

# Fundamentals of Urban Runoff Management

## TECHNICAL AND INSTITUTIONAL ISSUES

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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.

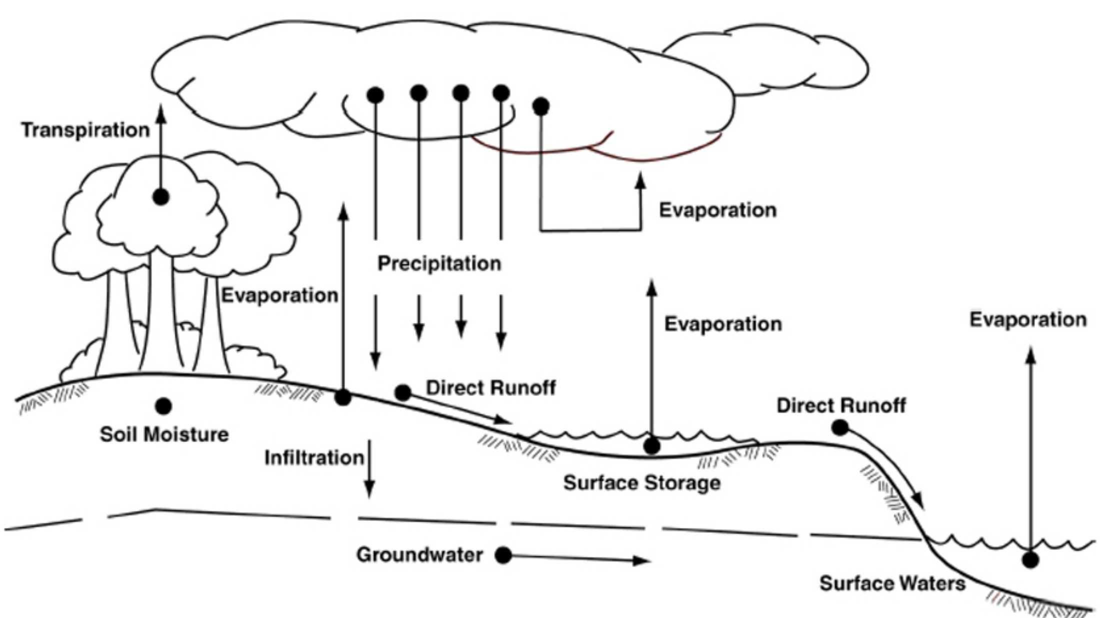
Finally, the role of urban runoff management program developers should not be overlooked in the above. That's because the theories, equations, models, and input parameter values they choose to incorporate into their programs will influence and even require designers to follow certain procedures, include 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.

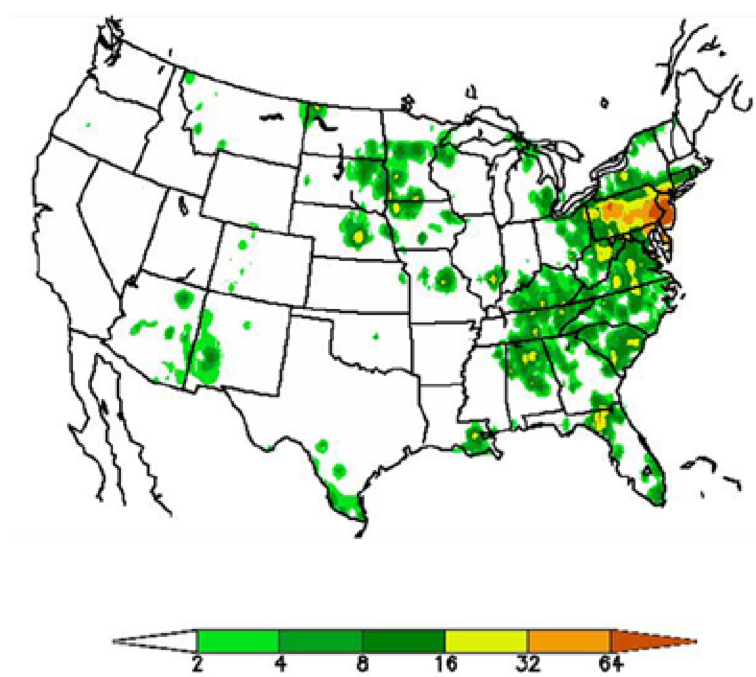
In general, our interest in rainfall not only focuses on real and hypothetical events, but also on both small and large rainfall amounts. From statistical analyses and experience, we know that small rainfalls occur much more frequently than large ones. As such, relatively small rainfalls are typically 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**

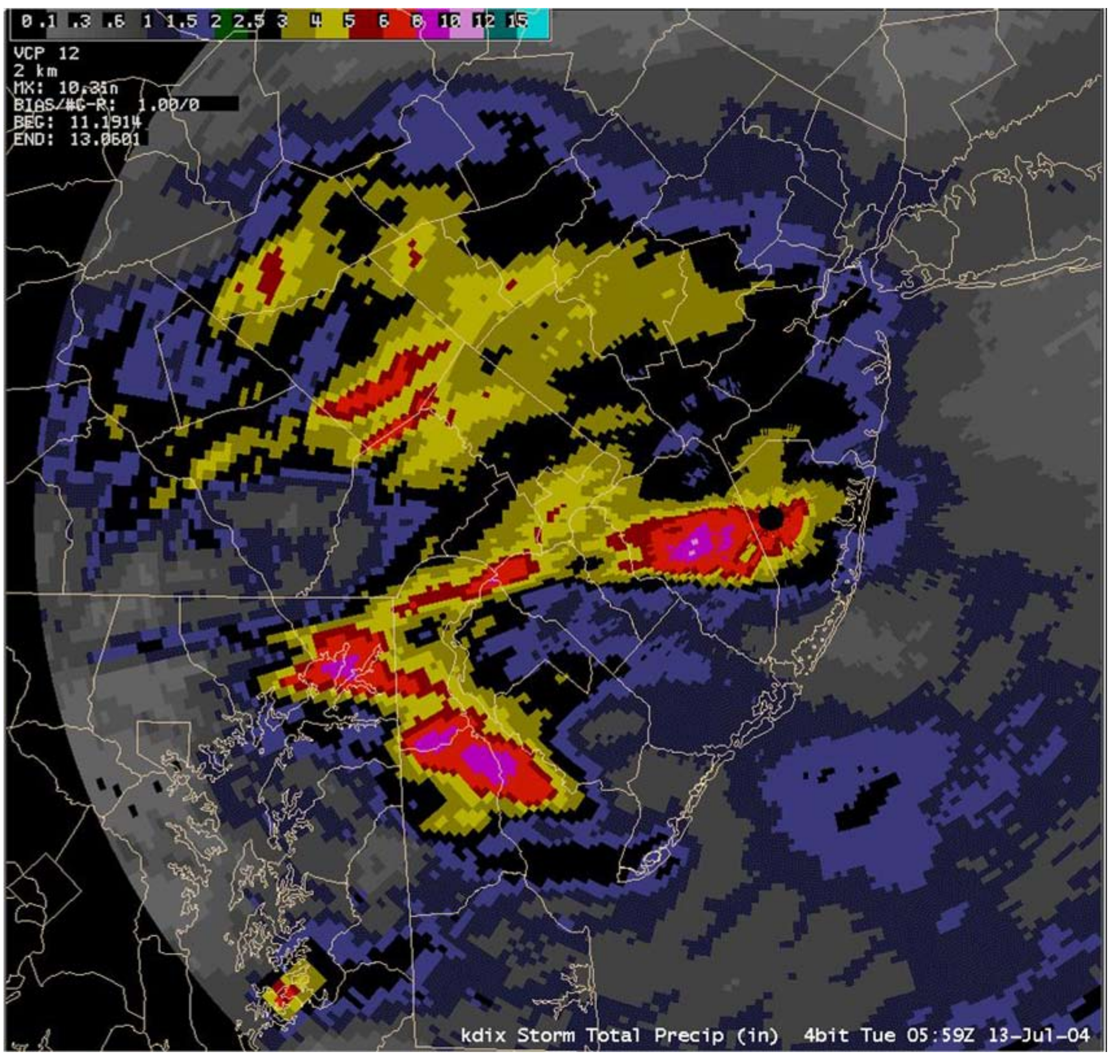


common rain events can also be used in the analysis and design of certain hydraulic structures. As described above, such rainfalls are not typically associated with structures intended to withstand the effects of a very large, rare rainfall event, such as a dam's spillway. Instead, they would be intended 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.

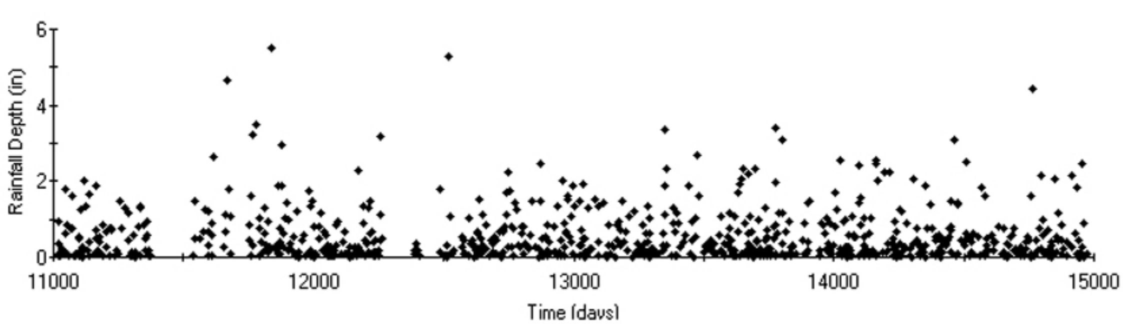
Despite this enhanced robustness or accuracy and its applicability to a range of analytic and design problems, the use of actual rainfall data, either from single, extreme events or over long time periods, is not without its problems. First and foremost is the availability of such data. While the number 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

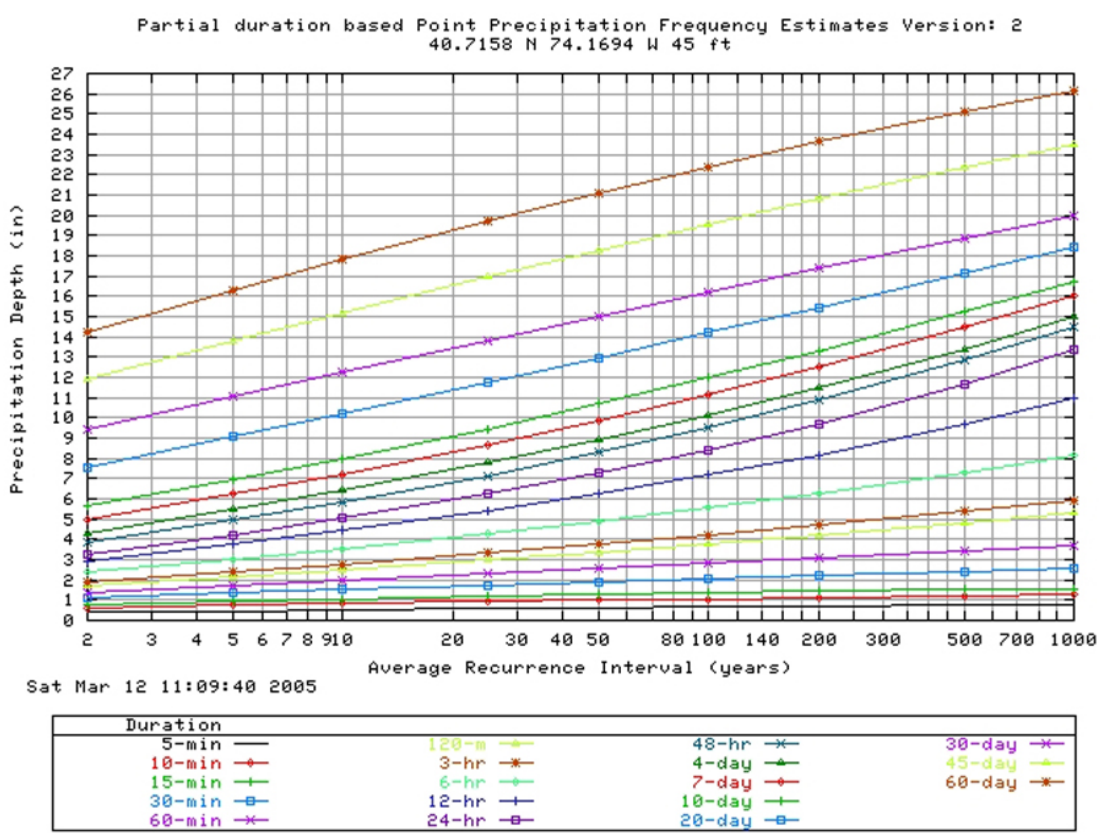
speeds of modern computers. If any computer-related problems remain in this area, it may be in the relatively limited number of computer programs that can accept long-term rainfall data.

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).

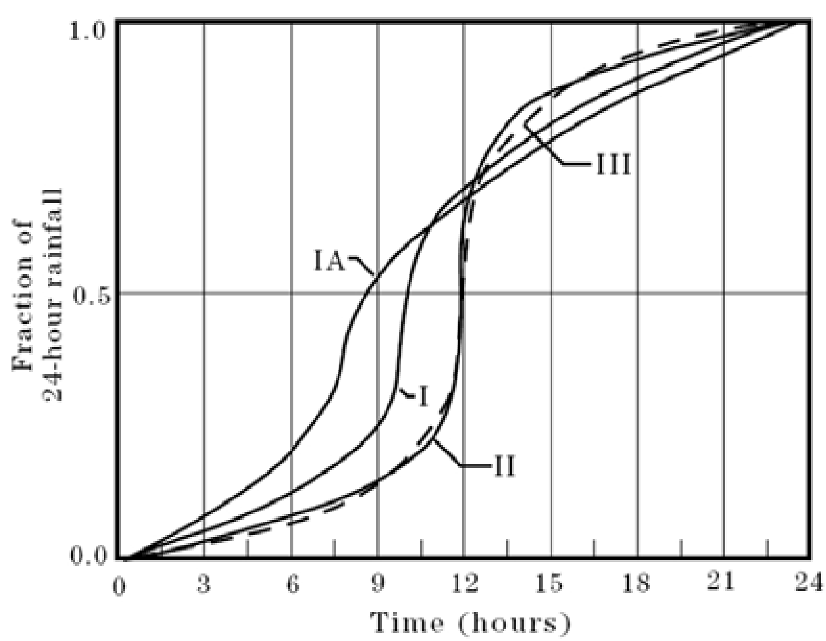
In addition, rainfall data like that shown in Figure 2-5 can be used to construct an entire hypothetical design storm. Such storms are typically needed when some or all of the runoff hydrograph (a depiction of how the runoff rate varies with time) is needed, not just the total runoff volume or peak runoff 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

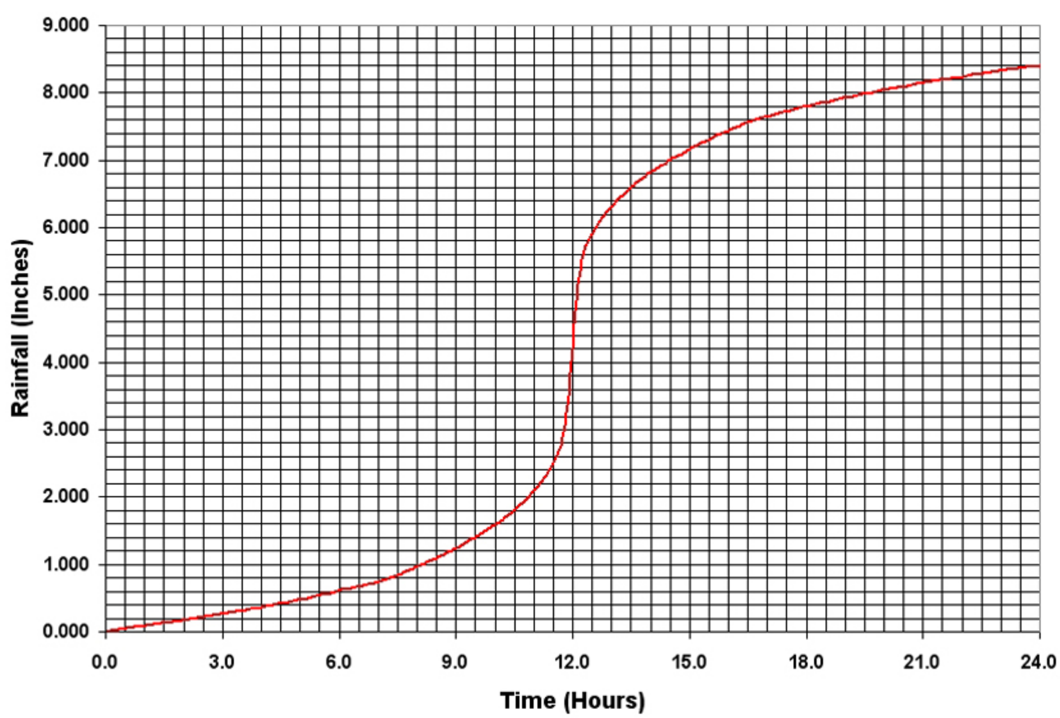
100-year frequency, 24-hour rainfall for Newark by the various rainfall depths shown in Figure 2-6 for the Type III design storm which the NRCS has designated as the most appropriate of the four design storms shown in Figure 2-6 for the city.

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 | | |

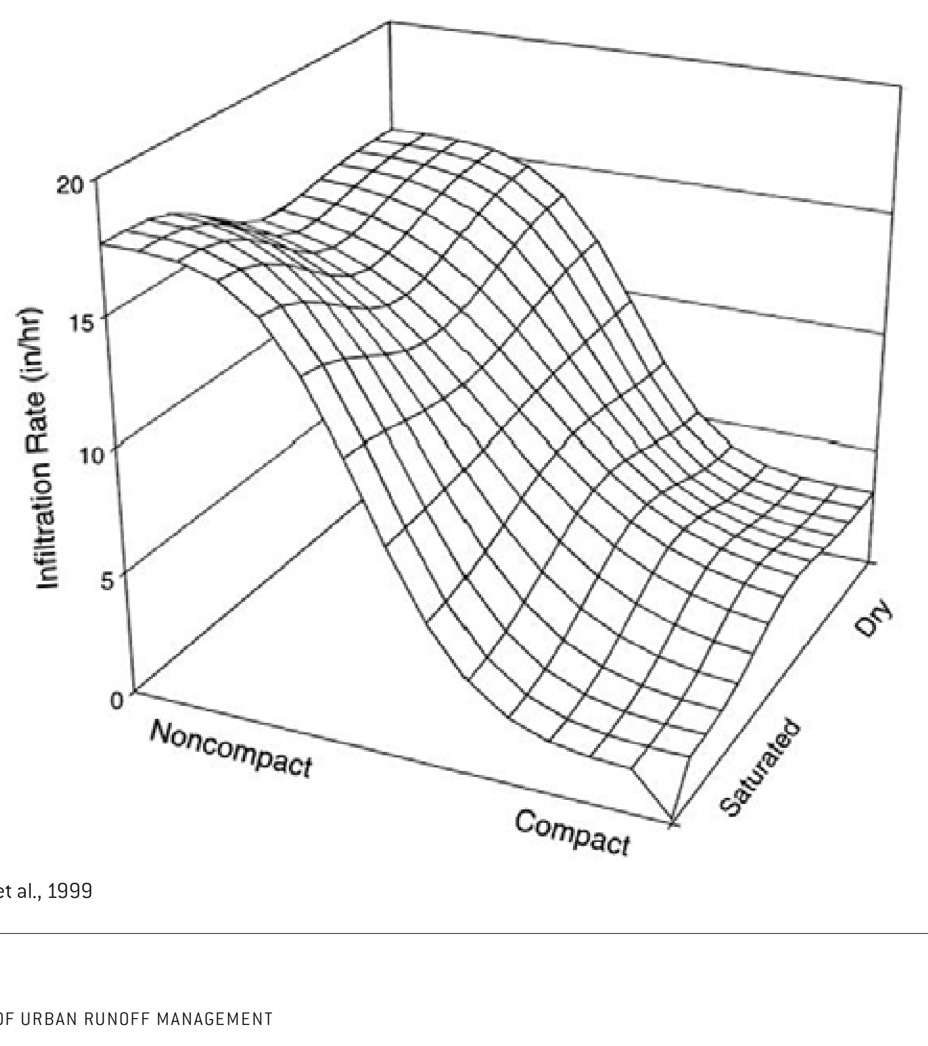
abstraction should not be overlooked, particularly for small rainfall depths.

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**



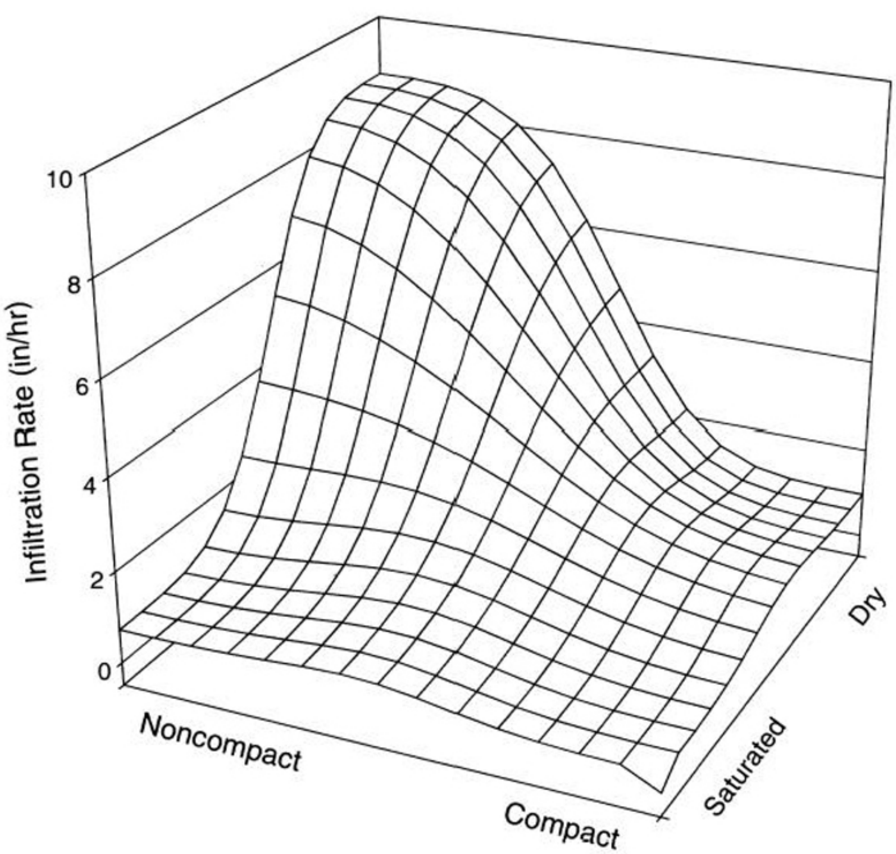

Source: Pitt et al., 1999

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.

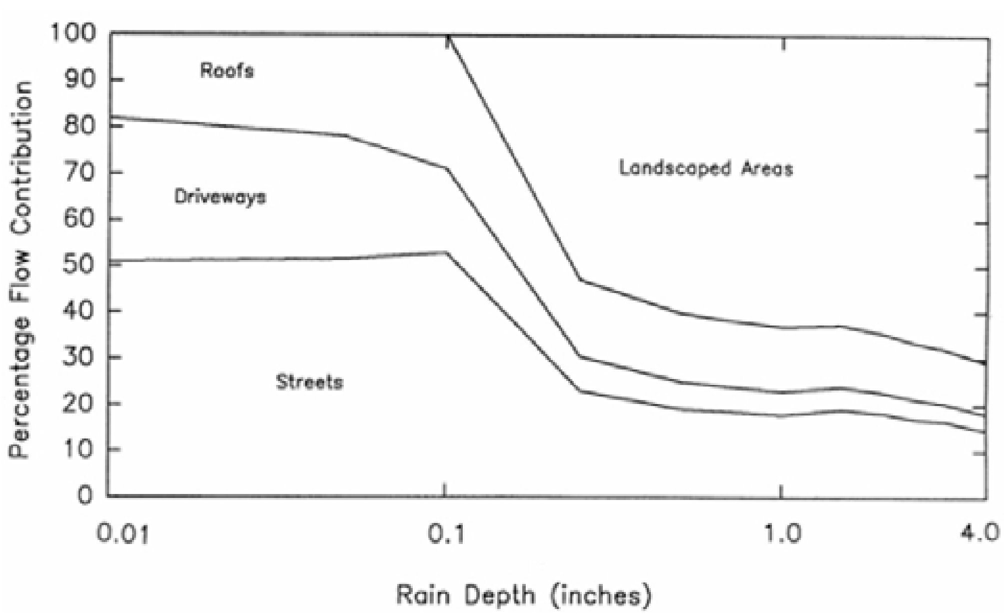
This difference is illustrated in Figure 2-10. It depicts the relative percentage of total runoff volume produced for a given amount of rain from various runoff source areas at a typical medium density residential housing site with clayey soils. As shown in the figure, site runoff would be entirely comprised 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.

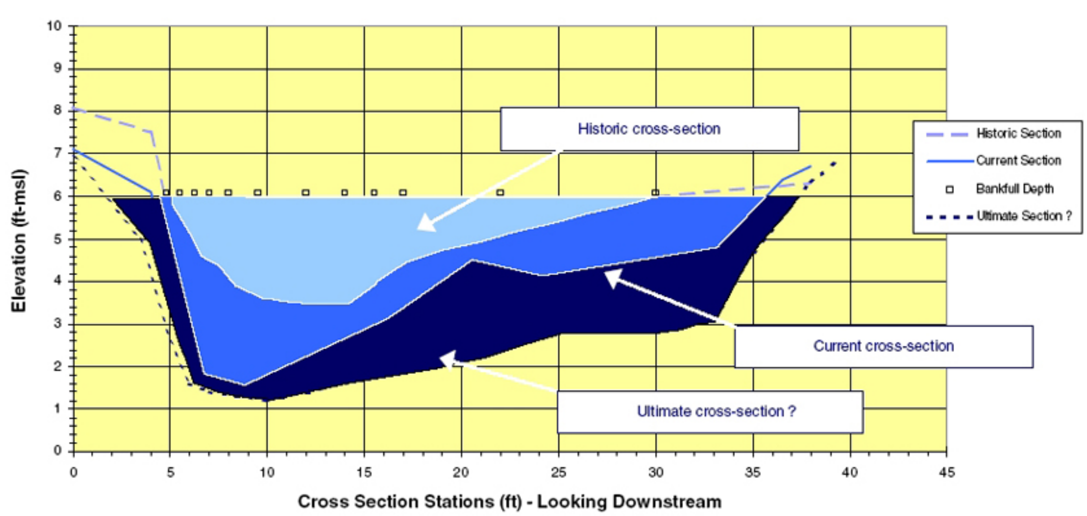
Such impacts can be quantified through a hypothetical land development example utilizing the NRCS Runoff Equation. The pre- and post-developed land uses and covers are summarized in Table 2-2. As shown in the table, the wooded land cover that exists in the pre-developed condition will be changed to a 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):

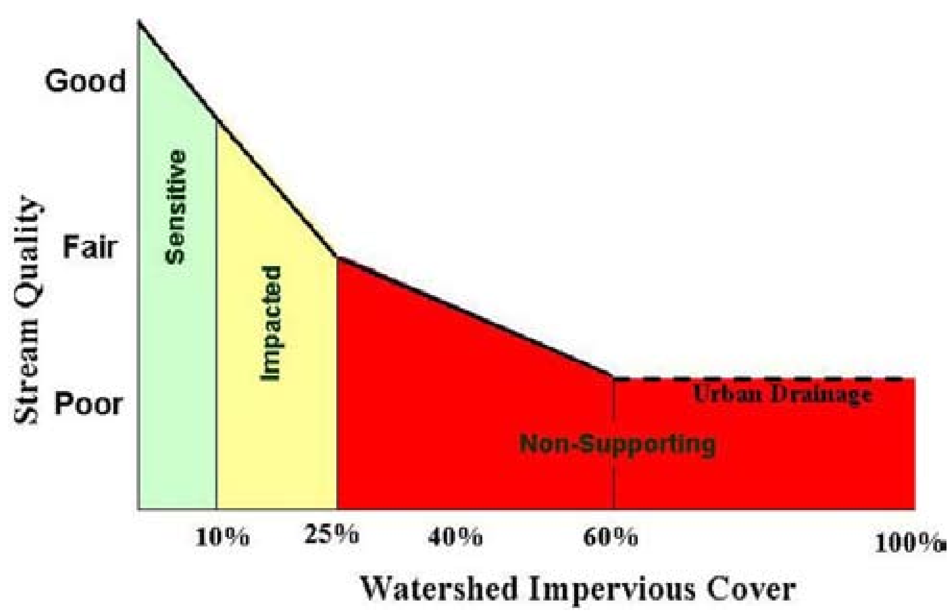
- Channel bank undercutting;
- Channel bottom incision;
- Loss of aquatic habitat;
- Increase in sediment yield and transport;
- Loss of riparian cover; and
- Increase in water temperature.

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

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# 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

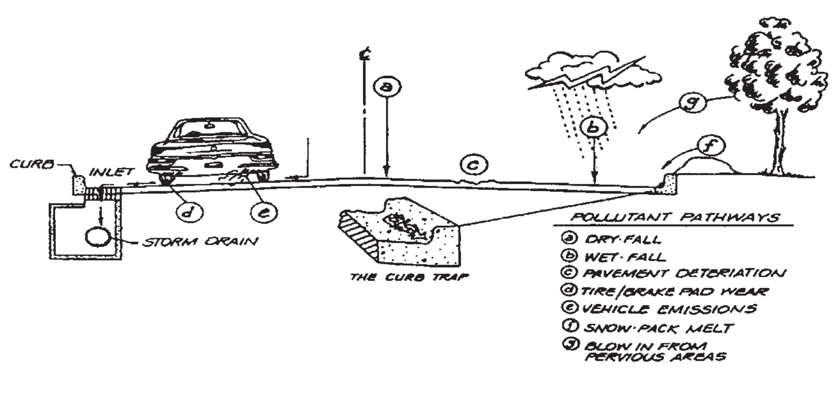
Stormwater runoff from urbanized areas is generated from a number of sources, including residential areas, commercial and industrial areas, roads, highways, and bridges. Essentially, any surface that does not have the capability to store and infiltrate water will produce runoff during storm events. These 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<br>Total suspended solids (TSS)<br>Turbidity (NTU) |
| Oxygen-demanding material | Biochemical oxygen demand (BOD)<br>Chemical oxygen demand (COD)<br>Organic matter (OM)<br>Total organic carbon (TOC) |
| Phosphorus (P) | Total phosphorus (TP)<br>Soluble reactive phosphorus (SRP)<br>Biologically available phosphorus (BAP) |
| Nitrogen (N) | Total nitrogen (TN)<br>Total kjeldahl nitrogen (TKN)<br>Nitrate + nitrite-nitrogen (NO3+NO2-N)<br>Ammonia-nitrogen (NH3-N) |
| Metals | Copper (Cu), lead (Pb), zinc (Zn), cadmium (Cd),<br>arsenic (As), nickel (Ni), chromium (Cr), mercury (Hg),<br>selenium (Se), silver (Ag) |
| Pathogens | Fecal coliform bacteria (FC)<br>Enterococcus bacteria (EC)<br>Total coliform bacteria (TC)<br>Viruses |
| Petroleum hydrocarbons | Oil and grease (OG)<br>Total petroleum hydrocarbons (TPH) |
| Synthetic organics | Polynuclear aromatic hydrocarbons (PAH)<br>Pesticides and herbicides<br>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<br>Automobiles<br>Industrial machinery |
| Copper (Cu) | Building materials<br>Paints and wood preservatives<br>Algicides<br>Brake pads |
| Zinc (Zn) | Galvanized metals<br>Paints and wood preservatives<br>Roofing and gutters<br>Tires |
| Lead (Pb) | Gasoline<br>Paint<br>Batteries |
| Chromium (Cr) | Electro-plating<br>Paints and preservatives |
| Cadmium (Cd) | Electro-plating<br>Paints and preservatives |
| Pesticides | Agriculture and grazing<br>Residential and commercial use |
| Herbicides | Agriculture and grazing<br>Residential and commercial use<br>Roadside vegetation maintenance |
| Organic compounds | Industrial processes<br>Power generation |
| Bacteria and pathogens | Human sewage<br>Livestock manure<br>Domestic animal fecal material |
| BOD | Agriculture and grazing<br>Human sewage |
| Nutrients (N and P) | Agriculture and grazing<br>Lawn and landscape fertilizer |
| Fine sediment | Agriculture and grazing<br>Timber harvest<br>Pavement wear<br>Construction sites<br>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

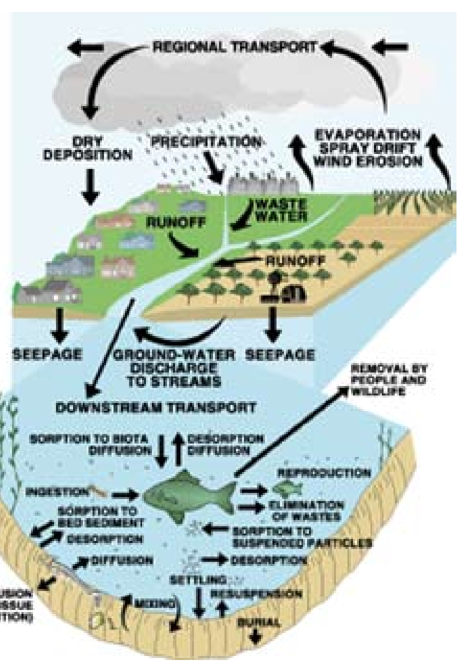
In general, the primary transport mechanism for most urban pollutants is stormwater runoff. The physio-chemical effects of watershed urbanization tend to be more variable than the hydro-geomorphic or physical habitat impacts discussed previously. As indicated above, stormwater can contain a variety of 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<br>NURP | 78<br>174 | 55<br>113 |
| BOD (mg/l) | Pooled<br>NURP | 14<br>10 | 12<br>8 |
| COD (mg/l) | Pooled<br>NURP | 53<br>66 | 45<br>55 |
| TP (mg/l) | Pooled<br>NURP | 0.32<br>0.34 | 0.26<br>0.27 |
| SRP (mg/l) | Pooled<br>NURP | 0.13<br>0.10 | 0.10<br>0.08 |
| TKN (mg/l) | Pooled<br>NURP | 1.73<br>1.67 | 1.47<br>1.41 |
| NO2-N and NO3-N (mg/l) | Pooled<br>NURP | 0.66<br>0.84 | 0.53<br>0.66 |
| Total Cu (ug/l) | Pooled<br>NURP | 14<br>67 | 11<br>55 |
| Total Pb (ug/l) | Pooled<br>NURP | 68<br>175 | 51<br>131 |
| Total Zn (ug/l) | Pooled<br>NURP | 162<br>176 | 129<br>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

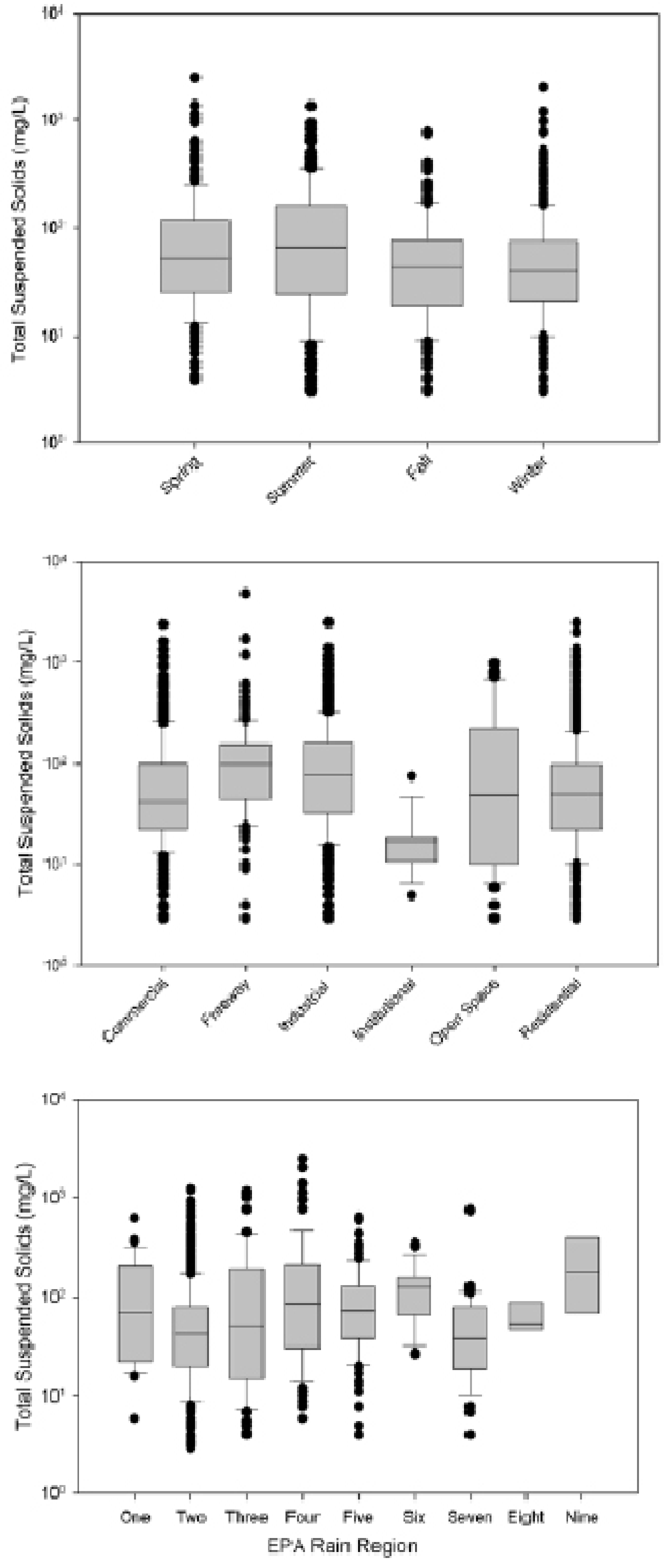
Source: NSQD, 2004

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 NO3 + NO2-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)

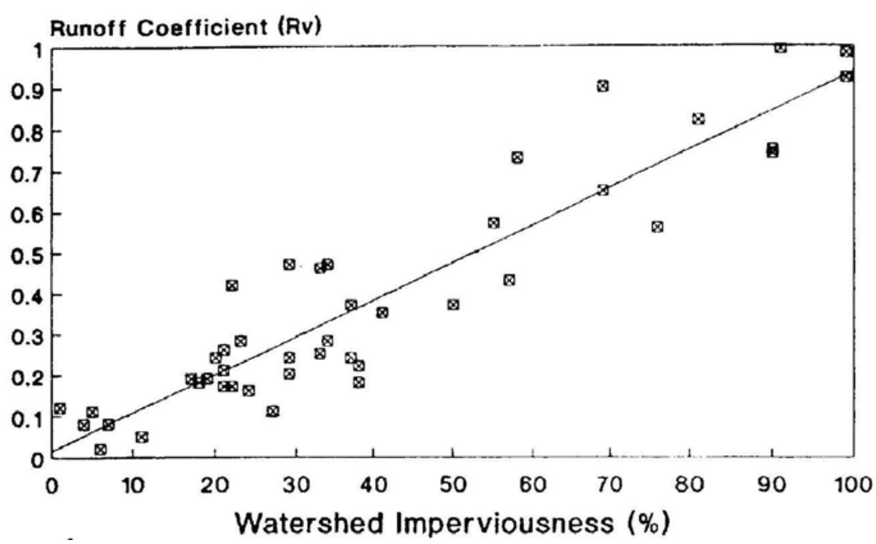
R = Annual runoff (inches)

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 ($N = 47$, $R^2 = 0.71$) 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

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.

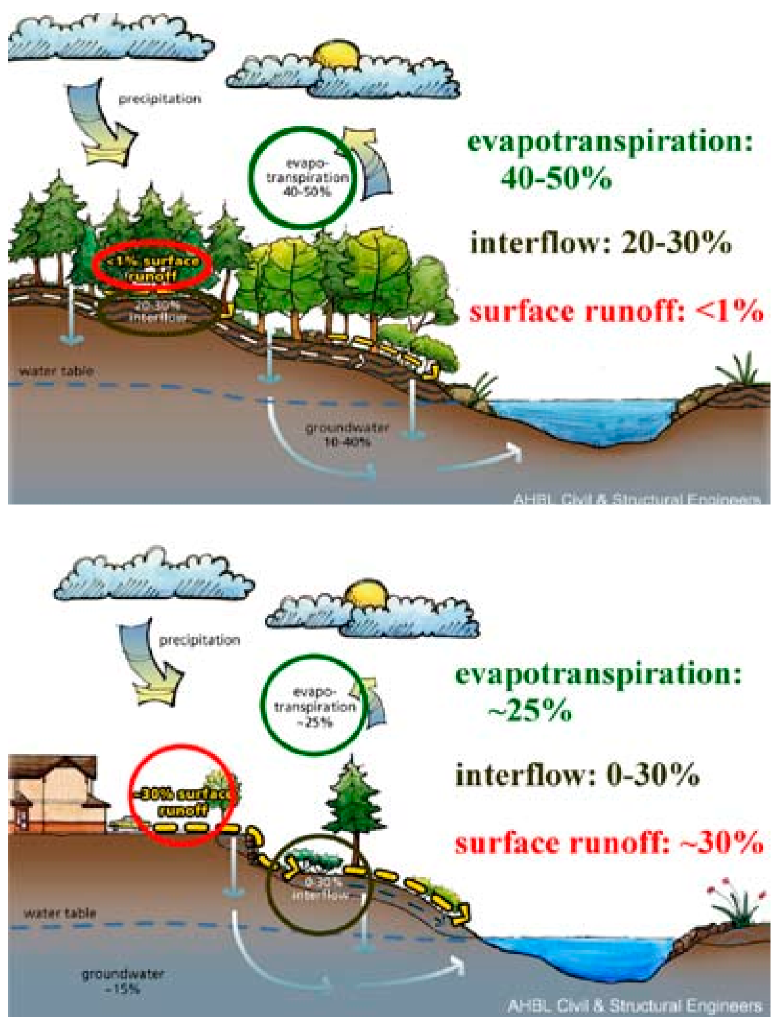
Savani and Kammerer (1961) first discussed the relationship between natural land cover and developed land use with respect to the stages of urbanization. This early research identified four stages of urbanization, each associated with characteristic changes in the hydrologic regime. These stages are 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

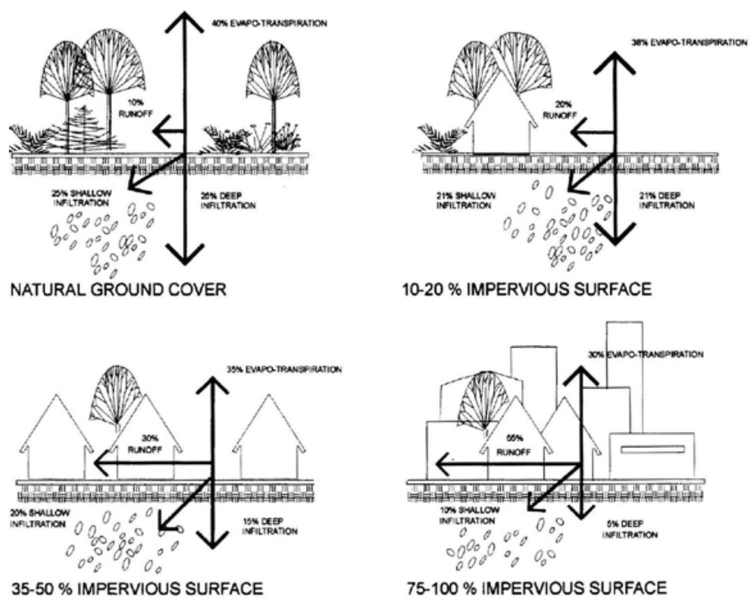
Figure 4-1: Comparison between the hydrologic regime for a natural, undeveloped watershed (upper) and an urbanized watershed in the Pacific Northwest



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).

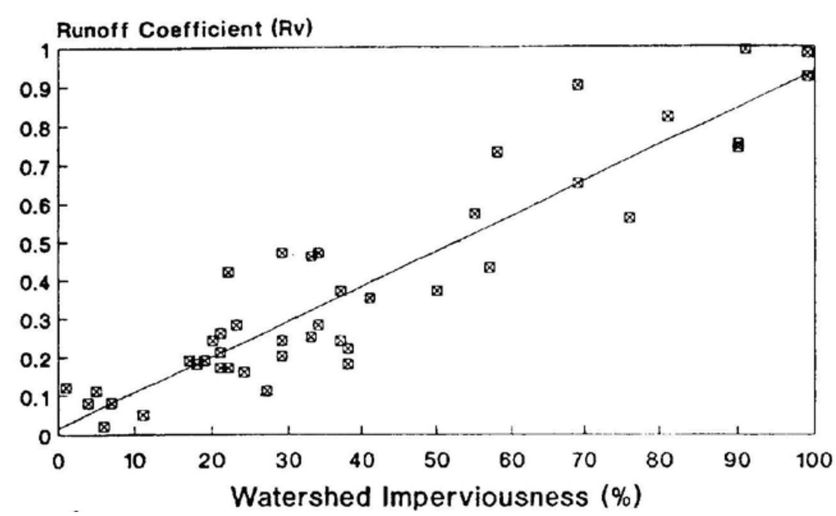
The total fraction of a watershed that is covered by impervious surface areas is typically referred to as the percent total impervious area (%TIA). The %TIA of a watershed is a landscape-level indicator that integrates several concurrent interactions influencing the hydrologic regime as well as water 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**

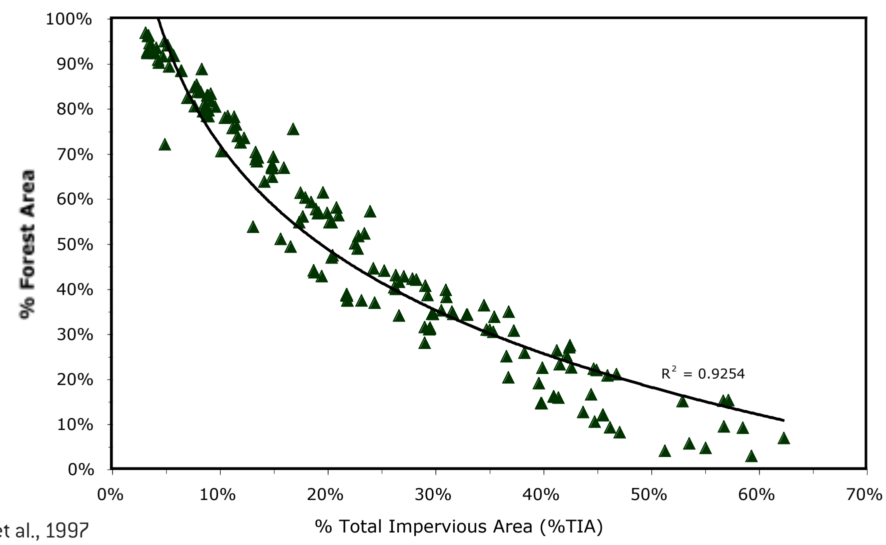



Source: Schueler, 1994

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).

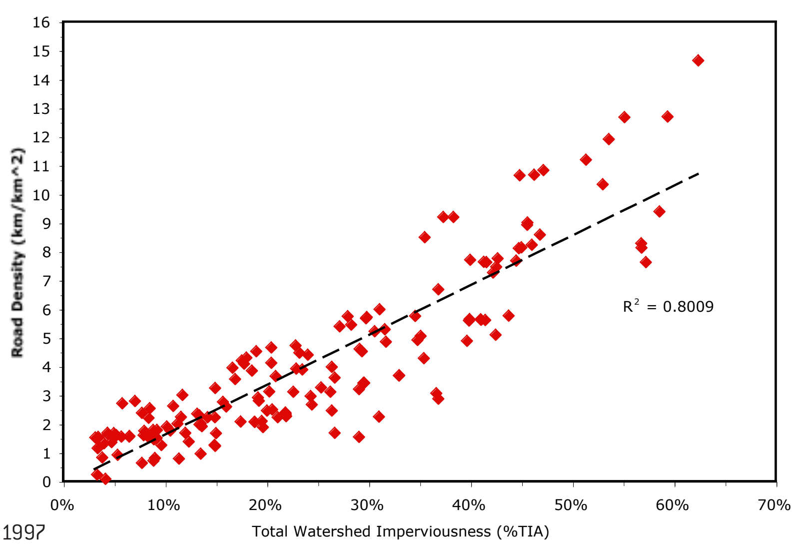
Another useful indicator of landscape-scale changes in watershed condition is the fraction of the basin that is covered by natural vegetation. In many areas, forest cover is the key parameter, but in other regions, prairie or shrub-savannah could be the key natural vegetation community. In any case, 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 streamflow 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).

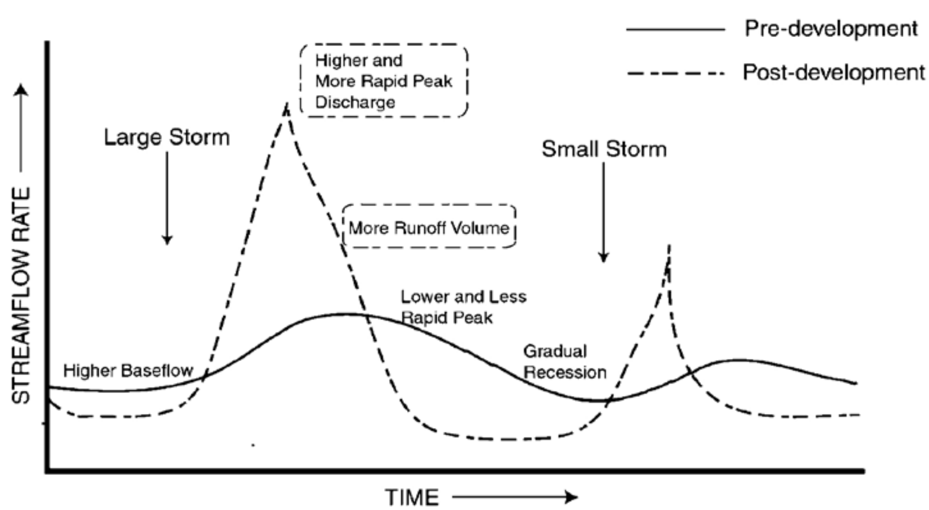
The many changes brought on by urbanization tend to alter streamflow patterns in characteristic ways.

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).

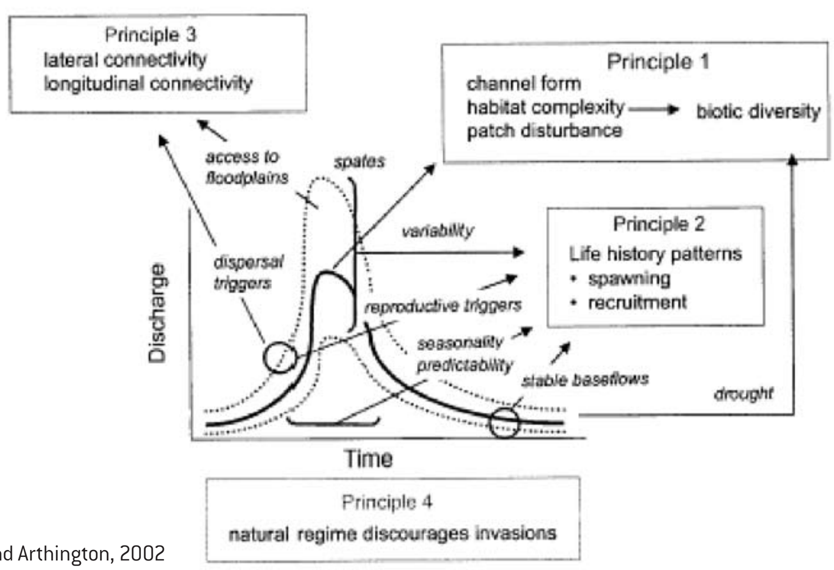
An extensive study comparing an urban (Kelsey Creek) and a non-urban (Big Bear Creek) stream in the Puget Sound region found that hydrologic changes from urbanization were the principal reasons that the urban stream failed to match its non-urban counterpart in diversity and size of salmonid fish populations 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



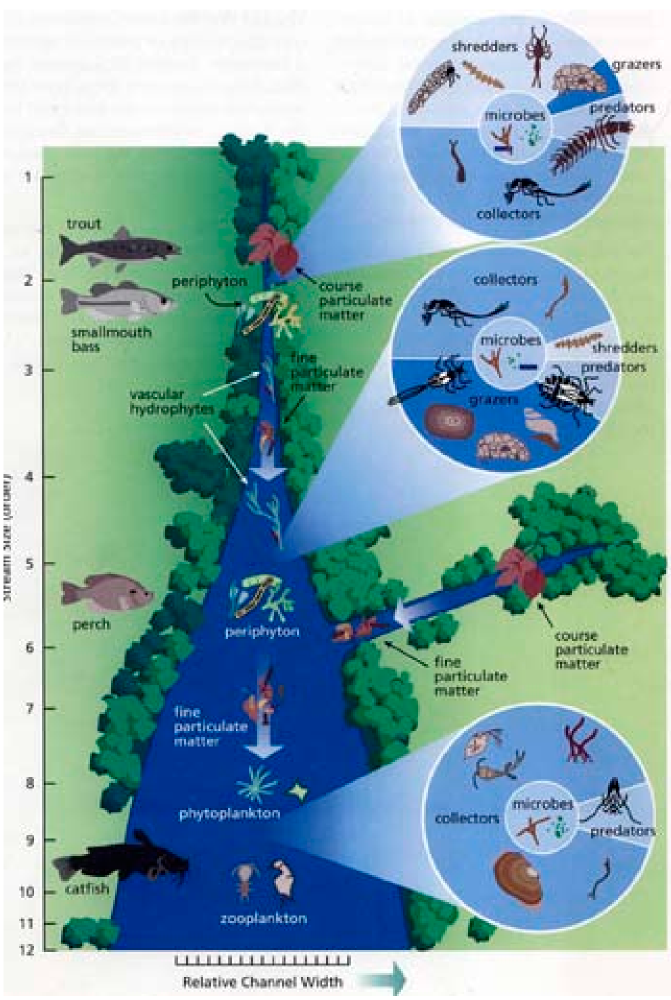

Source: Bunn and Arthington, 2002

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.

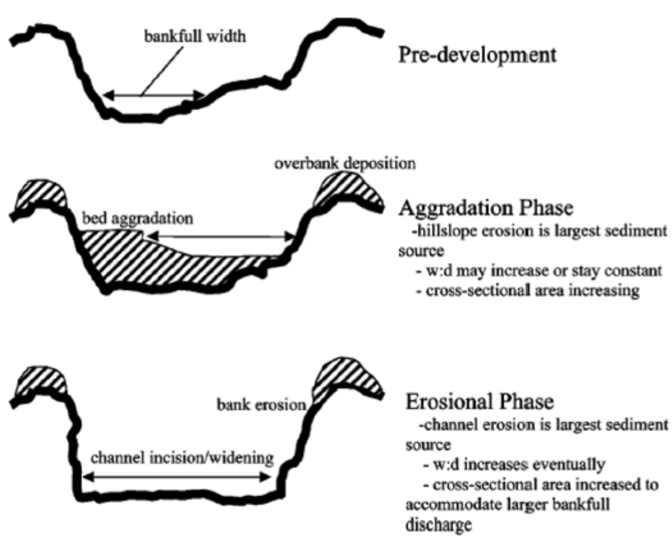
Channel enlargement can be a gradual process that follows the pace of urbanization, or it can frequently occur abruptly in response to particular storms (Hammer, 1972; Leopold, 1973; Booth, 1989; Booth and Henshaw 2001). Even in cases where the stream has been stable for many years, abrupt and sometimes 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

many other terrestrial predators or scavengers 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:

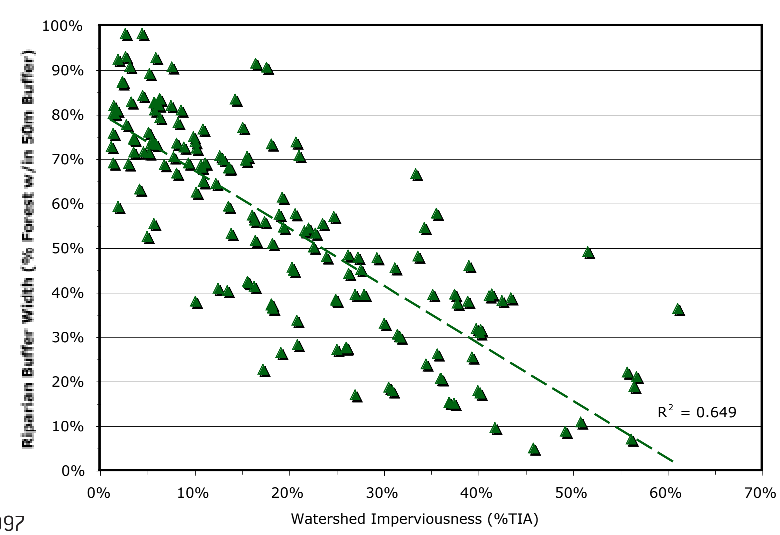
- Lateral riparian extent (so-called "buffer" width);
- Longitudinal riparian corridor connectivity (low fragmentation);
- Riparian quality (vegetation type, diversity, and maturity); and
- Floodplain and channel migration zone (CMZ) integrity.

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**



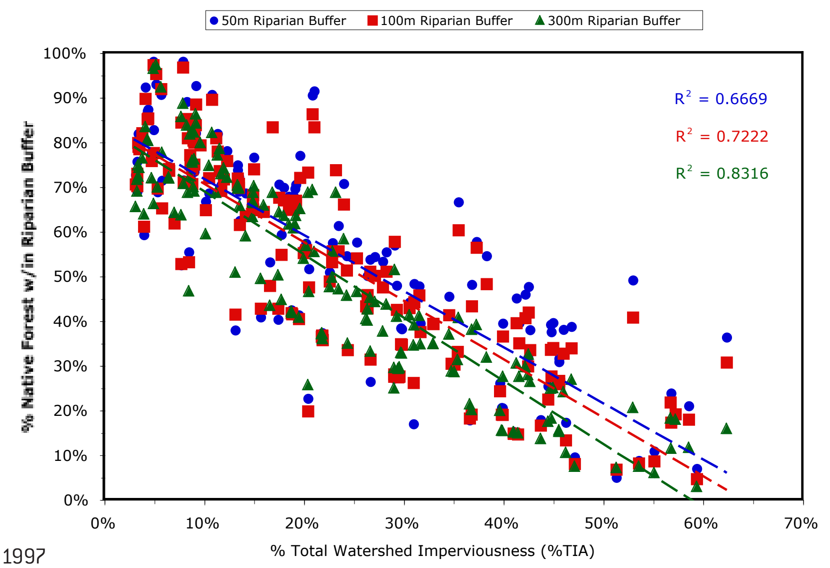

Source: May et al., 1997

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.

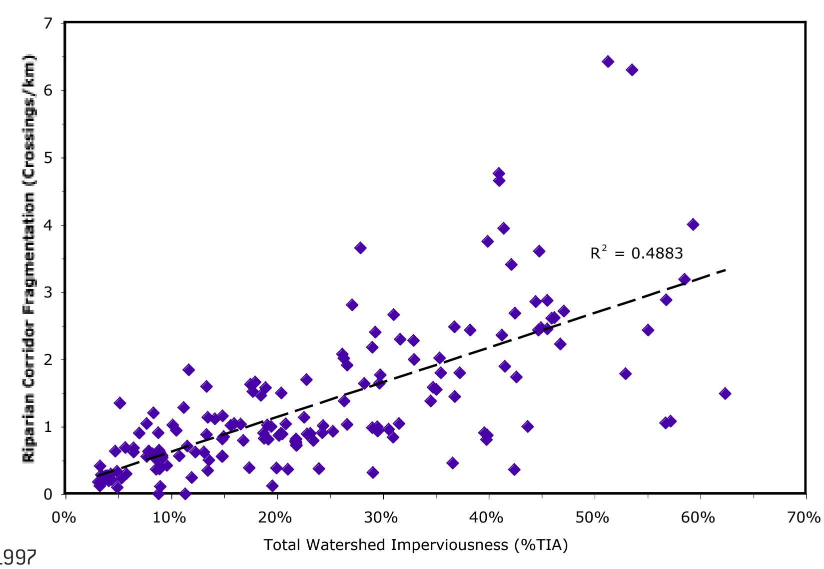
Invasive or exotic plants are another problem common to urban stream and wetland buffers. Human encroachment and landscaping activities can introduce

**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

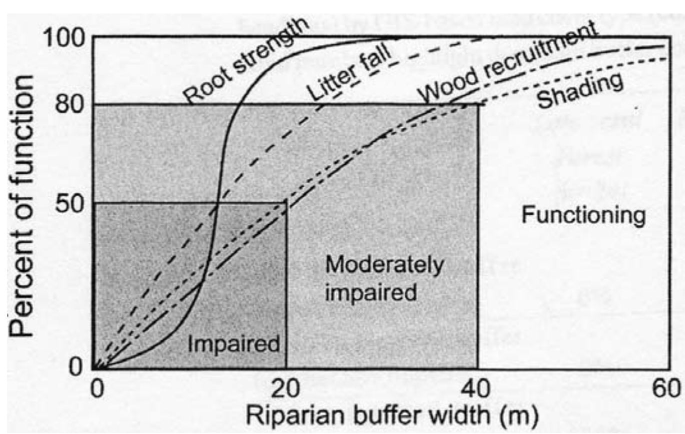
Degradation of aquatic habitat is one of the most significant ecological impacts of the changes that accompany watershed urbanization. The complex physical effects from elevated urban streamflows, stream channel alterations, and riparian encroachment can damage or destroy stream and wetland habitats. 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 |

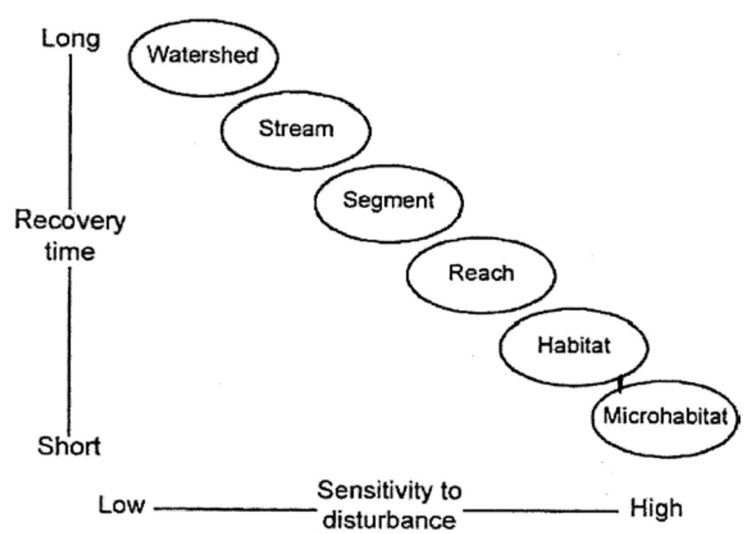
percent of the stream banks armored with "rip-rap" and the floodplain-riparian zone also highly modified. Bear Creek, on the other hand, had less than 10 percent stream bank armoring and a natural riparian corridor and CMZ. Road-crossing bridges and culverts were frequent on Kelsey Creek, but not on 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.

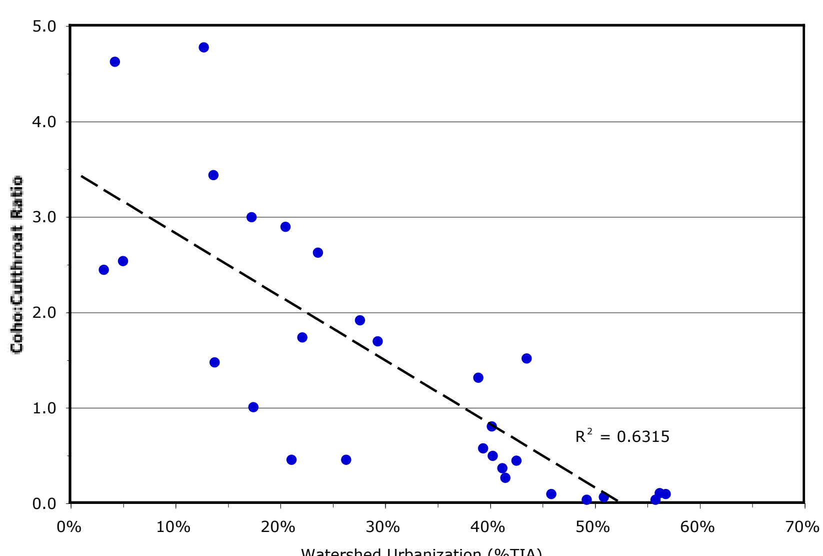
Similar to these studies in the Pacific Northwest, Morse (2003) observed that both instream habitat and water quality in small urbanizing streams in Maine declined in a linear fashion. Studies in Delaware (Maxted and Shaver, 1997), Wisconsin (Wang et al., 1997), and Minnesota (Nerbonne and Vondracek, 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).

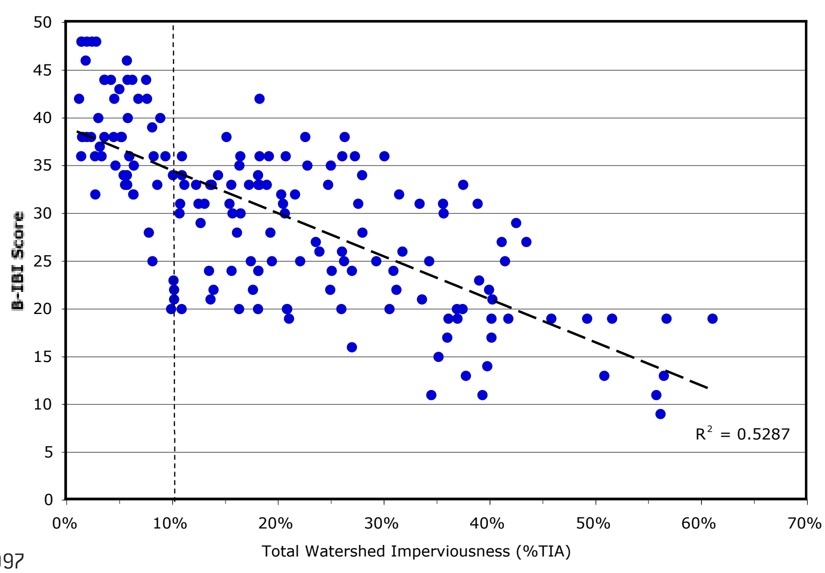
In several extensive studies of urbanizing streams in Wisconsin, a significant relationship was found between watershed land use and instream habitat as well as stream fish communities (Wang et al., 1997; Wang et al., 2000; Wang et al., 2001). In these studies, stream fish abundance and diversity both 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.

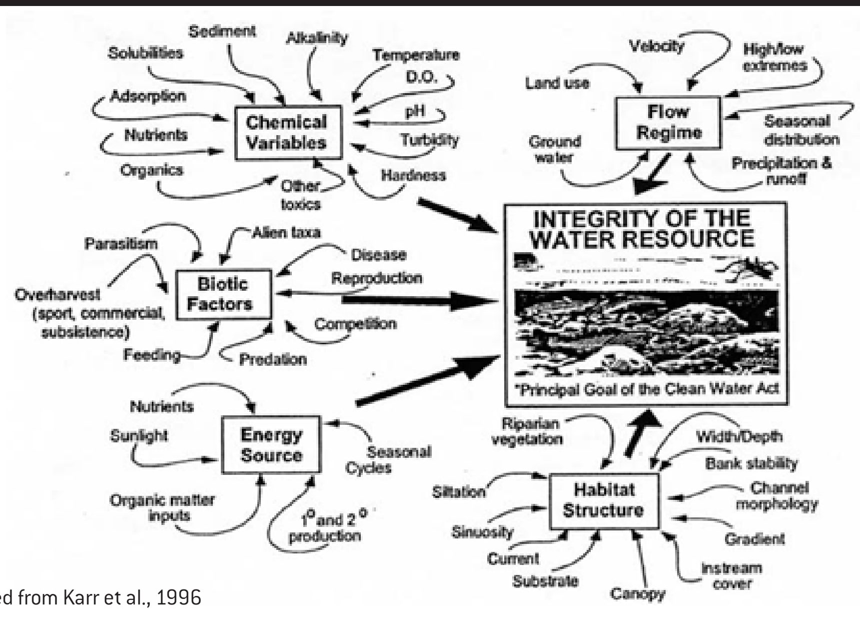
Impervious surfaces (roads, parking lots, rooftops, etc.) are the primary source areas for pollutants to collect within the built environment. Runoff from storm events then carries these pollutants into natural waters via the stormwater conveyance network. The land use (e.g., residential, commercial, 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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