstructural stormwater management is not to respond to the runoff produced by a development site the way structural practices do, but instead to intervene in the rainfall-runoff process in order to minimize the amount of runoff and associated impacts produced by the development. Stated in ideal terms, development site runoff responds to nonstructural practices, while structural practices respond to site runoff. To achieve this, however, nonstructural practices must generally be distributed throughout the site in order to optimally intervene in the rainfall-runoff process.

Other factors can also contribute to the number of nonstructural practices at a land development site being larger than the number of structural ones. These include the physical, chemical, and biological processes that must occur to effectively treat and convey stormwater runoff. Due to their relatively low design depths, widths, and/or heights, nonstructural stormwater management measures can typically manage only relatively small rates and volumes of runoff when compared to structural facilities. As a result, the relative size of their tributary drainage areas must also be small, requiring a greater number of nonstructural measures throughout the development site.

Finally, there is a relationship between nonstructural measure character, size, and efficiency that further promotes the use of more rather than fewer nonstructural measures at a land development site. Since, in addition to their smaller overall size, they have significantly less height, depth, and other distinct physical characteristics, nonstructural measures can more readily be located in the rear, side, and even front yard setback areas on individual development lots. For example, vegetated buffers, preserved open space areas, and pervious areas downstream of unconnected impervious surfaces can easily be located in setback areas. They can also serve as both active and passive open space areas. Since setback and open space areas are typically required at land development sites, using them to also locate nonstructural stormwater management measures can increase site utilization efficiency and even reduce overall site disturbance. It can also reduce or, in certain instances, even eliminate the need for a larger, centralized structural facility, which is typically too large to fit within required setback areas and often prohibited in required open space areas.

As a result, nonstructural stormwater management measures can be located on numerous individual lots throughout a land development site. While this conforms to nonstructural stormwater management principles and can increase land utilization efficiency and preserve open space, the maintenance implications can be troubling. For unlike a limited number of centralized structural facilities, a widely dispersed array of nonstructural measures will involve a similarly wide range of individual property owners, each with a different level of interest, ability, and resources to perform required measure maintenance. In addition, maintenance monitoring and inspection by regulatory agencies will be more difficult due to the greater number and dispersed locations of the nonstructural measures. Enforcement of maintenance requirements may also be more difficult due to the direct responsibility of individual property owners rather than a single, representative owners' association. Record keeping and other administrative functions can also be expected to be more complex and costly.

Therefore, it is important for an urban stormwater management program to recognize the maintenance challenges posed by its nonstructural stormwater management component and take appropriate steps to address them. This should include the following:

- Recognize the increased complexity and nontraditional character of nonstructural stormwater management measure maintenance.
- Identify the potential for and consequences of maintenance neglect and measure modification and elimination by private property owners.
- Review available maintenance inspection and enforcement options against such property owners.
- Include in the urban runoff management program only those nonstructural measures that the program's administrators can reasonably guarantee will remain functional in the future.
- Develop a property owner education program on nonstructural measure purpose, operation, and maintenance.
- Adopt appropriate maintenance inspection and enforcement measures.

Design Review

The success of both structural and nonstructural stormwater management practices, including their operation and maintenance, will depend to a great extent on the scope, accuracy, and basis of the planning and design standards used to create them. However, to be effective, such standards must be incorporated into the urban stormwater management program. In addition, it must be ascertained prior to its construction that a stormwater management practice has been designed in accordance with them. As a result, a design review and approval process must also be included in the urban stormwater management program. Such a process can be integrated into existing development review programs such as those conducted by planning boards, boards of adjustment, regional or state agencies, and sewer and water utilities. Lacking such existing programs, a new program must be developed with proper legal authority and appropriate submission, review, and approval requirements and procedures. In such cases, it may be possible to reach an agreement with an existing program at another level of jurisdiction to share or trade off actual project reviews in order to save time, effort, and expense and improve review coordination between agencies.

Construction Inspection

The success of both structural and nonstructural stormwater management practices also depends upon the accuracy and quality of their construction. Similar to design review, inspection of the construction is therefore essential to achieving both individual measure and overall program success. An effective construction inspection program includes:

- A sufficient number of adequately trained and experienced inspectors;
- Inspection standards and procedures for all phases and aspects of facility or measure construction including materials, dimensions, strengths, and construction equipment and practices;
- Pre-construction meetings to review inspection procedures and construction requirements prior to the start of construction;

- Periodic construction meetings to review progress, address problems, and anticipate and avoid future difficulties; and
- Post-construction documentation including the development, review, approval, and recording of as-built drawings.

Post-Construction Monitoring and Enforcement

Once construction is complete and the stormwater management practices are put into operation, monitoring of their maintenance must be performed to ensure compliance with the maintenance requirements contained in the urban runoff management program's regulations and/or a specific maintenance plan. Such monitoring can be performed directly by the urban stormwater management program agency or the practice owner. Monitoring by the practice owner should include the submission of monitoring reports to the program agency at least once per year. In addition, the program must also have provisions for noncompliance. Such provisions can include, in reverse order of severity:

- Informal, discretionary procedures to deal with isolated or inadvertent maintenance noncompliance;
- Formal, prescribed procedures and measures to address chronic or intentional noncompliance;
- Emergency measures to respond to noncompliance matters that pose an immediate health or safety threat; and
- Maintenance assumption in the case of total maintenance default and abandonment.

Finally, successful post-construction monitoring includes provisions for legal access to the stormwater management practice by program personnel through easements, right-of-ways, and access and inspection agreements with the practice owner. Bonds, letters of credit, and other financial instruments can also be required from the owner to finance emergency measures and overall maintenance assumption by the program agency.

Interagency Coordination

With the promulgation of the EPA's Stormwater Phase II Final Rule, municipal, county, and state governments throughout the country are developing new or upgrading existing stormwater management programs in order to comply with their Phase II Stormwater permits. Under such conditions, it is important that these various levels of government coordinate their efforts to maximize consistency and minimize conflicts between the various programs, including their maintenance components. This can perhaps best be achieved through a hierarchical approach that recognizes both the role each level of government should play in managing urban runoff and the relative proximity each level has to the actual stormwater management practices that must be properly maintained.

This approach can begin at the state program level with language that both mandates proper stormwater management practice maintenance and establishes general or minimum requirements to ensure it is achieved. Such requirements can include the need to design and construct stormwater management practices that require the least practical maintenance effort and cost, as well as the need to prepare a maintenance plan that details the actual maintenance tasks and equipment necessary to perform them. These general requirements can also specify what types of entities can and cannot be assigned maintenance responsibility and establish general record keeping and reporting requirements. Finally, it is important that, having established general maintenance standards and requirements, state and other higher levels of government recognize that those at the municipal and county level will have a more direct physical and regulatory relationship with the actual stormwater management practices and their owners. This recognition should come in the form of state program language that allows municipalities, counties, and other local government entities to both establish more specific maintenance standards and requirements and to decide the optimal procedures for implementing them. As noted in the 1997 *Operation, Maintenance, and Management of Stormwater Management Systems* (Watershed Management Institute, 1997), a key to a successful stormwater management practice maintenance program is providing the flexibility to attain maintenance standards within the institutional framework of the overall stormwater management program, whether at the state, regional, county, and/or municipal level.

For their part, local government entities must recognize that the general or minimum maintenance standards established for them by higher levels of government only represent the framework of an effective maintenance program. This recognition should then prompt the development and refinement of more specific maintenance standards, procedures, and guarantees that address local regulatory, physical, political, economic, and social conditions. This process begins with the identification of these conditions and the development and promulgation of specific maintenance standards for the various types of projects and stormwater management practices expected within a given jurisdiction.

Summary

The above section on the regulatory aspects of stormwater management practice maintenance presented the following ideas and information:

- To be successful, an urban stormwater management program must include provisions for effective maintenance of stormwater management practices.
- These maintenance provisions should address all applicable types of stormwater management practices, including structural facilities and nonstructural measures. Maintenance of nonstructural stormwater management measures poses unique challenges for an urban stormwater management program.
- The program's maintenance provisions must also encompass all phases of a stormwater management practice's development, from planning and design to construction and, ultimately, operation and maintenance. To do so, the program should include design review, construction inspection, and maintenance inspection, enforcement, and default procedures.
- The maintenance provisions must have a sound legal basis that allows the program to both impose maintenance requirements and check for compliance.
- Interagency coordination of maintenance standards will help avoid conflicts and duplication.

Planning and Design Considerations

It is self-evident that the efforts of planners, designers, and reviewers of stormwater management practices will directly affect the runoff performance of these practices. However, the efforts of these individuals can also have a direct effect on the amount, frequency, and difficulty of required practice maintenance. Research into the maintenance aspects of more than 50 structural stormwater management facilities in New Jersey indicated that approximately two thirds of the maintenance problems encountered at these facilities were at least partly due to shortcomings in the planning, design, and review process (N.J. Department of Environmental Protection, 1989). These shortcomings included:

- Inadequate planning and design standards in the urban runoff management program;
- Inadequate investigation and analysis of facility site conditions;
- Inadequate understanding of facility function and operational needs;
- Inattentive or inept design and design review; and
- Lack of consideration for facility maintenance needs.

The results of these shortcomings included increased maintenance complexity, effort, and cost, reduced facility performance, and decreased facility safety (Watershed Management Institute, 1997). According to the New Jersey research, some of the resultant maintenance problems "were virtually unsolvable without massive infusions of time, money, and hard work."

Fortunately, enlightened and focused planning, design, and review requirements and procedures can eliminate these shortcomings and actually improve maintenance effectiveness and efficiency. This, in turn, can lead to high levels of long-term practice performance and safety. It is therefore important that an urban stormwater management program require planners, designers, and reviewers to include maintenance as a key consideration in their efforts. In addition, the program should provide them with maintenance-based planning and design standards that help achieve the favorable program results described above.

In general, planning and design standards that help minimize and facilitate stormwater management practice maintenance typically include the following consideration for four important aspects: durability, constructability, maintainability, and accessibility. Discussions of each is presented below.

Durability

The required use of strong, durable materials, appurtenances, and fasteners can greatly reduce the maintenance required at a structural stormwater management facility. These long-term savings typically exceed the one-time expense of providing higher quality products, which justifies their inclusion in the program's maintenance standards. Durability extends across the entire range of facility components, from concrete outlet structures to vegetative covers and landscaping.

Durability

BAD: Inappropriate materials and poor construction increases maintenance effort and cost.

GOOD: Durable materials and sound construction decreases them.

Constructability

It must be remembered that a stormwater management practice must be properly constructed before it can produce any long-term runoff management benefits with reasonable levels of inspection and maintenance. This high degree of construction quality requires skilled, experienced, and properly equipped constructors and attentive and knowledgeable inspectors. However, achieving high quality construction begins with the creation of stormwater management practices at the planning and design levels that reflect the realities of construction. These realities require that a practice possess a reasonable degree of simplicity, standardization, and component availability. Required materials and equipment should be readily available and construction techniques safe and feasible. Construction plans and specifications, which are the constructors' and inspectors' instruction manual, must be clear, concise, and informative. They must contain all necessary information in a format and form that assists rather than hinders use in the field under all weather conditions. This is not meant to stifle creativity and imagination in the selection and design of a particular stormwater management practice. However, practices or components that require particularly new or complex construction materials, techniques, equipment, or sequencing must be given additional attention in the construction documents in the form of extra detailing, notes, warnings, and references.

Constructability

To help ensure sound construction, required construction materials and procedures should be as standard as practical. Unique procedures and complex components should be thoroughly described and detailed in construction documents.

Maintainability

Throughout the planning, design, and review process, every attempt should be made to both minimize and facilitate required maintenance. This approach must guide a wide range of decisions, from the type of selected stormwater management practice to its location, configuration, materials, and the techniques and equipment required to both construct and maintain it. The questions governing these decisions include:

- Is the type of selected stormwater management practice and its various components compatible with the physical conditions and constraints of its location?
- Are the selected materials durable? Are they reasonably available for both construction and replacement purposes?
- Are proposed slopes steep enough to promote proper drainage but flat enough to permit safe access and mobility of inspection and maintenance personnel?
- Can required levels of construction quality be reasonably achieved?
- Are the amount, cost, and complexity of required maintenance within the owner's ability to provide it?

Under optimum planning, design, and review conditions, all of the above questions will be answered affirmatively before the design of the stormwater management practice is completed and approved.

Accessibility

According to the New Jersey stormwater management facility maintenance research cited earlier (NJDEP, 1989), lack of accessibility was a major hindrance to stormwater management practice maintenance, with the access to approximately one third of practice components inspected in the field considered inadequate and, at times, unsafe. Lack of safe, adequate access can quickly defeat all planning, design, and review efforts to provide durable, constructable, and maintainable stormwater management practices as well as an owner's efforts to train, equip, fund, and motivate maintenance personnel. In other words, small oversights regarding access can create large maintenance problems. Personnel access to a stormwater management practice and its various components must include not only the personnel themselves, but their equipment and materials as well. Access can range from legal access through an easement or right-of-way to a stormwater management practice to physical access to the interior of its outlet structure through a hatch and ladder rungs.

Efforts to facilitate access and enhance safety during the planning, design, and review phases can often yield significant savings in subsequent inspection and maintenance efforts. For example, a maintenance inspector conducting a post-storm inspection of several stormwater management facilities for excessive debris build-up can complete their task more efficiently if a facility is visible from a road, driveway, or other location accessible by their vehicle. Similarly, the cost to remove

Maintainability

BAD: Lack of adequate bottom slope in dry detention basin causes unintended ponding that prevents mowing and cleaning.

GOOD: Adequate bottom slope in dry detention basin creates intended dry bottom that can be regularly mowed and cleaned.

any debris noted during the inspection can be reduced if the maintenance personnel have ready access to it. Similar to durability, the long-term savings achieved through enhanced accessibility typically exceed the one-time expense of providing it.

Another important but less noted aspect of stormwater management practice accessibility is how readily visible a particular practice is to people other than maintenance personnel. According to the New Jersey stormwater management facility maintenance research cited earlier, structural stormwater management facilities that were readily visible to pedestrians, motorists, customers, employees, and others not responsible for facility maintenance were more than twice as likely to receive high levels of maintenance than less visible ones. Less visible facilities were in turn three times more likely to receive fair to poor maintenance. This leads to the conclusion that visual accessibility may be equal in importance to physical and legal access for maintenance purposes.

Finally, attempts to minimize and facilitate stormwater management practice maintenance during the planning, design, and review phases can be aided by a series of questions that planners, designers, and reviewers should pose to themselves and each other. These questions include:

Who will perform the maintenance?

Will specialists be required for some or all of the maintenance or can it be performed by someone with general maintenance skills and equipment? The person or agency that will actually be performing the required maintenance must be identified with sufficient accuracy during the planning, design, and review phases so that their level of ability, equipment, and expertise can be taken into consideration.

What maintenance must be performed?

Each type of stormwater management practice requires specific and, at times, unique maintenance tasks. These tasks should be identified prior to final practice selection so that planners, designers, and reviewers can ensure that they correspond to the abilities and equipment of the designated maintainers. In addition, preparing a list of all required maintenance tasks may prompt a redesign that produces a shorter task list.

Accessibility

Lack of access can defeat the best maintenance program requirements and intentions.

BAD: Lack of depressed curb hinders access to practice by maintenance personnel and equipment.

GOOD: Readily accessible practices are easier and cheaper to maintain.

When will maintenance be required?

Once a day, week, month, or year? Recurring maintenance costs can be substantial over the life of the practice. In addition, certain stormwater management practices and their components require maintenance at specific times of year or only under certain weather conditions. For example, the turf grass in an extended detention basin or grass filter strip can only be moved in dry weather. How will maintenance and operation be affected if prolonged periods of wet weather are common? Finally, are emergency repairs or debris removal possible during a storm event, perhaps during nighttime hours? Addressing such questions during the planning, design, and review phases will be easier than during an actual post-construction emergency and can produce more appropriate practice selections and improved practice designs.

Where will maintenance be required?

Will maintenance personnel be able to get to the area or component that requires maintenance, along with their equipment and materials? Once there, will they have a safe, stable place to work in? In addition, where will the sediment, debris, and other material removed from the practice be disposed? This question becomes more critical when the character of the removed material (such as toxic or hazardous materials) affects the disposal location. Once again, addressing these questions during the planning, design, and review phases will be easier than during the first cleanout effort.

How will maintenance be performed? What equipment, training, and/or materials will be necessary? Will any safety equipment or procedures be necessary? Is a certain maintenance task exceptionally difficult, dangerous, and/or expensive? Can such conditions be eliminated through additional design effort or through selection of a different stormwater management practice?

All of the above questions are intended to make planners, designers, and reviewers more aware of maintenance tasks, schedules, costs, and problems and to encourage them to address these issues during the planning, design, and review phases of the practice. The goal of minimum maintenance cannot be achieved without doing so.

Summary

The above section on planning and design standards presented the following ideas and information:

- The efforts of planners, designers, and reviewers can have a direct effect on the amount, frequency, cost, and complexity of maintenance required at a stormwater management practice.
- As a result, a successful urban runoff management program must include planning, design, and review requirements that minimize and facilitate maintenance and provide specific guidance on how to achieve it.

Accessibility

GOOD AND BAD: Ladder rungs allow access to structure bottom, but large grating openings are hazardous to maintenance personnel.

GOOD: Lightweight, noncorroding aluminum top gratings are safe to stand on and easy to lift.

- Durability, maintainability, constructability, and accessibility are key planning and design considerations.
- Who, what, where, when, and how maintenance will be conducted are key questions that planners, designers, and reviewers can ask themselves and each other.

Public Maintenance Financing

Regardless of the combined efforts of regulators, planners, designers, reviewers, constructors, and inspectors, successful stormwater management practice maintenance cannot be achieved without adequate funding of required maintenance activities. This funding is not only needed for the direct costs of performing required maintenance tasks, but also to meet the costs of equipment, training, disposal, record keeping, and administration. When maintenance is financed and performed by a private entity such as a property or building owner, there is typically a wider range of available funding sources than if the maintenance is publicly financed. Private funding sources may include rental and lease incomes, tenant fees, service charges, or incorporating maintenance costs into company overhead, product prices, and/or service rates.

Maintenance financing by a public entity such as a municipal or county government can be more difficult, typically due to fewer funding sources, funding competition from other government activities, and the need to secure funding a year or more in advance through the government's budgeting process. This section will review these difficulties and explore ways in which they can be avoided and maintenance funding maximized.

Before reviewing potential funding sources, it is helpful to look at some of the reasons why a public entity would assume the responsibility for stormwater management practice maintenance and its financing. Most obvious is the case where the public entity is required to construct or implement a stormwater management practice to serve its own properties, roadways, buildings, and other public facilities. Such cases are expected to become more numerous with the arrival of the EPA's Phase II Stormwater Rules and associated NPDES permits, which require public entities to implement practices at new public facilities that disturb an acre or more of land.

In addition, a public entity such as a municipality or county may choose to assume the maintenance of a stormwater management practice at a privately-owned project or development in order to ensure it receives adequate care. While most public entities prefer that such maintenance remain in the responsibility of the practice's private owner, research has shown that the level and quality of private maintenance in many instances can be inadequate. As a result, a local government with overall responsibility for public health and safety as well as specific NPDES permit obligations may decide that the best way to meet these obligations and responsibilities is to perform the required maintenance itself. In other instances, a public entity may have been forced to assume maintenance responsibility of a private stormwater management practice due to the owner's failure to adequately perform it. Such assumption typically occurs some time after the practice's construction.

Whatever the reasons, once a public entity becomes responsible for stormwater management practice maintenance, it must develop and implement a program to finance the required maintenance activities. This is true whether the public entity performs the maintenance itself or hires an outside company or agency to do it. There are some general characteristics of a successful maintenance financing program that warrant special consideration (Livingston et al., 1997). These characteristics are summarized below:

- The success of any public financing program is determined in part by the amount and quality of program information provided to the public. This information must explain the purpose of and need for the stormwater management practice maintenance activities as well as sufficient details of the financing program. The information must be able to convince the public and their elected officials that it is in their interests to adequately fund the public entity's stormwater management practice maintenance activities.
- A public financing program should be based upon a stable, reliable source of funds. Stormwater management practice maintenance is a long-term activity that requires a funding source that will remain viable throughout the life of the maintenance program.

- Whenever possible, a public financing program should fit readily into the billing, collection, and bookkeeping operations of the public entity's existing financial system.
- A public financing program should include provisions not only for the actual maintenance activities but also for record keeping, accounting, and other administrative tasks.
- The fee or rate structure for a public financing program should be equitable, readily understandable, and defensible. It must be perceived by the public as being fair, reasonable, and based upon accurate information and sound decisions.
- In addition, the fee or rate structure should be flexible enough to allow both regular and emergency updating to address changes in maintenance program scope, schedule, and costs.
- Finally, a public financing program must be consistent with all applicable laws and regulations. To ensure such consistency, the program must be reviewed by legal counsel prior to its implementation.

In general, there are three funding sources typically available for public stormwater management practice maintenance (NJDEP, 1989) activities:

1. General Tax Revenues
2. Dedicated Contributions
3. Stormwater Utility Fees

Details of each funding source are presented below, including suggested criteria for evaluating the suitability of each for a particular public entity.

General Tax Revenues

General tax revenues are an obvious source of funding for public maintenance of stormwater management practices. Since taxes are raised to provide for a community's health, safety, and welfare as well as to meet its legal obligations, it can be shown that failure to provide adequate stormwater management practice maintenance can threaten these important objectives. As a result, the use of general tax revenues remains a popular source of funding for stormwater management practice maintenance. To obtain this funding, however, requires preparation of annual maintenance program budgets based upon forecasts or predictions of future maintenance obligations and costs. Low forecasts can lead to budget shortfalls that can prevent the performance of all required maintenance activities, while excessively high forecasts can hinder efforts to secure necessary funding.

Other aspects of the budgeting process can complicate the use of general tax revenues to fund public stormwater management practice maintenance. As part of a government's overall operating budget, stormwater management practice maintenance must compete for funding with all other government operations included in the budget, including police, fire, sanitation, and administrative services. It is in such competitive situations that the value of an effective public information program regarding urban stormwater management noted above becomes apparent. The legal obligation to comply with the maintenance requirements of a municipality's or county's NPDES permit can also give the maintenance program added importance during the budgeting process.

Nevertheless, several other difficulties may exist. First, it may be difficult to justify the use of general tax revenues from the entire community to maintain a stormwater management practice that only directly benefits a portion of that community. Second, the need to provide funding for unforeseen, emergency, or other one-time non-stormwater events occurring in the community may result in the diversion of normal, expected funding away from the stormwater management practice maintenance program. Finally, in light of the public's traditional resistance to tax increases, which can manifest itself at times in the adoption of tax increase caps and even tax cuts, it may be difficult to obtain required funding increases necessitated by increased maintenance costs. As a result, general tax revenues can be both the most readily available and least stable source of maintenance program funding. This realization has led to development of the alternative approaches described below.

Dedicated Contributions

The use of dedicated contributions to finance public maintenance of stormwater management practices is based upon the principle that those creating the need for the stormwater management practice and its maintenance should bear the cost. It applies to stormwater management programs in which a public entity assumes the maintenance responsibility for a stormwater management practice that has been created to serve a privately-owned land development or project. In exchange for the maintenance responsibility, the developer makes a one-time contribution to the public entity to fund the long-term required maintenance. This contribution is then periodically drawn upon by the public entity as the maintenance is performed. As a guarantee, the contribution is typically made prior to final approval of the development. It is placed in a dedicated account that can only be used to finance maintenance of that particular stormwater management practice. Accurate bookkeeping practices must be followed to ensure appropriate use of the funds.

In one sense, the use of dedicated contributions to finance public stormwater management practice maintenance can be considered an extension of the permit or inspection fees traditionally charged by local governments or other public entities to review and/or inspect the construction of a privately-owned development, building, or other project. In this case, the permit or inspection fee is used to offset the administrative and inspection costs incurred by the public entity. The dedicated contribution system extends this concept by applying the "fee" paid in the form of a dedicated contribution 1) only to a specific project or practice and 2) over an extended period of time. The application to a specific project requires, as mentioned above, a dedicated account in which to deposit the contribution and track withdrawals, while use of the contribution over an extended period of time requires consideration of both interest earnings and cost increases.

One key to a successful dedicated contribution financing system is an accurate method for estimating the long-term maintenance costs and then converting that amount into an equivalent one-time payment. Factors that should be considered when estimating the payment include:

- The type and maintenance needs of the specific stormwater management practice to be maintained, including the type, size, and location of the practice as well as the characteristics of the runoff it will receive;
- The number of years that maintenance must be provided;
- The present annual costs of practice maintenance, including maintenance activities, equipment repair and replacement, materials, insurance, record keeping, and other administrative tasks;
- Anticipated maintenance cost increases due to increases in salaries, overhead, materials and equipment costs, insurance premiums, and disposal costs; and
- The anticipated interest rate earned by the contribution over the life of the maintenance financing.

The use of dedicated contributions to finance stormwater management practice maintenance has many advantages (Livingston et al., 1997). The most important one may be that it provides a secure and stable maintenance funding source if properly managed. In addition, the source of the maintenance funding can be directly linked to the need for maintenance, eliminating the need to justify the expenditure of general revenues on a particular facility or area. Difficulties include the need for accurate estimates of annual maintenance costs which, in turn, require similarly accurate estimates of the required time, materials, and equipment. Administrative and insurance costs must also be accurately estimated along with potential cost increases for all aspects of the maintenance program. The duration of the maintenance program and the interest that may be earned on the one-time contribution during this period can also be difficult to accurately estimate. Typically, conservative estimates are used in order to provide a safety factor. The actual computation of the one-time contribution is based upon standard economic principles for capital recovery through a series of payments (Grant and Ireson, 1960). Formulas can be found in standard economics textbooks, particularly those that deal with the principles of engineering economics.

A final difficulty with the use of dedicated contributions is the fact that they are only directly applicable to the maintenance of new, privately-owned and financed stormwater management practices. They cannot be readily used to finance the public maintenance of new, publicly-owned measures or any existing public maintenance activities without special considerations and conditions. Consequently, full public financing of stormwater management practice maintenance may require the combined use of general tax revenues for publicly-owned practices and dedicated contributions for privately-owned ones.

Stormwater Utility Fees

The uncertainties associated with the use of general tax revenues to finance stormwater management practice maintenance has recently led many communities to create a specialized agency known as a stormwater utility. This agency is assigned responsibility for stormwater management practice maintenance within its jurisdictional area. To finance this maintenance, the utility is allowed to charge property owners within it jurisdiction a fee or other assessment. The amount of the fee is typically related to the property's stormwater impacts and, consequently, its dependence on a well-maintained stormwater management practice. In some instances, the utility's responsibilities may also include storm sewer construction and maintenance, waterway stewardship, and other drainage, erosion, and flood control activities.

The use of utility fees to finance publicly-owned water and sewerage systems began in the early 1990s, and they continue to be a stable source of funding for such systems. Over the last decade, the use of this public financing technique has been extended to the operation and maintenance of stormwater management systems. Once the utility has been established, it offers many advantages over other public financing sources. It can provide maintenance funds for both existing and proposed stormwater management practices. It does not have to compete with other government programs and needs. Moreover, the relationship between the fees for stormwater management practice maintenance and the benefits of performing it is more obvious in this approach. However, first the utility must be created and an equitable fee structure established. This entails the legal and physical establishment of an entirely new entity with sufficient staff and resources to properly function. This requirement can pose the greatest obstacle to the use of utility fees to finance public maintenance of stormwater management practices.

The utility rate structure must be based on several considerations (Livingston et al., 1997). It must, of course, reflect the costs of providing the stormwater management practice maintenance and other services for which the utility was established. But first and perhaps foremost is the premise that the fee is based upon the need for the stormwater management practice maintenance rather than the benefits provided by it. As a result, the fee structure can be based upon characteristics of the assessed properties that influence the volume of runoff they produce, such as their total area, the area of their impervious surfaces, or the type of land use. However, the fee structure should also remain as simple as possible in order to facilitate understanding of and, as a result, acceptance by those paying it. Simplicity will also facilitate utility administration and implementing future rate changes.

Summary

The above section on the public financing of stormwater management practice maintenance presented the following ideas and information:

- Successful implementation of a public stormwater management practice maintenance program requires adequate and stable funding sources.
- The reasons why a public entity would assume the maintenance of a stormwater management practice include direct ownership of the practice, the need or desire to have direct control over a privately-owned practice, and maintenance default by the owner.
- There are three general sources of funds for public stormwater management practice maintenance: 1) general tax revenues, 2) dedicated contributions, and 3) utility charges.
- The use of these financing techniques involves legal, financial, and economic considerations that must be thoroughly addressed before such use can begin.

Summary and Conclusions

This chapter demonstrates the importance of stormwater management practice maintenance and describes key features of an effective maintenance program. Such features include:

1. Legal authority to require and enforce stormwater management practice maintenance;
2. Planning and design standards that minimize and facilitate maintenance;
3. Design review procedures to ensure compliance with these standards;
4. Construction inspection procedures to ensure that the practice is being constructed in accordance with the design plans;
5. Post-construction monitoring to ensure proper maintenance is being conducted;
6. Recognition of the unique maintenance needs of nonstructural stormwater management practices; and
7. Ensuring that adequate and stable funding is available for maintenance.

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