CHAPTER 2

Receiving Water Uses, Impairments, and Sources of Stormwater Pollutants

"Bathing in sewage-polluted seawater carries only a negligible risk to health, even on beaches that are aesthetically very unsatisfactory."

Committee on Bathing Beach Contamination
Public Health Laboratory Service of the U.K.
1959

CONTENTS

INTRODUCTION

Wet-weather flow impacts on receiving waters have been historically misunderstood and de-emphasized, especially in times and areas of poorly treated municipal and industrial discharges. The above 1959 quote from the Committee on Bathing Beach Contamination of the Public Health Laboratory Service of the U.K. demonstrates the assumption that periodic combined sewer overflows (CSOs), or even raw sewage discharges, produced negligible human health risks. Is it any wonder then that the much less dramatically contaminated stormwater discharges have commonly been considered "clear" water by many regulators?

The EPA reported that only 57% of the rivers and streams in the United States fully support their beneficial uses (Figure 2.1). A wide variety of pollutants and sources are the cause of impaired

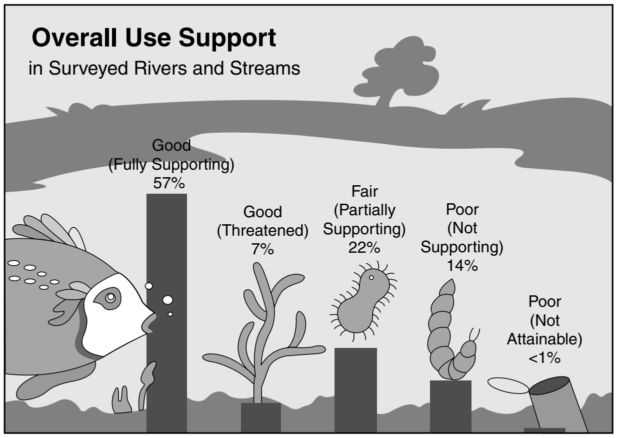

Figure 2.1 U.S. rivers and streams meeting designated beneficial uses. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

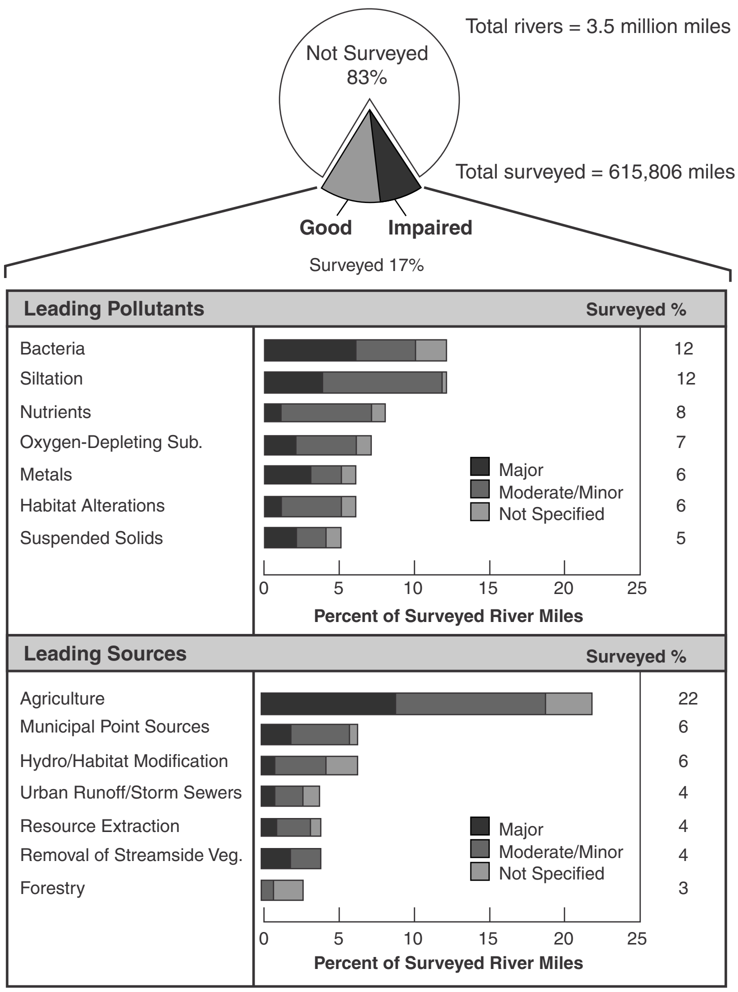

Figure 2.2 Pollutants and sources impairing U.S. rivers. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

uses (Figures 2.2 through 2.6) but runoff from urban and agricultural sources dominate. This book contains discussions of instances of beneficial use impairments associated with stormwater runoff and the possible sources of the stressors of these effects. However, stormwater effects on receiving waters are not always clear and obvious. As will be evident to the reader, most stormwater runoff

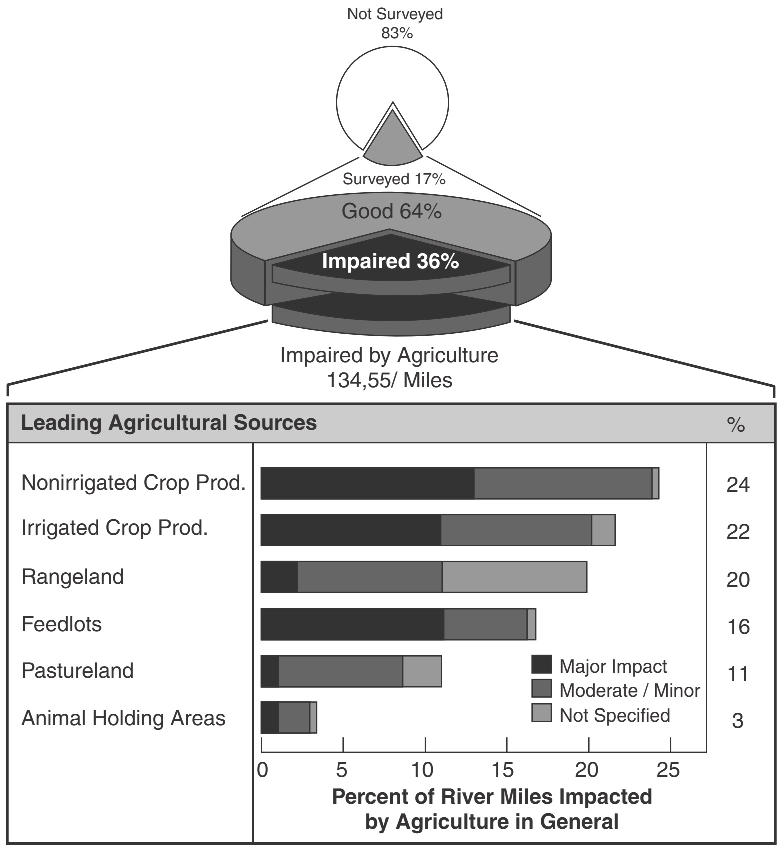

Figure 2.3 Agricultural activities affecting U.S. rivers and streams. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

assessments have been conducted in urban waterways, with fewer examples for agricultural systems. However, many of the approaches, methods, and receiving water effects are similar in both urban and agriculturally dominated waterways. In completely urbanized watersheds, the small urban streams are commonly severely degraded, but they typically have no official beneficial uses or monitoring programs (and may be intermittent in flow), and are therefore unrecognized as being impacted or important. Unfortunately, these streams receive substantial recreational use by neighborhood children. Besides the obvious safety concerns and potential drowning fears, the water quality of urban streams can present significant risks. In older cities, stream sediments downstream from historical industrial areas can be heavily contaminated by heavy metals and organic compounds. Even in nonindustrialized areas, metallic and organic contamination can be high. Unfortunately, bacteria concentrations, especially near outfalls during and soon after rains, are always very high in these small streams, although the health risks are poorly understood. Sediment bacteria conditions are also always high, as the sediments appear to be an excellent sink for bacteria. Children, and others, playing in and near the streams therefore are exposed to potentially hazardous conditions. In addition, inner-city residents sometimes rely on nearby urban waterways for fishing opportunities, both for recreation and to supplement food supplies.

In contrast to the above obvious conditions associated with small streams in completely urbanized watersheds, wet-weather flows from relatively large cities discharging into large waterways may not be associated with obvious in-stream detrimental conditions. In one example, frequent CSO discharges from Nashville, TN, into the Cumberland River were not found to produce any significant dissolved oxygen (DO) or fecal coliform problems (Cardozo et al. 1994). However, Nashville is currently investigating sources of high bacteria levels in the small urban streams draining heavily urbanized city watersheds. A series of studies of airport deicing compound runoff

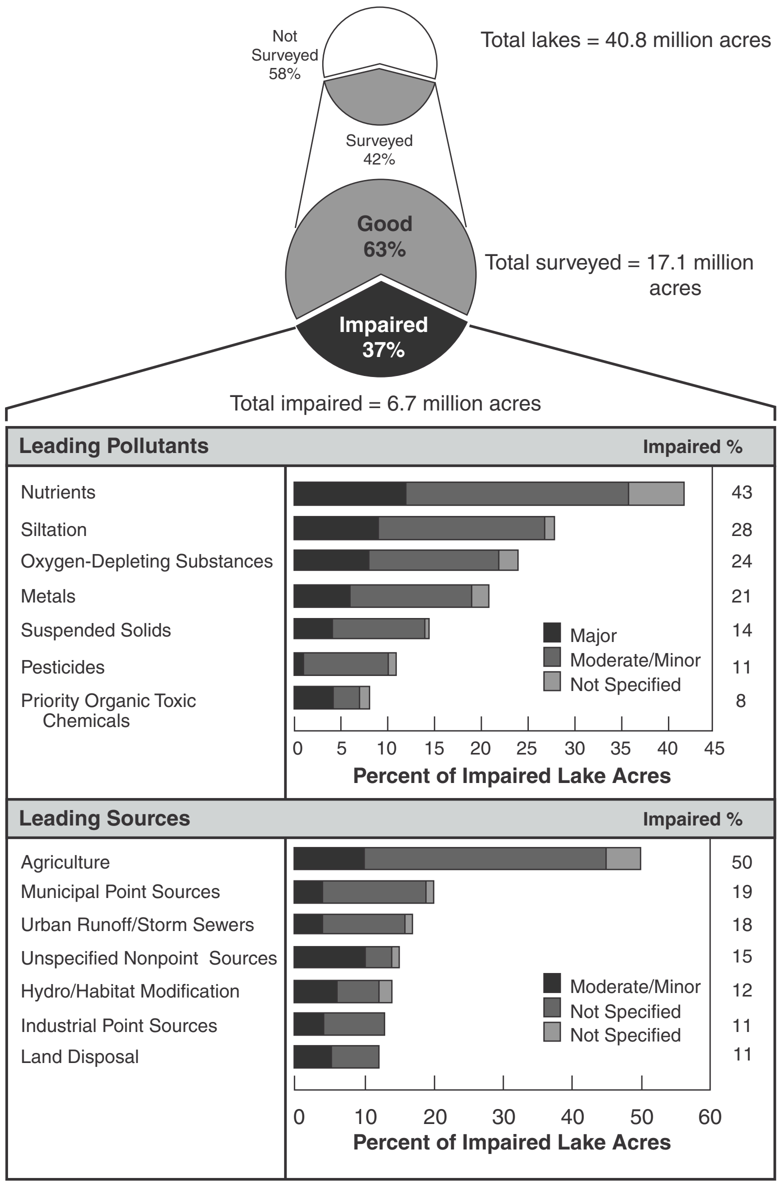

Figure 2.4 Pollutants and sources affecting U.S. lakes. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

at Milwaukee's Mitchell Field is another example that demonstrates unique site-specific conditions affecting receiving water impacts. This study, conducted by the USGS and the Wisconsin Department of Natural Resources, found that the extremely high BOD concentrations (several thousand mg/L) associated with the deicing runoff had negligible effects on the DO levels in the small streams draining the airport area to Lake Michigan. They concluded that the cold temperatures occurring during the times of deicing runoff significantly reduced the BOD decomposition rate, and that the small streams had short travel times before discharging into Lake Michigan, where it was well mixed. Under laboratory conditions, the BOD rate would be much faster, and would be expected to produce dramatically low DO conditions for almost any condition in these small streams.

Other obvious receiving water problems, such as fish kills, are also rarely associated with stormwater discharges, as described in Chapter 3. Stormwater discharges occur frequently, and normally do not create acute toxicity problems (or extremely low DO conditions). Fish surviving

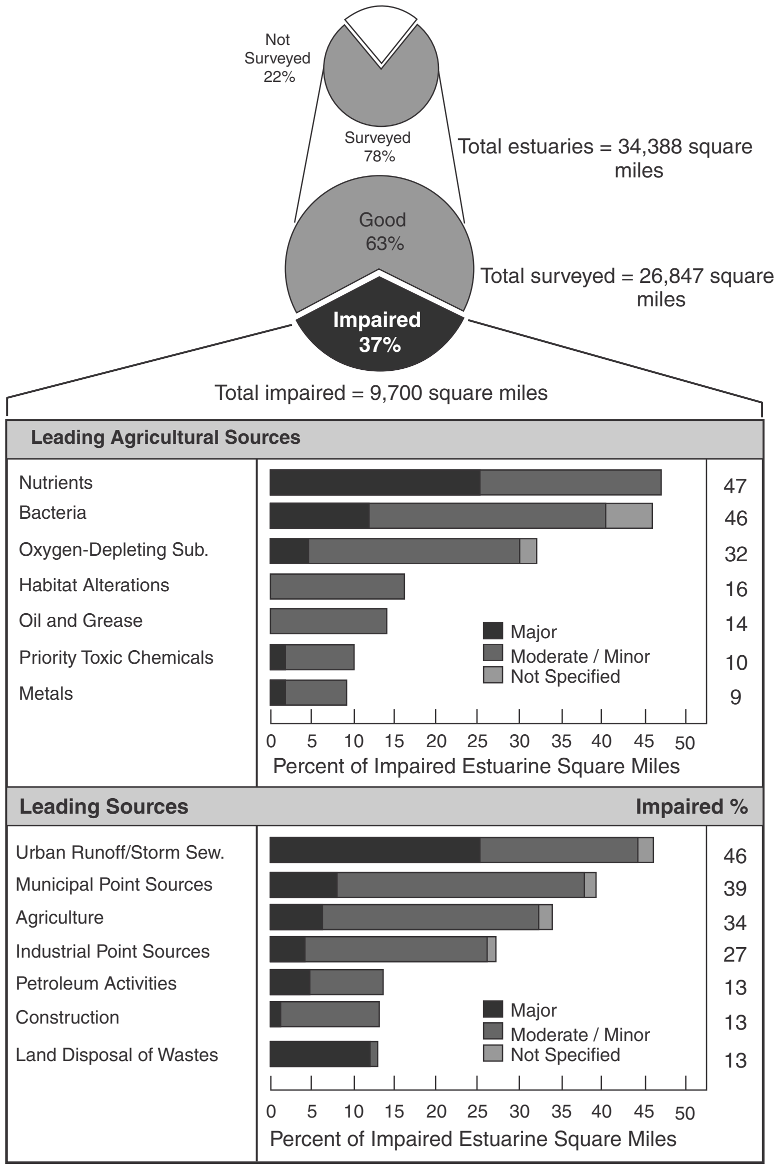

Figure 2.5 Pollutants and sources affecting U.S. estuaries. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

in urban streams are tolerant species, with most of the intolerant organisms long since gone. It is therefore unusual for fish kills to occur, unless severe inappropriate discharges infrequently occur (such as those associated with industrial accidents, runoff from fire fighting, or improper waste disposal activities). However, chronic toxicity, mostly associated with contaminated sediments or suspended solids, is associated with stormwater. The effects of this chronic toxicity, plus habitat problems, are the likely causes of the commonly observed significant shifts in the in-stream biological community from naturally diverse (mostly intolerant) species to a much less diverse assemblage of introduced tolerant species. There is increasing evidence that stormwaters in urban and agriculturally dominated watersheds are often toxic (see Chapter 6). However, traditional toxicity approaches often do not detect problems associated with pulse exposures and or particulate-associated toxicity. More recently, both laboratory and in-stream (*in situ*) toxicity tests, especially associated with moderate to long-term exposures to contaminated sediments and particulates, have shown significant stormwater toxicity.

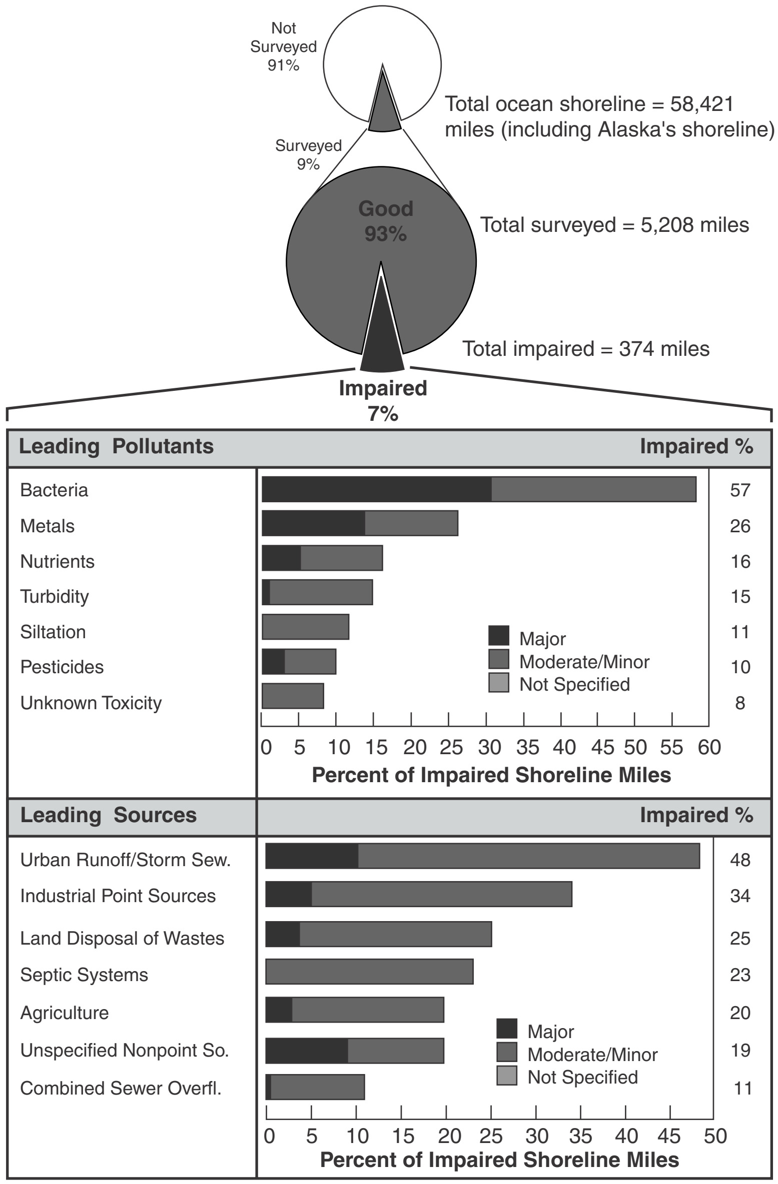

Figure 2.6 Pollutants and sources affecting U.S. ocean shorelines. Note: Percentages do not add to 100% because more than one pollutant or source may impair a segment of ocean shoreline. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

The discharges of stormwater are also periodic, causing different types of effects than the better-regulated continuous point source discharges. Stormwater causes episodic disturbances in aquatic ecosystems (Minshall 1988) whose patterns of occurrence are chaotic in nature (Pool 1989) and characteristics are unique to each event. The sciences of aquatic ecology and aquatic toxicology have progressed to the point where the effects of continuous levels of single stressors (e.g., dissolved oxygen, temperature, copper, DDT, diazinon, chlorpyrifos) on a wide variety of common aquatic species are known. The effects that the single stressors have, or may have, in stormwater are therefore known with reasonable certainty. However, as is shown in Table 2.1, nonpoint sources, including stormwater, contain multiple stressors that are applied intermittently, and science currently has a poor understanding of stressor interactions and effects.

The attributes of each stormwater event are a result of previous meteorological conditions (e.g., dry deposition, air patterns, humidity), land use patterns (e.g., traffic and parking patterns, construction and landscaping activities), storm intensity and duration, and other watershed character-

Table 2.1 Potential Effects of Some Sources of Alteration on Stream Parameters

Stream Parameter	Acid Mine Drainage or Acid Precipitation	Sewage Treatment Plant Discharge	Agriculture Runoff (pasture or cropland)	Urban Runoff	Channelization	Pulp and Paper	Textile	Metal Finishing and Electroplating	Petroleum	Iron and Steel	Paint and Ink	Dairy and Meat Products	Fertilizer Production and Lime Crushing	Plastics and Synthetics
pH	D					C	I	C		D	C		D,I	C
Alkalinity	D						I						D,I	
Hardness	I						I						I	
Chlorides		I	I	I							I	I		
Sulfates	I								I	I	I		I	I
TDS	I			I			I		I		I	I	I	
TKN		I	I	I					I	I	I	I	I	I
NH_3-N		I		I					I	I			I	I
Total-P		I	I	I				I	I			I	I	
Ortho-P		I		I				I			I		I	
BOD_5		I		I		I	I		I	I		I		I
COD	I	I		I		I	I			I	I	I		I
TOC		I	I	I		I			I	I	D	I		I
COD/BOD	I			I		I	I		D	I				
D.O.		D				D					I	D		
Volatile compounds			I	I		I			I					I
Fluoride						I		I			I		I	I
Cr				I		I	I	I	I					I
Cu	I			I				I			I			I
Pb				I				I	I	I	I			
Zn	I			I		I		I	I	I				I
Cd		I		I				I						I
Fe	I			I				I		I			I	I
Arsenic				I										
Mercury				I						I				I
Cyanide				I						I				I
Oil and grease				I		I	I	I	I			I		
Coliforms	D	I	I					D		D	I	I		
Chlorophyll	D	I	I		I	D		D	D	D	D	I	I	D
Diversity	D	D		D	D	D	D	D	D	D	D		D	D
Biomass	D	D	I		I		I		D		D	I	I	D
Riparian factors				C	C									
Temperature				I	I									
TSS			I	I	I	I	I	I					I	I
VSS				I	I	I	I							
Color						I						I		I
Conductivity	I	I					I						I	
Channel factors				C	C									

D = decrease, I = Increase, C = change.
From EPA (U.S. Environmental Protection Agency). *Results of the Nationwide Urban Runoff Program.* Water Planning Division, PB 84-185552, Washington, D.C. December 1983.

istics. Because of the potentials for extreme heterogeneity in stormwater and its associated quality, predicting effects to receiving waters is difficult and crude at best. Stormwaters often contain a large number of potential stressors to aquatic ecosystems. These stressors include oxygen demand, suspended solids, dissolved solids (including salts), altered ion ratios, nutrients, pathogens, metals, natural and synthetic organics, pH, and temperature. These stressors may interact to varying degrees in an antagonistic, additive, or synergistic fashion, affecting organisms in the receiving water.

There are numerous receiving water problems associated with stormwater that interfere with beneficial uses. The most obvious is the substantial increase in runoff causing increases in the frequency and magnitude of flooding along urban streams. Increases in stream flows also cause significant habitat problems in urban streams by attempting to enlarge the stream cross sections, causing significant channel erosion and unstable conditions. Stream-side residents also dramatically affect habitat by removing riparian vegetation and large organic debris from the streams. Another significant and obvious effect is the increase in sediment associated with poorly controlled construction site runoff. This sediment smothers coarse stream sediments that are needed by many spawning fish, and fills in stream pool areas. Another obvious receiving water problem associated with stormwater is the large amount of floating trash and litter (some hazardous) that is discharged by stormwater and that accumulates along urban waterways. This creates unsightly and potentially hazardous conditions interfering with noncontact recreational uses of the stream corridors.

The degree of impact on an exposed organism is dependent on numerous factors, such as the organism's sensitivity, life stage, feeding habits, frequency of exposure, and magnitude and duration of exposure. The organism or community affected by stormwater induces changes in other components of their ecosystem including habitat, food sources, predator–prey relationships, competition, and other behavior patterns. It is clear that there is no simple method by which to detect an effect of stormwaters on the receiving water ecosystem. Human health and safety concerns associated with stormwater discharges are also highly variable depending on many site conditions. Chapters 3 and 4 discuss ways in which effects can be assessed effectively, despite the complex, heterogeneous nature of the system, while Chapters 5 and 6 describe how specific monitoring activities can be carried out. Chapters 7 and 8 outline ways to evaluate the collected data to accomplish the study goals, outlined in Chapter 4.

The main purpose of treating stormwater is to reduce its adverse impacts on receiving water beneficial uses. Therefore, it is important in any stormwater runoff study to assess the detrimental effects that runoff is actually having on a receiving water. Below are discussions of the basic receiving water beneficial uses that need to be considered in all cases.

BENEFICIAL USE IMPAIRMENTS

Recognized Value of Human-Dominated Waterways

With full development in a watershed and with no stormwater controls, it is unlikely that any of the basic beneficial uses can be achieved. With less development, and with the application of stormwater controls, some uses may be possible. However, it is important that unreasonable expectations not be placed on urban or agricultural waters, as the cost to obtain these uses may be prohibitive. With full-scale development and lack of adequate stormwater controls, severely degraded streams will be common. In all cases, stormwater conveyance and aesthetics should be the basic beneficial use goals for all human-dominated waters. Biological uses should also be a goal, but with the realization that the natural stream ecosystem will be severely modified with urbanization and agricultural activities. Certain basic stormwater controls, installed at the time of development, plus protection of stream habitat, may enable partial to full use of some of these basic goals. Careful planning and optimal utilization of stormwater controls are necessary to obtain these basic goals in most watersheds. Water contact recreation, consumptive fisheries, and water

Figure 2.7 0 Original section of Riverwalk in San Antonio, TX.

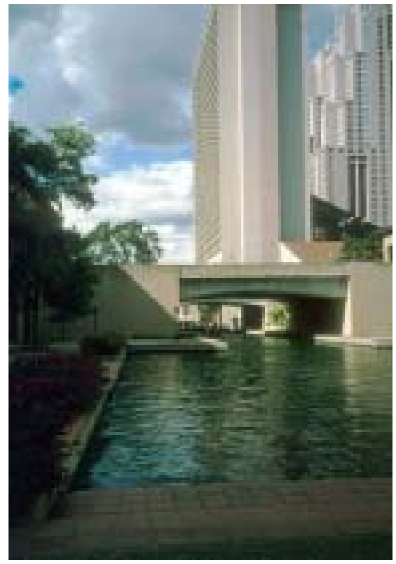

Figure 2.8 New section of Riverwalk in San Antonio, TX.

supplies are not appropriate goals for most heavily developed watersheds. However, these higher uses may be possible in urban areas where the receiving waters are large and drain mostly undeveloped areas.

There are many examples throughout the world where local citizens have recognized the added value that aesthetically pleasing waters contribute to cities. With this recognition comes a local pride in these waters and a genuine desire to improve their condition. In many cases, water has played an important part in the economic renewal of an inner city area. Dreiseitl (1998) states that "stormwater is a valuable resource and opportunity to provide an aesthetic experience for the city dweller while furthering environmental awareness and citizen interest and involvement." He found that water flow patterns observed in nature can be duplicated in the urban environment to provide healthy water systems of potentially great beauty. Without reducing safety, urban drainage elements can utilize water's refractive characteristics and natural flow patterns to create very pleasing urban areas. Successful stormwater management in Germany has been best achieved by using several measures together. Small open drainage channels placed across streets have been constructed of cobbles. These collect and direct the runoff, plus slow automobile traffic and provide dividing lines for diverse urban landscaping elements. The use of rooftop retention and evaporation areas reduce peak flows. Dreiseitl has found that infiltration and retention ponds can also be used to great advantage by providing a visible and enjoyable design element in urban landscapes.

Probably the most famous U.S. example of the economic benefits that water has contributed in an older part of a city is Riverwalk in San Antonio, TX. Many cities would like to emulate Riverwalk, with the great economic benefit that it has provided to San Antonio (Figures 2.7 through 2.9). Riverwalk was conceived and constructed many decades ago, but only in recent years has its full value been realized. Bellingham, WA (Figure 2.10), Austin, TX (Figure 2.11), and Denver, CO (Figures 2.12 through 2.14) are some of the other U.S. cities that have long enjoyed central city urban creek corridors.

Dreiseitl (1998) described the use of stormwater as an important component of the Potsdamer Platz in the center of Berlin. Roof runoff will be stored in large underground cisterns, with some filtered and used for toilet flushing and irrigation. The rest of the roof runoff will flow into a 1.4-ha (3.8-acre) concrete-lined lake in the center of the project area. The small lake provides an important natural element in the center of this massive development and regulates the stormwater discharge rate to the receiving water (Landwehrkanal). The project is also characterized by numerous fountains, including some located in underground parking garages.

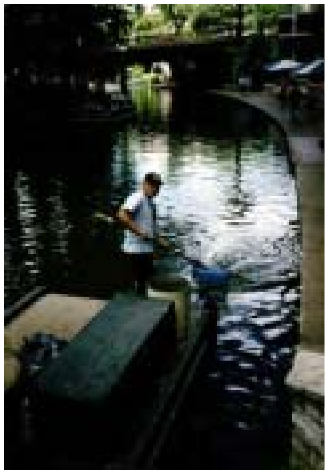

Figure 2.9 0 Litter control along Riverwalk, San Antonio, TX.

Figure 2.10 Bike and walking trail along Watcom Creek, Bellingham, WA.

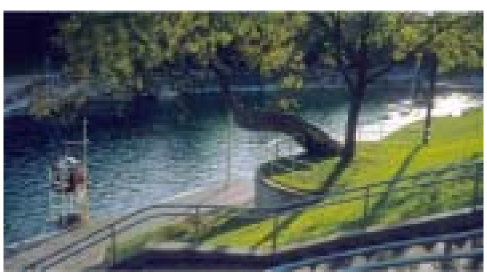

Figure 2.11 0 Barton Springs swimming area, Austin, TX.

Figure 2.12 Cherry Creek walkway, downtown Denver, CO.

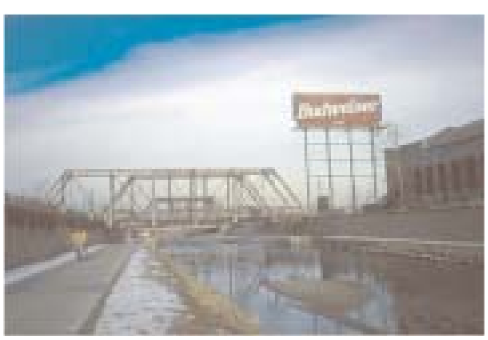

Figure 2.13 Cherry Creek walk in Denver, CO.

Figure 2.14 0 Cherry Creek and Platte River junction in Denver, CO.

Göransson (1998) also described the aesthetic use of stormwater in Swedish urban areas. The main emphasis was to retain the stormwater in surface drainages instead of rapidly diverting it to underground conveyances. Small, sculpted rainwater channels are used to convey roof runoff downspouts to the drainage system. Some of these channels are spiral in form and provide much visual interest in areas dominated by the typically harsh urban environment. Some of these spirals are also formed in infiltration areas and are barely noticeable during dry weather. During rains, increasing water depths extenuate the patterns. Glazed tile, small channels with perforated covers, and geometrically placed bricks with large gaps to provide water passage slightly below the surface help urban dwellers better appreciate the beauty of flowing water.

Tokyo has instituted major efforts to restore historical urban rivers that have been badly polluted, buried, or have had all of their flows diverted. Fujita (1998) describes how Tokyo residents place great value on surface waterways: "Waterfront areas provide urban citizens with comfort and joy as a place to observe nature and to enjoy the landscape." Unfortunately, the extensive urbanization that has taken place in Tokyo over the past several decades has resulted in severe stream degradation, including the disappearance of streams altogether. However, there has recently been a growing demand for the restoration of polluted urban watercourses in Tokyo. This has been accomplished in many areas by improved treatment of sanitary sewage, reductions in combined sewer overflows, and by infiltration of stormwater.

Fujita (1998) repeatedly states the great importance the Japanese place on nature, especially flowing water and the associated landscaping and attracted animals. They are therefore willing to perform what seems to be extraordinary efforts in urban stream recovery programs in one of the world's largest cities. The stream recovery program is but one element of the local efforts to provide a reasonably balanced urban water program. Water reuse and conservation are also important elements in their efforts. Stormwater infiltration to recharge groundwaters and the use of treated wastewaters for beneficial uses (including stream restoration, plus landscaping irrigation, train washing, sewer flushing, fire fighting, etc.) are all important elements of these efforts, although this reuse currently only amounts to about 7% of the total annual water use in Tokyo.

At many U.S. wet detention pond project sites, the stormwater treatment pond is used to increase the value of the property. Figures 2.15 and 2.16 show two examples (in Austin, TX, and in Lake Oswego, OR, respectively). Many people live near wet detention ponds because of the close presence of the wetlands, and their property values are typically greater than lots farther from the ponds (Marsalek et al. 1982). They also reported that small (well-maintained) wet detention ponds are less subject to controversy than larger ponds (that are more commonly neglected). Debo and Ruby (1982) summarized a survey conducted in Atlanta, GA, of residents living near and downstream of 15 small detention ponds and found that almost half the people surveyed who lived in the immediate areas of the ponds did not even know that they existed. Wiegand et al. (1986) found that wet detention ponds, when properly maintained, are preferred by residents over any other urban runoff control practice.

Figure 2.15 0Advertising the benefits of a stormwater pond (Austin, TX).

Figure 2.16 Stormwater pond adding value to apartment complex (Lake Oswego, OR).

Emmerling-DiNovo (1995) reported on a survey of homeowners in the Champaign-Urbana, IL, area living in seven subdivisions having either dry or wet detention ponds. She reported that past studies have recognized that developers are well aware that proximity to water increases the appeal of a development. Detention ponds can create a sense of identity, distinguishing one development from another, and can be prominent design elements. Increased value is important because the added cost of the detention facility, including loss of developable land, must be recovered by increasing the housing costs. Others have also found that the higher costs of developments having stormwater detention facilities can also be offset by being able to sell the housing faster. In a survey in Columbia, MD, 73% of the respondents were found to be willing to pay more for property located in an area having a wet detention pond if designed to enhance fish and wildlife use. Although the residents were concerned about nuisances and hazards, they felt that the benefits outweighed these concerns. In her survey, Emmerling-DiNovo (1995) received 143 completed surveys. Respondents reported that the overall attractiveness of the neighborhood was the most important factor in their decision to purchase their home. Resale value was the second most important factor, while proximity to water was slightly important. More than 74% of the respondents believed that wet detention ponds contributed positively to the image of the neighborhood and that they were a positive factor in choosing that subdivision. In contrast, the respondents living in the subdivisions with dry ponds felt that the dry ponds were not a positive factor for locating in their subdivision. Respondents living adjacent to wet ponds felt that the presence of the pond was very positive in the selection of their specific lot. The lots adjacent to the wet ponds were reported to be worth about 22% more than lots that were not adjacent to the wet ponds. Lots adjacent to the dry ponds were actually worth less (by about 10%) than other lots in two of the three dry basin subdivisions studied. The respondents favored living adjacent to wet ponds even more than next to golf courses. Living adjacent to dry ponds was the least preferred location.

Stormwater Conveyance (Flood Prevention)

This is a basic beneficial use of streams and storm drainage systems that must be considered. Problems are caused by increases in peak runoff flow rates that are associated with large increases in runoff volume and decreases in the drainage time of concentration. Because of high flows during wet weather, it is common for urban streams to have much lower flows during dry weather due to lack of recharge from shallow groundwaters (Color Figure 2.1).* Debris and obstructions in the receiving waters, which assist aquatic life uses, typically degrade flooding and drainage uses and are often cleared to provide better drainage. Other common conflicts are associated with the desire to have homogeneous channels (smooth bottoms and straight alignments) for drainage (Figure 2.17), while aquatic life requires diversity in the channel characteristics. These conflicts must be resolved through comprehensive planning, including source controls and drainage controls that have minimal effects on aquatic life. The best solutions would provide for the necessary flooding and drainage benefits while also providing suitable biological habitat (including improved channel stability, decreased bank erosion, artificial pools and riffle areas, overstory shading, gravel linings, low flow meandering channel alignments, and other refuge areas).

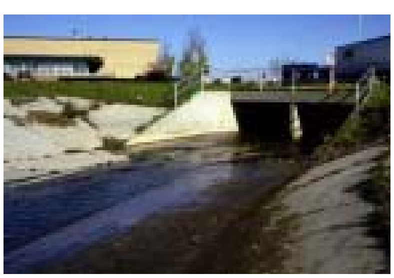

Figure 2.17 Channelized urban stream, Nor-X-Way, Menomonee Falls, WI.

Recreation (Non-water Contact) Uses

This basic beneficial use is concerned with odors, trash, beauty, access, and rapidly fluctuating flows. Safety is an important issue in urban

* Color figures follow p. 370.

Figure 2.18 Degraded stream banks along New York City shoreline.

Figure 2.19 Debris in riparian area, New York City.

Figure 2.20 Algal mats and other floating debris, Orlando, FL.

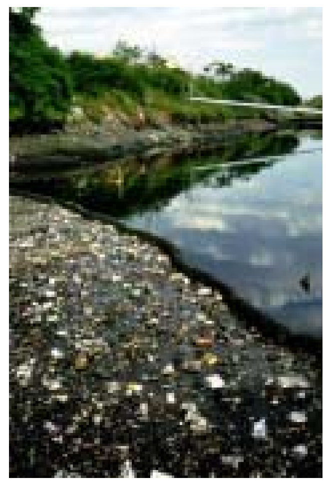

Figure 2.21 Litter controlled behind floating booms, New York City.

areas where children frequently play near small streams. Bank stability and rapidly fluctuating flows are, therefore, of prime importance (Figures 2.18 and 2.19). Many communities have also established linear parks along urban streams as part of their flood control and parks programs. In these cases, aesthetics (trash, odor, and beauty), access (paths and bridges), and the above safety issues are also important. Excessive algal growths, with attendant odors and unsightly conditions, may also occur along stressed urban waterways (Figures 2.20 and Color Figure 2.2). Some simple controls have been instituted in some areas to reduce aesthetic impacts (Figure 2.21). Human health may be an issue if water contact (especially by wading children) or if consumptive fishing occurs. These human health uses will be very difficult to maintain in urban areas.

Biological Uses (Warm-Water Fishery, Aquatic Life Use, Biological Integrity, etc.)

This basic beneficial use is also important, but it is defined differently by different people. It is unreasonable to expect natural receiving water conditions in agricultural or urbanized streams. Some degradation is inevitable. The goal is to have an acceptable diversity of aquatic life and an absence of episodic fish kills, at a minimum. It is unfortunate if sensitive and important species exist in an agricultural or urbanized stream and need special protection, as it is probably unrealistic to believe that it is possible to maintain these species in the absence of dramatic and extensive stormwater controls (which are not likely to occur). The most significant impairments to aquatic life beneficial uses are likely: habitat destruction (including channel and bank instability, sedimentation, and loss of refuge areas and vegetative overstory/canopy), highly fluctuating flow rates, inappropriate dry-weather contaminated discharges (toxicants and pathogens), polluted sediment (toxicants and oxygen-demanding materials), and possibly wet weather water quality degradation. Decreases in groundwater recharge and increased peak flows during periods of storm events are obviously associated with decreased flows during dry periods. Aquatic life undergoes additional stress during periods of low flow due to associated increased water temperatures, decreased pollutant mixing and transport, and simple decreased mobility and forage opportunities.

It may be possible to obtain significant short-term biological beneficial use improvements in a degraded stream with improvements in habitat conditions alone. Longer-term benefits would likely require sediment removal and control, plus the control of inappropriate dry-weather toxic discharges. It is unlikely that large improvements in wet weather water quality would be possible in heavily developed watersheds, nor may it be needed to obtain acceptable (but degraded) biological uses. The retrofitting of stormwater controls to improve wet-weather runoff quality in an urban area is very costly and is limited in effectiveness. However, the basic use of construction site erosion controls and biofiltration/infiltration and sedimentation stormwater controls in newly developing areas should be mandatory to decrease the further degradation of biological conditions in receiving waters.

Human Health-Related Uses (Swimming, Fishing, and Water Supply)

In many areas of the country, urban and agricultural runoff drains into public water supplies, swimming areas, or fisheries. In these cases, additional concerns need to be considered, especially relating to toxicants and pathogens. Public water supplies are frequently affected by upstream wastewater discharges (both point and nonpoint sources) and are designed to reduce and monitor constituents of concern. As upstream discharges increase, water treatment becomes more difficult and costly, with increased probabilities of waterborne disease outbreaks and increased (but "legal") taste and odor problems. Swimming areas in urban receiving waters (large rivers and lakes) have also been more frequently closed to the public because of high bacteria counts for extended periods after rains, and because of other unsafe conditions (Figures 2.22 through 2.25 and Color Figure 2.3). In addition, although fishing in urban and agricultural areas is relatively common (Figures 2.26 and 2.27), many communities are posting fishing advisories to discourage this practice (Figure 2.28).

Figure 2.22 Swimming restriction in urban lake, San Francisco, CA.

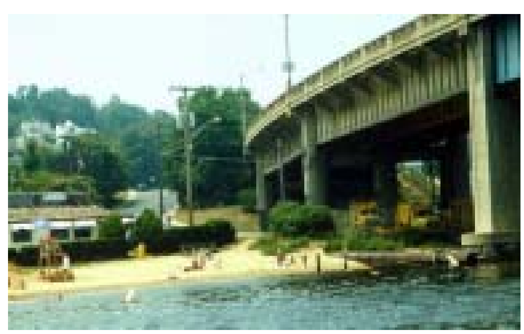

Figure 2.23 Swimming near stormwater outfall, Navesink River, NJ.

Figure 2.24 Children playing in Lincoln Creek, Milwaukee, WI. (Courtesy of Wisconsin Department of Natural Resources.)

Figure 2.25 Floatable trash from CSO and stormwater discharges, New York City.

Figure 2.26 Fishing in urban stream, Birmingham, AL.

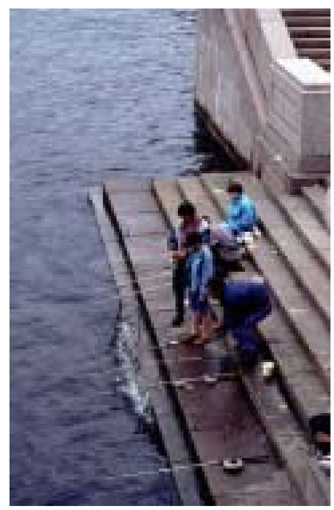

Figure 2.27 Urban fishing in Neva River, St. Petersburg, Russia.

Unfortunately, pathogen levels in stormwater may be high. Fecal coliform levels can be very high, but fecal coliform levels are not thought to be a good indicator of pathogens in stormwater (see also Chapter 4). Direct pathogen monitoring in stormwater has shown very large numbers of some specific pathogens, however, requiring careful consideration for human health issues. In addition, sediments may contain elevated levels of pathogens which live for extended periods following high flow events (Burton et al. 1987). It is very difficult to reduce the high levels using typical stormwater controls. Common disinfection controls are also very costly and may create additional problems associated with trihalomethane production. The consumption of fish or shellfish in waters receiving agricultural and urban runoff is also a cause of concern because of pathogens and toxicants. This has been shown with the recent outbreaks of *Pfiesteria* in nutrient-laden waters of the East Coast. Many of the toxic compounds found in stormwater may readily bioaccumulate in aquatic organisms, and pathogens can also contaminate the aquatic organisms. All of these human health issues require careful study by epidemiologists and public health professionals.

Figure 2.28 Fish advisory for Village Creek, Jefferson Co., AL.

LIKELY CAUSES OF RECEIVING WATER USE IMPAIRMENTS

In general, monitoring of urban and agricultural stormwater runoff has indicated that the biological beneficial uses of receiving waters are most likely affected by habitat destruction and long-term pollutant exposures (especially to macroinvertebrates via contaminated sediment). Pulse exposures to suspended solids and toxicants and contaminated sediments have also been shown to be common in urban and agricultural waterways (see Chapter 6; also review by Burton et al. 2000). Mancini and Plummer (1986) have long been advocates of numeric water quality standards for stormwater that reflect the partitioning of the toxicants and the short periods of exposure during rains. Unfortunately, this approach attempts to isolate individual runoff events and does not consider the accumulative adverse effects caused by the frequent exposures of receiving water organisms to stormwater (Davies 1995; Herricks et al. 1996a,b). Recent investigations have identified acute toxicity problems associated with intermediate-term (about 10 to 20 days) exposures to adverse toxicant concentrations in urban receiving streams (Crunkilton et al. 1996). The most severe receiving water problems may be associated with chronic exposures to contaminated sediment and to habitat destruction.

Heaney et al. (1980) conducted a comprehensive evaluation of the early literature pertaining to urban runoff effects on receiving waters. They found that well-documented cases of receiving water detrimental effects were scarce. Through their review of many reports, they found several reasons to question the implied cause-and-effect relationships between urban runoff and receiving water conditions. Impacts that were attributed to urban runoff were probably caused, in many cases, by other water pollution sources (such as combined sewer overflows, agricultural nonpoint sources, etc.). One of the major difficulties encountered in their study was the definition of “problem” that had been used in the reviewed projects. They found that very little substantive data had been collected to document beneficial use impairments. In addition, urban runoff impacts are most likely to be associated with small receiving waters, while most of the existing urban water quality monitoring information exists for larger bodies of water. It was also very difficult for many researchers to isolate urban runoff effects from other water pollutant sources, such as municipal and industrial wastes. This was especially important in areas that had combined sewers that overflowed during wet weather, contributing to the receiving water impacts during wet-weather conditions.

Claytor (1996a) summarized the approach developed by the Center for Watershed Protection as part of their EPA-sponsored research on stormwater indicators (Claytor and Brown 1996). The 26 stormwater indicators used for assessing receiving water conditions were divided into six broad categories: water quality, physical/hydrological, biological, social, programmatic, and site. These were presented as tools to measure stress (impacting receiving waters), to assess the resource itself, and to indicate stormwater control program implementation effectiveness. The biological communities in Delaware’s Piedmont streams have been severely impacted by stormwater, after the extent of imperviousness in the watersheds exceeded about 8 to 15%, according to a review article by Claytor (1996b). If just conventional water quality measures are used, almost all (87%) of the state’s nontidal streams supported their designated biological uses. However, when biological assessments are included, only 13% of the streams were satisfactory.

MAJOR URBAN RUNOFF SOURCES

Soil erosion from construction sites and increased stormwater runoff generated from newly established urban areas cause significant economic, social, and environmental problems. These problems may result from all land development activities such as subdivision development, individual homesite construction, large-scale construction projects such as shopping centers and industrial sites, highway construction, and public utility construction projects. Problems caused by construction site erosion and stormwater runoff include sediment that destroys fish habitat and fills in lakes; urban runoff volumes and flow rates that increase flooding; nutrient discharges that produce nuisance algae growths; toxic heavy metal and organic discharges that result in inedible fish, undrinkable water, and shifts in aquatic life to more pollution-tolerant species; and pathogenic bacteria discharges that necessitate swimming beach closures.

Erosion losses and downstream sedimentation peak during construction, when soil exposure is greatest, and decline after construction is completed. Thus, while the impacts of erosion and sedimentation may be severe, they are relatively short term in nature for any specific construction site.

Stormwater runoff and pollutant discharges, on the other hand, increase steadily as development progresses and remain at an elevated level for the lifetime of the development. This happens because impervious surfaces such as roads, sidewalks, driveways, rooftops, etc., permanently reduce infiltration of rainfall and runoff into the ground.

Accelerated stormwater runoff rates also occur with development and can significantly increase the water's ability to detach sediment and associated pollutants, to carry them off site, and to deposit them downstream. Increased runoff rates may also cause stream bank and channel erosion. Increased stormwater runoff volumes and flow rates also increase urban flooding and the resultant loss of human life and property.

Urbanization may also affect groundwater adversely. In some cases, polluted stormwater contaminates groundwater. More frequently, impervious surfaces block infiltration of rainfall and runoff that otherwise would recharge groundwater supplies. Reduced infiltration affects not only groundwater levels but also the amount of groundwater-derived stream flow available during low flow periods. From a water quality standpoint, low flow periods are critical because the amount of water available to dilute stream pollutants is at a minimum at those times. Reduced flows during extended dry periods also adversely affect aquatic life.

Urban runoff, which includes stormwater, construction site runoff, snowmelt, and contaminated baseflows, has been found to cause significant receiving water impacts on aquatic life. The effects are obviously most severe for small receiving waters draining heavily urbanized and rapidly developing watersheds. However, some studies have shown important aquatic life impacts for streams in watersheds that are less than 10% urbanized.

In order to best identify and understand these impacts, it is necessary to include biological monitoring (using a variety of techniques) and sediment quality analyses in a monitoring program. Water column testing alone has been shown to be very misleading. Most aquatic life impacts associated with urbanization are probably related to chronic long-term problems caused by habitat destruction, polluted sediments, and food web disruption. Transient water column quality conditions associated with urban runoff probably rarely cause significant direct aquatic life acute impacts.

The underlying theme of many researchers is that an adequate analysis of receiving water biological impacts must include investigations of a number of biological organism groups (fish, benthic macroinvertebrates, algae, rooted macrophytes, etc.) in addition to studies of water and sediment quality. Simple studies of water quality alone, even with possible comparisons with water quality criteria for the protection of aquatic life, are usually inadequate to predict biological impacts associated with urban runoff.

Duda et al. (1982) presented a discussion on why traditional approaches for assessing water quality, and selecting control options, in urban areas have failed. The main difficulties of traditional approaches when applied to urban runoff are the complexity of pollutant sources, wet weather monitoring problems, and limitations when using water quality standards to evaluate the severity of wet weather receiving water problems. They also discuss the difficulty of meeting water quality goals (that were promulgated in the Water Pollution Control Act of 1972) in urban areas.

Relationships between observed receiving water biological effects and possible causes have been especially difficult to identify, let alone quantify. The studies reported in this chapter have identified a wide variety of possible causative agents, including sediment contamination, poor water quality (low dissolved oxygen, high toxicants, etc.), and factors affecting the physical habitat of the stream (high flows, unstable stream beds, absence of refuge areas, etc.). It is expected that all of these factors are problems, but their relative importance varies greatly depending on the watershed and receiving water conditions. Horner (1991), as an example, notes that many watershed, site, and organism-specific factors must be determined before the best combination of runoff control practices to protect aquatic life can be determined.

Construction Site Erosion Characterization

Sediment is, by weight, the greatest pollutant of water resources. Willett (1980) estimated that approximately 5 billion tons of sediment reach U.S. surface waters annually, of which 30% is generated by natural processes and 70% by human activities. Half of this 70% is attributed to eroding croplands. Although urban construction accounts for only 10%, this amount equals the combined contributions of forestry, mining, industrial, and commercial activities (Willett 1980; Virginia 1980).

Construction accounts for a much greater proportion of the sediment load in urban areas — sometimes more than 50% — than it does in the nation as a whole. Urban areas experience large sediment loads from construction site erosion because construction sites usually have extremely high erosion rates and because urban construction sites are efficiently drained by stormwater drainage systems. Construction sites at most U.S. locations have an erosion rate of approximately 20 to 200 tons per acre per year, a rate that is about 3 to 100 times that of croplands. Construction site erosion losses vary greatly depending on local rain, soil, topographic, and management conditions. As an example, the Birmingham, AL, area may have some of the highest erosion rates in the nation because of its combination of very high-energy rains, moderately erosive soils, and steep topography. The typically high erosion rates mean that even a small construction project may have a significant detrimental effect on local water bodies. While construction occurs on only about 0.007% of U.S. land, it accounts for about 10% of the sediment load to U.S. surface waters (Willett 1980).

Data from the highly urbanized Menomonee River watershed in southeastern Wisconsin illustrate the impact of construction on water quality. These data indicate that construction sites have much greater potential for generating sediment and phosphorus than do areas in other land uses (Chesters et al. 1979). For example, construction sites can generate approximately 8 times more sediment and 18 times more phosphorus than industrial sites, the land use that contributes the second highest amount of these pollutants, and 25 times more sediment and phosphorus than row crops. In fact, construction contributes more sediment and phosphorus to the river than any other land use. In 1979, construction comprised only 3.3% of the watershed's total land area, but it contributed about 50% of the suspended sediment and total phosphorus loading at the river mouth (Novotny et al. 1979).

Similar conclusions were reported by the Southeastern Wisconsin Regional Planning Commission in a 1978 modeling study of the relative pollutant contributions of 17 categories of point and nonpoint pollution sources to 14 watersheds in the southeast Wisconsin regional planning area (SEWRPC 1978). This study revealed construction as the first or second largest contributor of sediment and phosphorus in 12 of the 14 watersheds. Although construction occupied only 2% of the region's total land area in 1978, it contributed approximately 36% of the sediment and 28% of the total phosphorus load to inland waters, making construction the region's second largest source of sediment and phosphorus. The largest source of sediment was estimated to be cropland; livestock operations were estimated to be the largest source of phosphorus. By comparison, cropland comprised 72% of the region's land area and contributed about 45% of the sediment and only 11% of the phosphorus to regional watersheds. This study again points out the high pollution-generating ability of construction sites and the significant water quality impact a small amount of construction may have on a watershed.

A monitoring study of construction site runoff water quality in the Village of Germantown (Washington County, WI) yielded similar results (Madison et al. 1979). Several large subdivisions being developed with single and multifamily residences were selected for runoff monitoring. All utility construction, including the storm drainage system and streets, was completed before monitoring began.

Analysis of the monitoring data showed that sediment leaving the developing subdivisions averaged about 25 to 30 tons per acre per year (Madison et al. 1979). Construction practices identified as contributing to these high yields were removing surface vegetation; stripping and stockpiling topsoil; placing large, highly erodible mounds of excavated soil on and near the streets; pumping water from flooded basement excavations; and tracking of mud into the streets by construction vehicles. If the amount of sediment leaving the sites during utility development had been added in, the total amount of eroded sediment leaving the site would have been substantially greater.

Analysis of the Germantown data also showed that the amount of sediment leaving areas undergoing development is a function of the extent and duration of development and is independent of the type of development. In other words, there is no difference in the per acre sediment loads produced by single-family or multifamily construction. This finding is significant because local and state regulatory programs sometimes exempt single-family home construction from erosion control requirements.

Almost all eroded sediment from the Germantown construction areas entered the receiving waters. The delivery of sediment to the receiving waters was found to be nearly 100% when 10% or more of the watershed was experiencing development. The smallest delivery value obtained during the Germantown monitoring was 50%, observed when only 5% of the watershed was undergoing development. These high delivery values occurred (even during periods with small amounts of development) because storm drainage systems, which efficiently transport water and its sediment load, had been installed during an early stage of development.

Local Birmingham, AL, erosion rates from construction sites can be 10 times the erosion rates from row crops and 100 times the erosion rates from forests or pastures (Nelson 1996). The site-specific factors affecting construction site erosion include:

- Rainfall energy (Alabama has the highest in the nation)
- Soil erodibility (northern part of the state has fine-grained, highly erosive soils)
- Site topography (northeastern part of the state has steep hills under development)
- Surface cover (usually totally removed during initial site grading)

The rain energy is directly related to rainfall intensity, and the rainfall erosion index varies from 250 to 550+ for Alabama (most of the state is about 350), which is the highest in the United States. The months having the greatest erosion potential are February and March, while September through November have the lowest erosion potential. Nelson (1996) monitored sediment quantity and particle size from 70 construction site runoff samples from the Birmingham area. He measured suspended solids concentrations ranging from 100 to more than 25,000 mg/L (overall median about 4000 mg/L), while the turbidity values ranged from about 300 to >50,000 NTU (average of about 4000 NTU). About 90% of the particles (by mass) were smaller than about 20 μm (0.02 mm) in diameter, and the median size was about 5 μm (0.005 mm). The local construction site erosion discharges were estimated to be about 100 tons/acre/year. Table 2.2 summarizes the measured suspended solids and median particle sizes as a function of rain intensity. High-intensity rains were found to have the most severe erosion problems, as expected, with much greater suspended solids

Table 2.2 Birmingham (AL) Construction Site Erosion Runoff Characteristics

	Low-Intensity Rains (<0.25 in/hr)	Moderate-Intensity Rains (about 0.25 in/hr)	High-Intensity Rains (>1 in/hr)
Suspended solids, mg/L	400	2000	25,000
Particle size (median), µm	3.5	5	8.5

Data from Nelson, J. *Characterizing Erosion Processes and Sediment Yields on Construction Sites.* M.S.C.E. thesis. Department of Civil and Environmental Engineering, University of Alabama at Birmingham. 94 pp. 1996.

concentrations. Typical small particle sizes of erosion particulates make it very difficult to remove these particulates after they have been eroded from the site. The extreme turbidity values also cause very high in-stream turbidity conditions in local receiving waters for great distances downstream of eroding sites.

Urban Runoff Contaminants

Urban runoff is comprised of many different flow phases. These may include dry-weather base flows, stormwater runoff, combined sewer overflows (CSOs), and snowmelt. The relative magnitudes of these discharges vary considerably, based on a number of factors. Season (such as cold vs. warm weather, or dry vs. wet weather) and land use have been identified as important factors affecting baseflow and stormwater runoff quality.

Land development increases stormwater runoff volumes and pollutant concentrations. Impervious surfaces, such as rooftops, driveways, and roads, reduce infiltration of rainfall and runoff into the ground and degrade runoff quality. The most important hydraulic factors affecting urban runoff volume (and therefore the amount of water available for groundwater infiltration) are the quantity of rain and the extent of impervious surfaces directly connected to a stream or drainage system. Directly connected impervious areas include paved streets, driveways, and parking areas draining to curb and gutter drainage systems, and roofs draining directly to a storm or combined sewer pipe. Table 2.3 presents older stormwater quality data (APWA 1969), while Table 2.4 is a summary of the Nationwide Urban Runoff Program (NURP) stormwater data collected from about 1979 through 1982 (EPA 1983). The NURP data are the most comprehensive stormwater data available from throughout the nation. The recently collected data for the stormwater NPDES permits is a potentially large and important database of information, but it has not been made conveniently available. Land use and source areas (parking areas, rooftops, streets, landscaped areas, etc.) all have important effects on stormwater runoff quality. BOD_5 bacteria and nutrient concentrations in stormwater are lower than in raw sanitary wastewater. However, urban stormwater still has relatively high concentrations of bacteria, along with high concentrations of many metallic and some organic toxicants.

NURP found that stormwater pollutant concentrations, runoff volumes, and therefore annual pollutant yields often vary with land use. Although inconsistencies in local development practices within a single land use category make land use a less than perfect indicator of urban runoff characteristics, land use must serve as a surrogate for more appropriate indicators because development data are typically reported in land use categories. The amount of directly connected impervious area is a very good indicator of an area's runoff volume. The extent of "effective" impervious surfaces, however, is a function of local development customs (lot sizes, use of swale drainages, single or multilevel buildings, type of landscaping, etc.), which can vary significantly within a single land use category (such as medium-density residential). Development characteristics are not uniform throughout a region, and they may also vary by age of development or location within a single city.

Bannerman et al. (1979) found a high correlation between pollutant loading values and percent connected-imperviousness during monitoring of seven subwatersheds of the Menomonee River basin: pollutant loading to the river increased as the extent of impervious areas directly connected to the storm drainage system increased. Although larger amounts of runoff and pollutants were

Table 2.3 Characteristics of Stormwater Runoff from Early Studies

City	BOD_5 (mg/L)	Total Solids (mg/L)	Suspended Solids (mg/L)	Chlorides (mg/L)	COD (mg/L)
East Bay Sanitary District:					
Oakland, California					
Minimum	3	726	16	300	
Maximum	7700		4400	10,260	
Average	87	1401	613	5100	
Cincinnati, Ohio					
Maximum Seasonal Means	12	260			110
Average	17		227		111
Los Angeles County					
Average 1962–63	161	2909		199	
Washington, D.C.					
Catch-basin samples during					
storm	6		26	11	
Minimum	625		36,250	160	
Maximum	126		2100	42	
Average					
Seattle, Washington	10				
Oxney, England	100[a]	2045			
Moscow, Russia	186–285	1000–3500[a]			
Leningrad, Russia	36	14,541			
Stockholm, Sweden	17–80	30–8000			18–3100
Pretoria, South Africa					
Residential	30				29
Business	34				28
Detroit, Michigan	96–234	310–914	102–213[b]		

[a] Maximum.
[b] Mean.

From APWA (American Public Works Association). *Water Pollution Aspects of Urban Runoff*. Water Pollution Control Research Series WP-20-15, Federal Water Pollution Control Administration. January 1969.

Table 2.4 Median Stormwater Pollutant Concentrations for All Sites by Land Use

Constituent	Residential		Mixed Land Use		Commercial		Open/Non-urban	
	Median	COV[a]	Median	COV	Median	COV	Median	COV
BOD_5, mg/L	10	0.41	7.8	0.52	9.3	0.31	—	—
COD, mg/L	73	0.55	65	0.58	57	0.39	40	0.78
TSS, mg/L	101	0.96	67	1.14	69	0.85	70	2.92
Total Kjeldahl nitrogen, μg/L	1900	0.73	1288.8	0.50	1179	0.43	965	1.00
NO_2 + NO_3 (as N) μg/L	736	0.83	558	0.67	572	0.48	543	0.91
Total P, μg/L	383	0.69	263	0.75	201	0.67	121	1.66
Soluble P, μg/L	143	0.46	56	0.75	80	0.71	26	2.11
Total lead, μg/L	144	0.75	114	1.35	104	0.68	30	1.52
Total copper, μg/L	33	0.99	27	1.32	29	0.81	—	—
Total zinc, μg/L	135	0.84	154	0.78	226	1.07	195	0.66

[a] COV: coefficient of variation = standard deviation/mean.

From EPA (U.S. Environmental Protection Agency). *Results of the Nationwide Urban Runoff Program*. Water Planning Division, PB 84-185552, Washington, D.C. December 1983.

generated in low-density residential areas, compared to undisturbed areas, runoff and pollutant delivery from the source areas to streams was still low due to the use of grass-lined roadside drainage channels. Soil and vegetation have a greater chance to reduce runoff water and pollutants in areas drained by grass-lined drainage channels than in similar areas drained by conventional curb-and-gutter drainage systems.

Table 2.5 presents estimates of typical urban area pollutant yields from several separate studies. Local conditions and development characteristics significantly affect these estimates. The most significant factor is the drainage efficiency of the areas, specifically if the areas are drained by grass swales. The low-density residential area values shown on this table reflect grass swale drained areas. If conventional curbs and gutters were used instead of grass swales, the yields would be about 10 times greater. Other important development characteristics affecting runoff yields include roof drainage connections and the presence of alleyways. Increased drainage efficiency invariably leads to increased pollutant discharges.

A number of urban runoff monitoring projects (such as EPA 1983; Pitt and McLean 1986) have found inorganic and organic hazardous and toxic substances in urban runoff. The NURP data, collected from mostly residential areas throughout the United States, did not indicate any regional differences in the substances detected, or in their concentrations. However, residential and industrial data obtained by Pitt and McLean (1986) in Toronto found significant concentration and yield differences for these two land uses and for dry weather and wet weather urban runoff flows.

Tables 2.6 and 2.7 list the toxic and hazardous organic substances that have been found in greater than 10% of industrial and residential urban runoff samples. NURP data do not reveal toxic urban runoff conditions significantly different for different geographical areas throughout North America (EPA 1983). The pesticides shown were mostly found in urban runoff from residential areas, while other hazardous materials were much more prevalent in industrial areas. Urban runoff dry weather baseflows may also be important contributors of hazardous and toxic pollutants.

Urban Runoff Pollutant Sources

Sources of the toxic and hazardous substances found in urban runoff vary widely. Table 1.3 listed the major expected sources of these substances. Automobile use contributes significantly to many of these materials. Polycyclic aromatic hydrocarbons (PAHs), the most commonly detected toxic organic compounds found in urban runoff, are mostly from fossil fuel combustion. Phthalate esters, another group of relatively common toxic organic compounds, are derived from plastics. Pentachlorophenol, also frequently found, comes from preserved wood. Such compounds are very hard to control at their sources, and, unfortunately, their control by typical stormwater management practices is little understood.

Urban runoff includes warm and cold weather baseflows, stormwater runoff, and snowmelt. Table 2.8 shows median concentrations of some of the pollutants monitored in a mixed residential and commercial catchment and from an industrial area in Toronto, Ontario, for these different flow phases (Pitt and McLean 1986). Samples were obtained from baseflow discharges, stormwater runoff, and snowmelt. The baseflows had surprisingly high concentrations of several pollutants, especially dissolved solids (filtrate residue) and fecal coliforms from the residential catchment. The concentrations of some constituents in the stormwater from the industrial watershed were typically much greater than the concentrations of the same constituents in the residential stormwater. The industrial warm weather baseflows were also much closer in quality to the industrial stormwater quality than the residential baseflows were to the residential stormwater quality. The data collected for pesticides and PCBs indicate that the industrial stormwater and baseflows typically contained much greater concentrations of these pollutants than the residential waters. Similarly, the more commonly analyzed heavy metals were also more prevalent in the industrial stormwater. However, herbicides were only detected in residential urban runoff, especially the baseflows.

During cold weather, the increases in filtrate residue were quite apparent for both study catchments and for both baseflows and snowmelt. These increases were probably caused by high chlorides from road salt applications. In contrast, bacteria populations were noticeably lower in all outfall discharges during cold weather. Few changes were noted in concentrations of nutrients and heavy metals at the outfall, between cold- and warm-weather periods.

Table 2.5 Typical Urban Area Pollutant Yields (lb/acre/year or kg/ha/yr)[a]

Land Use	Total Solids	Suspended Solids	Chloride	Total Phosphorus	TKN	NH_3	NO_3 plus NO_2	BOD_5
Commercial	2100	1000	420	1.5	6.7	1.9	3.1	62
Parking lot	1300	400	300	0.7	5.1	2.0	2.9	47
High-density residential	670	420	54	1.0	4.2	0.8	2.0	27
Medium-density residential	450	250	30	0.3	2.5	0.5	1.4	13
Low-density residential[b]	65	10	9	0.04	0.3	0.02	0.1	1
Freeways	1700	880	470	0.9	7.9	1.5	4.2	NA[b]
Industrial	670	500	25	1.3	3.4	0.2	1.3	NA
Parks	NA[c]	3	NA	0.03	NA	NA	NA	NA
Shopping center	720	440	36	0.5	3.1	0.5	1.7	NA

Land Use	COD	Lead[d]	Zinc	Chromium	Copper	Cadmium	Arsenic
Commercial	420	2.7	2.1	0.15	0.4	0.03	0.02
Parking lot	270	0.8	0.8	NA	0.06	0.01	NA
High-density residential	170	0.8	0.7	NA	0.03	0.01	NA
Medium-density residential	50	0.05	0.1	0.02	0.03	0.01	0.01
Low-density residential[e]	7	0.01	0.04	0.002	0.01	0.001	0.001
Freeways	NA	4.5	2.1	0.09	0.37	0.02	0.02
Industrial	200	0.2	0.4	0.6	0.10	0.05	0.04
Parks	NA	0.005	NA	NA	NA	NA	NA
Shopping center	NA	1.1	0.6	0.04	0.09	0.01	0.02

[a] The difference between lb/acre/year and kg/ha/yr is less than 15%, and the accuracy of the values shown in this table cannot differentiate between such close values.
[b] The monitored low-density residential areas were drained by grass swales.
[c] NA = Not available.
[d] The lead unit area loadings shown on this table are currently expected to be significantly less than shown on this table, as these values are from periods when leaded gasoline adversely affected stormwater lead quality.
[e] The monitored low-density residential areas were drained by grass swales.

Data from Bannerman et al. (1979, 1983); Madison et al. (1979); EPA (1983); Pitt and McLean (1986).

Table 2.6 Hazardous Substances Observed in Urban Runoff

Hazardous Substances	Residential Areas	Industrial Areas
Benzene	5 µg/L	5 µg/L
Chlordane	17 ng/L	—
Chloroform	—	5 µg/L
Dieldrin	2 to 6 ng/L	—
Endrin	44 ng/L	—
Methoxychlor	20 ng/L	—
Pentachlorophenol	70 to 280 ng/L	50 to 710 ng/L
Phenol	1 µg/L	4 µg/L
Phosphorus	0.1 mg/L	0.5 µg/L
Toluene	—	5 µg/L

Data from EPA 1983; Pitt and McLean 1986 (Toronto); and Pitt et al. 1996 (Birmingham).

Table 2.7 Other Toxic Substances Observed in Urban Runoff

GC/MS Volatiles	Residential Areas	Industrial Areas
1,2-Dichloroethane	—	6 µg/L
Methylene chloride	—	5 µg/L
Tetrachloroethylene	—	High in some source areas
GC/MS Base/Neutrals		
Bis (2-ethylene) phthalate	8 µg/L	18 µg/L
Butyl benzyl phthalate	5 µg/L	58 µg/L
Diethyl phthalate	—	20 µg/L
Di-N-butyl phthalate	3 µg/L	4 µg/L
Isophorone	2 µg/L	—
N-Nitrosodimethylamine	—	3 µg/L
Phenanthrene	—	High in some source areas
Pyrene	—	High in some source areas
GC/MS Pesticides		
BHC	up to 20 ng/L	—
Chlordane	up to 15 ng/L	—
Dieldrin	up to 6 ng/L	—
Endosulfan sulfate	up to 10 ng/L	—
Endrin	up to 45 ng/L	—
PCB-1254	—	up to 630 ng/L
PCB-1260	—	up to 440 ng/L

Data from EPA 1983; Pitt and McLean 1986 (Toronto); and Pitt et al. 1996 (Birmingham).

Table 2.9 compares the estimated annual discharges from the residential and industrial catchments during the different runoff periods. The unit area annual yields for many of the heavy metals and nutrients are greater from the industrial catchment. Industrial catchments contribute most of the chromium to the local receiving waters, and approximately equal amounts with the residential and commercial catchments for phosphorus, chemical oxygen demand, copper, and zinc. This table also shows the great importance of warm weather baseflow discharges to the annual urban runoff pollutant yields, especially for industrial areas. Cold weather bacteria discharges are insignificant when compared to the warm weather bacteria discharges, but chloride (and filtrate residue) loadings are much more important during cold weather.

Table 2.10 shows the fraction of the annual estimated yields for different warm and cold periods (warm weather baseflow, stormwater flows, cold weather baseflow, and snowmelt). Typical storm-

Table 2.8 Median Urban Runoff Pollutant Concentrations

Constituent	Warm-Weather Baseflow Residential	Warm-Weather Baseflow Industrial	Warm-Weather Stormwater Residential	Warm-Weather Stormwater Industrial
Total residue	979	554	256	371
Filterable residue	973	454	230	208
Particulate residue	<5	43	22	117
Total phosphorus	0.09	0.73	0.28	0.75
Total Kjeldahl N	0.9	2.4	2.5	2.0
Phenolics (μg/L)	<1.5	2.0	1.2	5.1
COD	22	108	55	106
Fecal coliforms (no./100 mL)	33,000	7000	40,000	49,000
Fecal streptococci (no./100 mL)	2300	8800	20,000	39,000
Chromium	<0.06	0.42	<0.06	0.32
Copper	0.02	0.045	0.03	0.06
Lead	<0.04	<0.04	<0.06	0.08
Zinc	0.04	0.18	0.06	0.19
Constituent	**Cold-Weather Baseflow Residential**	**Cold-Weather Baseflow Industrial**	**Cold-Weather Melting Periods Residential**	**Cold-Weather Melting Periods Industrial**
Total residue	2230	1080	1580	1340
Filterable residue	2210	1020	1530	1240
Particulate residue	21	50	30	95
Total phosphorus	0.18	0.34	0.23	0.50
Total Kjeldahl N	1.4	2.0	1.7	2.5
Phenolics (μg/L)	2.0	7.3	2.5	15.0
COD	48	68	40	94
Fecal coliforms (no./100 mL)	9800	400	2320	300
Fecal streptococci (no./100 mL)	1400	2400	1900	2500
Chromium	<0.01	0.24	<0.01	0.35
Copper	0.015	0.04	0.04	0.07
Lead	<0.06	<0.04	0.09	0.08
Zinc	0.065	0.15	0.12	0.31

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water flow contributions from these separate stormwater outfalls were only about 20 to 30% of the total annual discharges (by volume). Baseflows contributed the majority of flows. Many constituents were also contributed mostly by snowmelt and baseflows, with the stormwater contributions being less than 50% of the total annual yields. The ratios of expected annual pollutant yields from the industrial catchment divided by the yields from the residential/commercial catchment can be high, as summarized below.

Ratios of Industrial to Mixed Residential/Commercial Unit Area Yields	
Particulate residue (suspended solids)	4.4
Phosphorus	3.0
Phosphates	5.1
Chemical oxygen demand	2.0
Fecal streptococci bacteria	2.6
Chromium	53.0
Zinc	2.5

The only constituents with annual unit area yields that were lower in the industrial catchment than in the mixed residential/commercial catchment were chloride and filtrate residue (dissolved

Table 2.9 Monitored Annual Pollutant Discharges for Toronto's Humber River Watershed Test Sites

		Thistledowns (Residential/Commercial)					Emery (Industrial)						
		Warm		Cold			Warm		Cold				
Constituent	Units	Base-flow	Storm-water	Base-flow	Melt-water	Approx. Total	Base-flow	Storm-water	Base-flow	Melt-water	Approx. Total	Approx. Indus. to Resid. Total Yield Ratios	Weighted Indus. to Resid. Total Yield Ratios[a]
Runoff	m^3/ha	1700	950	1100	1800	5600	2100	1500	660	830	5100	0.9	0.3
Total residue	kg/ha	1700	240	2400	1700	6100	1100	670	710	1500	4000	0.7	0.2
Chlorides	kg/ha	480	33	1200	720	2400	160	26	310	700	1200	0.5	0.2
Total P	g/ha	150	290	200	570	1200	1500	1300	220	540	3600	3.0	1.0
Total Kjeldahl N	g/ha	1500	2800	1500	3500	9300	4900	3400	1300	2800	12,000	1.3	0.4
Phenolics	g/ha	<2.6	1.2	2.3	23	26	4.1	8.1	4.8	14	31	1.2	0.4
COD	kg/ha	38	51	52	130	270	220	170	45	91	530	2.0	0.7
Chromium	g/ha	<100	21	<10	15	36	860	600	160	290	1900	50	18
Copper	g/ha	35	30	16	77	160	92	120	26	76	310	1.9	0.7
Lead	g/ha	<70	41	<70	170	210	<75	170	<25	150	320	1.5	0.5
Zinc	g/ha	70	74	70	270	480	370	430	100	350	1200	2.5	0.8
Fecal coliform	10^9 org/ha	560	480	110	62	1200	144	760	3	6	910	0.8	0.3

"Warm weather" is for the period from about March 15 through December 15, while "cold weather" is for the period from about December 15 through March 15.

[a] The Humber River basin is about 25% industrial and 75% residential and commercial.

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Table 2.10 Major Concentration Periods by Parameter

	Runoff Volume		Total Residue		Filtrate Residue		Particulate Residue		Chlorides	
	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial
Warm baseflow	31%	41%	28%	28%	28%	30%	4%	16%	20%	13%
Stormwater	17	29	4	17	4	10	18	53	1	2
Cold baseflow	20	13	40	18	40	18	14	5	49	26
Meltwater	33	16	29	38	27	41	63	26	29	58

	Phosphorus		Phosphate		Total Kjeldahl Nitrogen		Ammonia Nitrogen	
	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial
Warm baseflow	12	42	—	35	16	39	—	—
Stormwater	24	36	24	51	30	27	21	24
Cold baseflow	16	6	—	—	16	10	—	—
Meltwater	47	15	76	14	38	23	78	76

	Phenolics		COD		Fecal Coliform		Fecal Streptococci		*Pseudomonas aeruginosa*	
	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial
Warm baseflow	—	13	14	42	46	16	12	20	53	41
Stormwater	5	27	19	32	40	84	61	73	46	58
Cold baseflow	9	16	19	9	9	—	4	2	1	—
Meltwater	87	45	48	17	5	—	22	4	—	1

	Chromium		Copper		Lead		Zinc	
	Residential	Industrial	Residential	Industrial	Residential	Industrial	Residential	Industrial
Warm baseflow	—	45	22	29	—	—	14	30
Stormwater	59	31	19	38	19	54	15	35
Cold baseflow	—	8	10	8	—	—	14	8
Meltwater	41	16	49	24	81	46	56	27

Warm period included samples from Thistledowns from July 28 through Nov. 15, 1983, and from Emery from May 14 through Nov.15, 1983. Cold period samples from Thistledowns were from Feb. 2 through March 25, 1984, and from Emery from Jan. through March 22, 1984.

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solids). The annual unit area yields from the residential/commercial catchment were approximately twice the annual unit area yields from the industrial catchment for these constituents.

If only warm weather stormwater runoff is considered (and not baseflows and snowmelts), then significant yield and control measure selection errors are probable. Residential/commercial unit area annual yields for total residue (total solids) for stormwater alone are approximately 240 kg/ha, compared with approximately 670 kg/ha for the industrial catchment. These yields are similar to yields reported elsewhere for total annual total residue unit area yields. However, these warm weather stormwater runoff yields only contributed approximately 5 to 20% of the total annual total residue yields for these study catchments. Annual yields of several constituents were dominated by cold weather processes irrespective of the land use monitored. These constituents include total residue, filtrate residue, chlorides, ammonia nitrogen, and phenolics. The only constituents for which the annual yields were dominated by warm weather processes, irrespective of land use, were bacteria (fecal coliforms, fecal streptococci, and *Pseudomonas aeruginosa*), and chromium. Lead and zinc were both dominated by either stormwater or snowmelt runoff, with lower yields of these heavy metals originating from baseflows.

Warm weather stormwater runoff alone was the most significant contributor to the annual yields for a number of constituents from the industrial catchment. These constituents included particulate residue, phosphorus, phosphates, the three bacteria types, copper, lead, and zinc. In the residential/commercial catchment, only fecal streptococcus bacteria and chromium were contributed by warm weather stormwater runoff more than by the other three sources of water shown. Either warm or cold weather baseflows were most responsible for the yields of many constituents from the industrial catchment. These constituents included runoff volume, phosphorus, total Kjeldahl nitrogen, chemical oxygen demand, and chromium. Important constituents that have high yields in the baseflow from the residential/commercial catchment included total residue, filtrate residue, chlorides, and fecal coliform and *P. aeruginosa* bacteria. More recently, agricultural pesticides have been detected in urban rainfall and urban pesticides in agricultural rainfall and have also been detected in receiving waters.

SUMMARY

This chapter reviewed some of the major receiving water use impairments that have been associated with urban stormwater discharges. The problems associated with urban stormwater discharges can be many, but varied, depending on the specific site conditions. It is therefore important that local objectives and conditions be considered when evaluating local receiving water problems. There has been a great deal of experience in receiving water assessments over the past decade, especially focusing on urban nonpoint source problems. The main purpose of this book is to provide techniques and direction that can be applied to local waters to assess problems based on actual successful field activities. Of course, monitoring and evaluation techniques are constantly changing and improving, and this book also periodically presents short summaries of emerging techniques that hold promise, but may require additional development to be easily used by most people.

Generally, receiving water problems are not readily recognized or understood if one relies on only a limited set of tools. It is critical that conventional water quality measurements be supplemented with habitat evaluations and biological studies, for example. In many cases, receiving water problems caused by urbanization may be mostly associated with habitat destruction, contaminated sediment, and inappropriate discharges, all of which would be poorly indicated by relying only on conventional water quality measurements. In contrast, eliminating water quality measurements from an assessment and relying only on less expensive indicators, such as the currently popular citizen monitoring of benthic conditions, is also problematic, especially from a human health perspective.

A well-balanced assessment approach is therefore needed to understand the local problems of most concern and is the focus of this book.

This chapter also summarized stormwater characteristics. Runoff from established urban areas may not be the major source of some of the problem pollutants in urban areas. Obviously, construction site runoff is typically the major source of sediment in many areas, but snowmelt contributions of sediment (and many other constituents) is also very important in northern areas. Dry weather flows in separate storm drainage systems can be contaminated with inappropriate discharges from commercial and industrial establishments and sewage. Obviously, these inappropriate discharges need to be identified and corrected.

The rest of this book establishes an approach for investigating receiving water use impairments and in identifying the likely causes for these problems. When this information is known, it is possible to begin to develop an effective stormwater management program.

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CHAPTER 3

Stressor Categories and Their Effects on Humans and Ecosystems

"As for Paris, within the last few years, it has been necessary to move most of the mouths of the sewers down stream below the last bridge."

Victor Hugo, 1862

CONTENTS

EFFECTS OF RUNOFF ON RECEIVING WATERS

Many studies have shown the severe detrimental effects of urban and agricultural runoff on receiving waters. These studies have generally examined receiving water conditions above and below a city, by comparing two parallel streams, or by comparing to an ecoregion reference. However, only a few studies have examined direct cause-and-effect relationships of runoff for receiving water aquatic organisms (Heaney and Huber 1984; Burton and Moore 1999; Werner et al. 2000; Vlaming et al. 2000; Bailey et al. 2000; Wenholz and Crunkilton 1995). Chapter 4 presents several case studies representing the major approaches to assessing receiving water problems, while this chapter presents a review of the major stressor categories and summarizes their observed effects.

Indicators of Receiving Water Biological Effects and Analysis Methodologies

There are a number of useful, well-proven tools that can detect adverse biological effects in receiving waters (see also Chapter 6). When these tools are used correctly and combined in the proper framework, they can be used to identify runoff-related problems. Kuehne (1975) studied the usefulness of aquatic organisms as indicators of pollution. He found that invertebrate responses are indicative of pollution for some time after an event, but they may not give an accurate indication of the nature of the pollutants. In-stream fish studies were not employed as biological indicators much before 1975, but they are comparable in many ways to invertebrates as quality indicators and can be more easily identified. However, because of better information pertaining to invertebrates and due to their limited mobility, certain invertebrate species may be sensitive to minor changes in water quality. Fish can be highly mobile and cover large sections of a stream, as long as their passage is not totally blocked by adverse conditions. Fish disease surveys were also used during the Bellevue, WA, urban runoff studies as an indicator of water quality problems (Scott et al. 1982; Pitt and Bissonnette 1984). McHardy et al. (1985) examined heavy metal uptake in green algae (*Cladophora glomerata*) from urban runoff for use as a biological monitor of specific metals.

It is necessary to use a range of measurement endpoints to characterize ecosystem quality in systems that receive multiple stressors (Marcy and Gerritsen 1996; Baird and Burton 2001). Dyer and White (1996) examined the problem of multiple stressors affecting toxicity assessments. They felt that field surveys can rarely be used to verify simple single parameter laboratory experiments. They developed a watershed approach integrating numerous databases in conjunction with *in situ* biological observations to help examine the effects of many possible causative factors (see also Chapter 6).

The interactions of stressors such as suspended solids and chemicals can be confounding and easily overlooked. Ireland et al. (1996) found that exposure to UV radiation (natural sunlight) increased the toxicity of PAH-contaminated sediments to *C. dubia*. The toxicity was removed when the UV wavelengths did not penetrate the water column to the exposed organisms. Toxicity was also reduced significantly in the presence of UV when the organic fraction of the stormwater was removed. Photo-induced toxicity occurred frequently during low flow conditions and wet-weather runoff and was reduced during turbid conditions.

Johnson et al. (1996) and Herricks et al. (1996a,b) describe a structured tier testing protocol to assess both short-term and long-term wet-weather discharge toxicity that they developed and tested. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures. The use of standard whole effluent toxicity (WET) tests were found to overestimate the potential toxicity of stormwater discharges.

The effects of stormwater on Lincoln Creek, near Milwaukee, WI, were described by Crunkilton et al. (1996). Lincoln Creek drains a heavily urbanized watershed of 19 mi^2 that is about 9 miles long. On-site toxicity testing was conducted with side-stream flow-through aquaria using fathead minnows, plus in-stream biological assessments, along with water and sediment chemical measurements. In the basic tests, Lincoln Creek water was continuously pumped through the test tanks, reflecting the natural changes in water quality during both dry and wet-weather conditions. The continuous flow-through mortality tests indicated no toxicity until after about the 14th day of exposure, with more than 80% mortality after about 25 days, indicating that short-term toxicity tests likely underestimate stormwater toxicity. The biological and physical habitat assessments supported a definitive relationship between degraded stream ecology and urban runoff.

Rainbow (1996) presented a detailed overview of heavy metals in aquatic invertebrates. He concluded that the presence of a metal in an organism cannot tell us directly whether that metal is poisoning the organism. However, if compared to concentrations in a suite of well-researched biomonitors, it may be possible to determine if the accumulated concentrations are atypically high, with a possibility that toxic effects may be present. The user should be cautious, however, when attempting to relate tissue concentrations to effects or with bioconcentration factors. Many metals are essential and/or regulated by organisms and their internal concentrations might bear no relationship to the concentrations in surrounding waters or sediments.

A battery of laboratory and *in situ* bioassay tests are most useful when determining aquatic biota problems (Burton and Stemmer 1988; Burton et al. 1996; Chapter 6). The test series may include microbial activity tests, along with exposures of zooplankton, amphipods, aquatic insects, bivalves, and fish. Indigenous microbial activity responses correlated well with *in situ* biological and chemical profiles. Bascombe et al. (1990) also reported on the use of *in situ* biological tests, using an amphipod exposed for 5 to 6 weeks in urban streams, to examine urban runoff receiving water effects. Ellis et al. (1992) examined bioassay procedures for evaluating urban runoff effects on receiving water biota. They concluded that an acceptable criteria for protecting receiving water organisms should not only provide information on concentration and exposure relationships for *in situ* bioassays, but also consider body burdens, recovery rates, and sediment-related effects.

During the Coyote Creek, San Jose, CA, receiving water study, 41 stations were studied in both urban and non-urban perennial flow stretches of the creek. Short- and long-term sampling techniques were used to evaluate the effects of urban runoff on water quality, sediment properties, fish, macroinvertebrates, attached algae, and rooted aquatic vegetation (Pitt and Bozeman 1982).

Fish Kills and Advisories

Runoff impacts are sometimes difficult for many people to appreciate in urban and agricultural areas. Fish kills are the most obvious indication of water quality problems for many people. However, because receiving water quality is often so poor, the aquatic life in typical urban and agricultural receiving waters is usually limited in abundance and diversity, and quite resistant to poor water quality. Sensitive native organisms have typically been displaced, or killed, long ago, and it usually requires an unusual event to cause a fish fill (Figure 3.1). Ray and White (1979) stated that one of the complicating factors in determining fish kills related to heavy metals is that the fish mortality may lag behind the first toxic exposure by several days and is usually detected many miles downstream from the discharge location. The actual concentrations of the water quality constituents that may have caused the kill could then be diluted beyond detection limits, making probable sources of the toxic materials impossible to determine in many cases.

Heaney et al. (1980) reviewed fish kill information reported to government agencies from 1970 to 1979. They found that less than 3% of the reported 10,000 fish kills was identified as having been caused by urban runoff. This is fewer than 30 fish kills per year nationwide. However, the cause of most of these 10,000 fish kills could not be identified. It is expected that many of these fish kills could have been caused by runoff, or a combination of problems that could have been worsened by runoff. For example, elevated nutrient loading causes eutrophication that may lead to dissolved oxygen deficits and subsequent fish kills. These events are exacerbated by natural stressors such as low flow conditions. More recent surveys have found nearly 30% of fish kills is attributable to runoff (Figure 3.2; EPA 1995).

During the Bellevue, WA, receiving water studies, some fish kills were noted in the unusually clean urban streams (Pitt and Bissonnette 1984). The fish kills were usually associated with inappropriate discharges to the storm drainage system (such as cleaning materials and industrial chemical spills) and not from "typical" urban runoff. However, as noted later, the composition of the fish in the Bellevue urban stream was quite different, as compared to the control stream (Scott et al. 1986).

Figure 3.1 Fish kill in Village Creek, Birmingham, AL, due to Dursban entering storm drainage during warehouse fire.

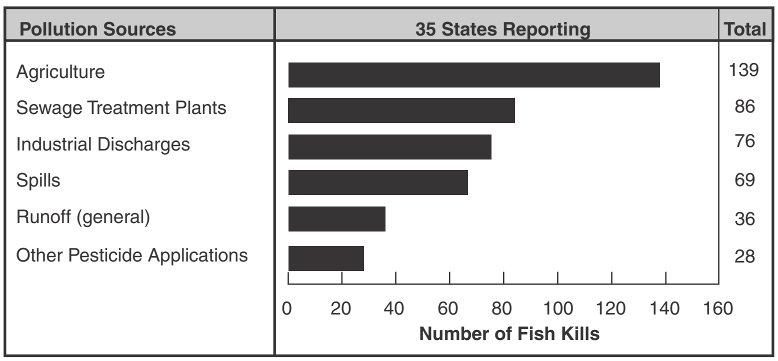

Figure 3.2 Sources associated with fish kills. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

Fish kill data have, therefore, not been a good indicator for identifying stressor categories or types. However, the composition of the fisheries and other aquatic life taxonomic information are sensitive indicators of receiving water problems in streams.

In addition to fish kills, a significant concern is the increasing number of fish advisories being issued by states across the nation (Figure 3.3; EPA 1995). The causes of fish contamination and fish kills vary, but runoff is a primary contributor.

Adverse Aquatic Life Effects Caused by Runoff

Aquatic organisms are sensitive indicators of water quality. There have been many studies that describe aquatic life impairments that may result from exposure to contaminated runoff and/or habitat degradation. The following section summarizes some of these studies, which are typical of urban and agricultural watersheds.

Klein (1979) studied 27 small watersheds having similar characteristics, but having varying land uses, in the Piedmont region of Maryland. During an initial phase of the study, definite relationships were found between water quality and land use. Subsequent study phases examined aquatic life relationships in the watersheds. The principal finding was that stream aquatic life problems were first identified with watersheds having imperviousness areas comprising at least 12% of the watershed. Severe problems were noted after the imperviousness quantities reached 30%.

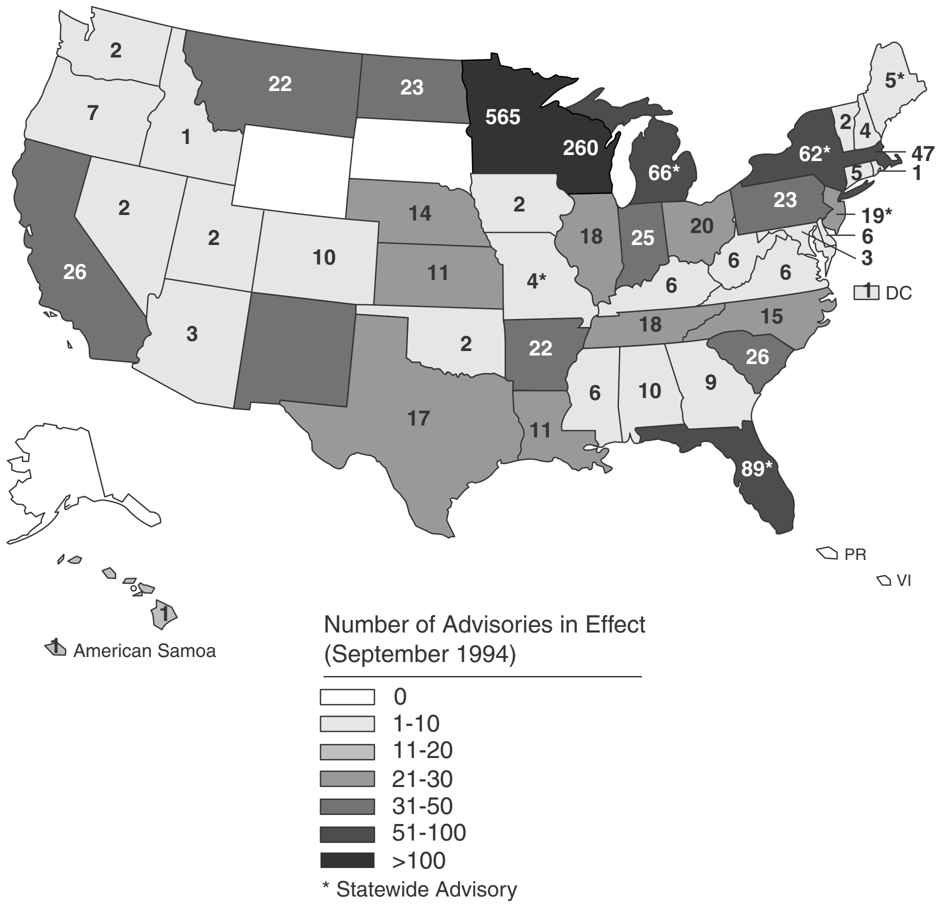

Figure 3.3 U.S. fish consumption advisories. **Note:** States that perform routine fish tissue analysis (such as Great Lake States) will detect more cases of fish contamination and issue more advisories than states with less rigorous fish sampling programs. In many cases, the states with the most fish advisories support the best monitoring programs for measuring toxic contamination in fish, and their water quality is no worse than the water quality in other states. (From U.S. Environmental Protection Agency. *National Water Quality Inventory. 1994 Report to Congress*. Office of Water. EPA 841-R-95-005. Washington, D.C. December 1995.)

Receiving water impact studies were also conducted in North Carolina by Lenat et al. (1979), Lenat and Eagleson (1981), and Lenat et al. (1981). The benthic fauna occurred mainly on rocks. As sedimentation increased, the amount of exposed rocks decreased, with a decreasing density of benthic macroinvertebrates. Data from 1978 and 1979 in five cities showed that urban streams were grossly polluted by a combination of toxicants and sediment. Chemical analyses, without biological analyses, would have underestimated the severity of the problems because the water column quality varied rapidly, while the major problems were associated with sediment quality and effects on macroinvertebrates. Macroinvertebrate diversities were severely reduced in the urban streams, compared to the control streams. The biotic indices indicated “very poor” conditions for all urban streams. Occasionally, high populations of pollutant-tolerant organisms were found in the urban streams, but would abruptly disappear before subsequent sampling efforts. This was probably caused by intermittent discharges of spills or illegal dumping of toxicants. Although the cities studied were located in different geographic areas of North Carolina, the results were remarkably uniform.

A major nonpoint runoff receiving water impact research program was conducted in Georgia (Cook et al. 1983). Several groups of researchers examined streams in major areas of the state.

Benke et al. (1981) studied 21 stream ecosystems near Atlanta having watersheds of 1 to 3 square miles each and land uses ranging from 0 to 98% urbanization. They measured stream water quality but found little relationship between water quality and degree of urbanization. The water quality parameters also did not identify a major degree of pollution. In contrast, there were major correlations between urbanization and the number of species. They had problems applying diversity indices to their study because the individual organisms varied greatly in size (biomass). CTA (1983) also examined receiving water aquatic biota impacts associated with nonpoint sources in Georgia. They studied habitat composition, water quality, macroinvertebrates, periphyton, fish, and toxicant concentrations in the water, sediment, and fish. They found that the impacts of land use were the greatest in the urban basins. Beneficial uses were impaired or denied in all three urban basins studied. Fish were absent in two of the basins and severely restricted in the third. The native macroinvertebrates were replaced with pollution-tolerant organisms. The periphyton in the urban streams were very different from those found in the control streams and were dominated by species known to create taste and odor problems.

Pratt et al. (1981) used basket artificial substrates to compare benthic population trends along urban and nonurban areas of the Green River in Massachusetts. The benthic community became increasingly disrupted as urbanization increased. The problems were not only associated with times of heavy rain, but seemed to be affected at all times. The stress was greatest during summer low flow periods and was probably localized near the stream bed. They concluded that the high degree of correspondence between the known sources of urban runoff and the observed effects on the benthic community was a forceful argument that urban runoff was the causal agent of the disruption observed.

Cedar swamps in the New Jersey Pine Barrens were studied by Ehrenfeld and Schneider (1983). They examined 19 swamps subjected to varying amounts of urbanization. Typical plant species were lost and replaced by weeds and exotic plants in urban runoff-affected swamps. Increased uptakes of phosphorus and lead in the plants were found. It was concluded that the presence of stormwater runoff to the cedar swamps caused marked changes in community structure, vegetation dynamics, and plant tissue element concentrations.

Medeiros and Coler (1982) and Medeiros et al. (1984) used a combination of laboratory and field studies to investigate the effects of urban runoff on fathead minnows. Hatchability, survival, and growth were assessed in the laboratory in flow-through and static bioassay tests. Growth was reduced to one half of the control growth rates at 60% dilutions of urban runoff. The observed effects were believed to be associated with a combination of toxicants.

The benthos in the upper reaches of Coyote Creek (San Jose, CA) consisted primarily of amphipods and a diverse assemblage of aquatic insects (Pitt and Bozeman 1982). Together those groups comprised two thirds of the benthos collected from the non-urban portion of the creek. Clean water forms were abundant and included amphipods (*Hyaella azteca*) and various genera of mayflies, caddisflies, black flies, crane flies, alderflies, and riffle beetles. In contrast, the benthos of the urban reaches of the creek consisted almost exclusively of pollution-tolerant oligochaete worms (tubificids). Tubificids accounted for 97% of the benthos collected from the lower portion of Coyote Creek.

There were significant differences in the numbers and types of benthic organisms found during the Bellevue Urban Runoff Program (Pederson 1981; Perkins 1982; Richey et al. 1981; Richey 1982; Scott et al. 1982). Mayflies, stoneflies, caddisflies, and beetles were rarely observed in urbanized Kelsey Creek, but were quite abundant in rural Bear Creek. These organisms are commonly regarded as sensitive indicators to environmental degradation. As an example of a degraded aquatic habitat, a species of clams (*Unionidae*) was not found in Kelsey Creek, but was found in Bear Creek. These clams are very sensitive to heavy siltation and unstable sediments. Empty clam shells, however, were found buried in the Kelsey Creek sediments indicating their previous presence in the creek and their inability to adjust to the changing conditions. The benthic organism composition in Kelsey Creek varied radically with time and place, while the organisms were much more stable in Bear Creek.

Introduced fishes often cause radical changes in the nature of the fish fauna present in a given water body. In many cases, they become the dominant fishes because they are able to outcompete the native fishes for food or space, or they may possess greater tolerance to environmental stress. In general, introduced species are most abundant in aquatic habitats modified by man, while native fishes tend to persist mostly in undisturbed areas. Such is apparently the case within Coyote Creek, San Jose, CA (Pitt and Bozeman 1982).

Samples from the non-urban portion of the study area were dominated by an assemblage of native fish species such as hitch, three spine stickleback, Sacramento sucker, and prickly sculpin. Rainbow trout, riffle sculpin, and Sacramento squawfish were captured only in the headwater reaches and tributary streams of Coyote Creek. Collectively, native species comprised 89% of the number and 79% of the biomass of the 2379 fishes collected from the upper reaches of the study area. In contrast, native species accounted for only 7% of the number and 31% of the biomass of the 2899 fishes collected from the urban reach of the study area.

Hitch was the most numerous native fish species present. Hitch generally exhibit a preference for quiet water habitat and are characteristic of warm, low elevation lakes, sloughs, sluggish rivers, and ponds. Mosquitofish dominated the collections from the urbanized section of the creek and accounted for over two thirds of the total number of fish collected from the area. This fish is particularly well adapted to withstand extreme environmental conditions, including those imposed by stagnant waters with low dissolved oxygen concentrations and elevated temperatures. The second most abundant fish species in the urbanized reach of Coyote Creek, the fathead minnow, is equally well suited to tolerate extreme environmental conditions. The species can withstand low dissolved oxygen, high temperature, high organic pollution, and high alkalinity. Often thriving in unstable environments such as intermittent streams, the fathead minnow can survive in a wide variety of habitats.

The University of Washington (Pederson 1981; Perkins 1982; Richey et al. 1981; Richey 1982; Scott et al. 1982) conducted a series of studies to contrast the biological and chemical conditions in urban Kelsey Creek with rural Bear Creek. The urban creek was significantly degraded when compared to the rural creek, but still supported a productive but limited and unhealthy salmonid fishery. Many of the fish in the urban creek, however, had respiratory anomalies. The urban creek was not grossly polluted, but flooding from urban developments has increased dramatically in recent years. These increased flows have dramatically changed the urban stream's channel, by causing unstable conditions with increased stream bed movement, and by altering the availability of food for the aquatic organisms. The aquatic organisms are very dependent on the few relatively undisturbed reaches. Dissolved oxygen concentrations in the sediments depressed embryo salmon survival in the urban creek. Various organic and metallic priority pollutants were discharged to the urban creek, but most of them were apparently carried through the creek system by the high storm flows to Lake Washington. The urbanized Kelsey Creek also had higher water temperatures (probably due to reduced shading) than Bear Creek. This probably caused the faster fish growth in Kelsey Creek.

The fish population in Kelsey Creek had adapted to its degrading environment by shifting the species composition from coho salmon to less sensitive cutthroat trout and by making extensive use of less-disturbed refuge areas (Figure 4.22). Studies of damaged gills found that up to three fourths of the fish in Kelsey Creek were affected with respiratory anomalies, while no cutthroat trout and only two of the coho salmon sampled in Bear Creek had damaged gills. Massive fish kills in Kelsey Creek and its tributaries were observed on several occasions during the project due to the dumping of toxic materials down the storm drains.

Urban runoff impact studies were conducted in the Hillsborough River near Tampa Bay, FL, as part of the NURP program (Mote Marine Laboratory 1984). Plants, animals, sediment, and water quality were all studied in the field and supplemented by laboratory bioassay tests. Effects of saltwater intrusion and urban runoff were both measured because of the estuarine environment. During wet weather, freshwater species were found closer to the bay than during dry weather. In coastal areas, these additional natural factors make it even more difficult to identify the

Figure 3.4 Installation of side-stream fish bioassay test facilities at Lincoln Creek, Milwaukee, WI.

Figure 3.5 Lincoln Creek side-stream fish bioassay test facilities nearing completion.

cause-and-effect relationships for aquatic life problems. During another NURP project, Striegl (1985) found that the effects of accumulated pollutants in Lake Ellyn (Glen Ellyn, IL) inhibited desirable benthic invertebrates and fish and increased undesirable phytoplankton blooms. LaRoe (1985) summarized the off-site effects of construction sediment on fish and wildlife. He noted that physical, chemical, and biological processes all affect receiving water aquatic life.

The number of benthic organism taxa in Shabakunk Creek in Mercer County, NJ, declined from 13 in relatively undeveloped areas to 4 below heavily urbanized areas (Garie and McIntosh 1986, 1990). Periphyton samples were also analyzed for heavy metals, with significantly higher metal concentrations found below the heavily urbanized area than above.

The Wisconsin Department of Natural Resources, in conjunction with the USGS and the University of Wisconsin, conducted side-stream fish bioassay tests in Lincoln Creek in Milwaukee (Figures 3.4 and 3.5) (Crunkilton et al. 1996). They identified significant acute toxicity problems associated with intermediate-term (about 10 to 20 day) exposures to adverse toxicant concentrations in urban receiving streams, with no indication of toxicity for shorter exposures. These toxicity effects were substantially (but not completely) reduced through the removal of stormwater particulates using a typical wet detention pond designed to remove most of the particles larger than 5 μm.

Observed Habitat Problems Caused by Runoff

Some of the most serious effects of urban and agricultural runoff are on the aquatic habitat of the receiving waters. These habitat effects are in addition to the pollutant concentration effects. The major effects of sediment on the aquatic habitat include silting of spawning and food production areas and unstable bed conditions (Cordone and Kelley 1961). Other major habitat destruction problems include rapidly changing flows and the absence of refuge areas to protect the biota during these flow changes. Removal of riparian vegetation can increase water temperatures and eliminate a major source of debris, which provides important refuge areas. The major source of these habitat problems is the increased discharge volumes and flow rates associated with stormwater in developing areas that cause significant enlargements and unstable banks of small and moderate sized streams (Figures 3.6 and 3.7). Other habitat problems are caused by attempts to “correct” these problems by construction of lined channels (Figures 3.8 and 3.9) or small drop structures which hinder migration of aquatic life and create areas for the accumulation of contaminated silt (Figure 3.10).

Figure 3.6 Creek blowout after initial significant spring rains in newly developed area. (Courtesy of Wisconsin Department of Natural Resources.)

Figure 3.7 Unstable banks and trash along Five-Mile Creek, Birmingham, AL.

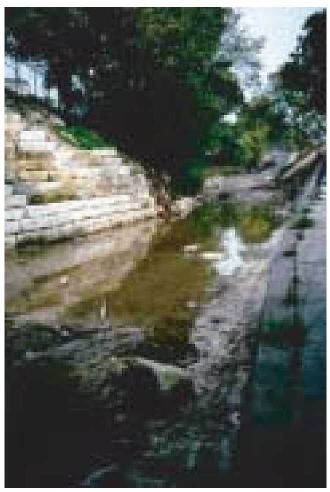

Figure 3.8 Lined embankment along Waller Creek, Austin, TX.

Figure 3.9 Lined channel in Milwaukee, WI.

Schueler (1996) stated that channel geometry stability can be a good indicator of the effectiveness of stormwater control practices. He also found that once a watershed area has more than about 10 to 15% effective impervious cover, noticeable changes in channel morphology occur, along with quantifiable impacts on water quality and biological conditions. Stephenson (1996) studied changes in streamflow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about twofold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration.

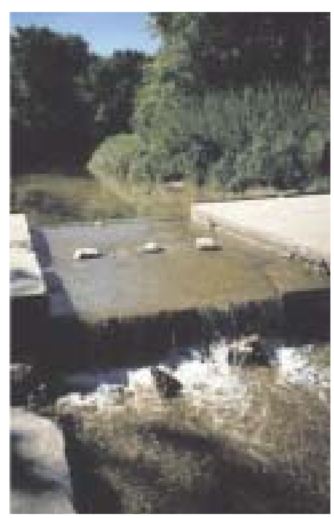

Figure 3.10 Small drop structure obstruction in Lincoln Creek, Milwaukee, WI.

Brookes (1988) has documented many cases in the United States and Great Britain of stream morphological changes associated with urbanization. These changes are mostly responsible for habitat destruction which is usually the most significant detriment to aquatic life. In many cases, water quality improvement would result in very little aquatic life benefit if the physical habitat is grossly modified. The most obvious habitat problems are associated with stream "improvement" projects, ranging from removal of debris, to straightening streams, to channelization projects. Brookes (1988, 1991) presents a number of ways to minimize habitat problems associated with stream channel projects, including stream restoration.

Wolman and Schick (1967) observed deposition of channel bars, erosion of channel banks, obstruction of flows, increased flooding, shifting of channel bottoms, along with concurrent changes in the aquatic life, in Maryland streams affected by urban construction activities. Robinson (1976) studied eight streams in watersheds undergoing urbanization and found that the increased magnitudes and frequencies of flooding, along with the increased sediment yields, had considerable impact on stream morphology (and therefore aquatic life habitat).

The aquatic organism differences found during the Bellevue Urban Runoff Program were probably most associated with the increased peak flows in Kelsey Creek caused by urbanization and the resultant increase in sediment-carrying capacity and channel instability of the creek (Pederson 1981; Perkins 1982; Richey et al. 1981; Richey 1982; Scott et al. 1982). Developed Kelsey Creek had much lower flows than rural Bear Creek during periods between storms. About 30% less water was available in Kelsey Creek during the summers. These low flows may also have significantly affected the aquatic habitat and the ability of the urban creek to flush toxic spills or other dry-weather pollutants from the creek system (Ebbert et al. 1983; Prych and Ebbert undated). Kelsey Creek had extreme hydrologic responses to storm. Flooding substantially increased in Kelsey Creek during the period of urban development; the peak annual discharges have almost doubled in the last 30 years, and the flooding frequency has also increased due to urbanization (Ebbert et al. 1983; Prych and Ebbert undated). These increased flows in urbanized Kelsey Creek resulted in greatly increased sediment transport and channel instability. The Bellevue studies (summarized by Pitt and Bissonnette 1984) indicated very significant interrelationships between the physical, biological, and chemical characteristics of the urbanized Kelsey Creek system. The aquatic life beneficial uses were found to be impaired, and stormwater conveyance was most likely associated with increased flows from the impervious areas in the urban area. Changes in the flow characteristics could radically alter the ability of the stream to carry the polluted sediments into the other receiving waters. If the stream power (directly related to sediment-carrying capacity) of Kelsey Creek were reduced, these toxic materials could be expected to be settled into its sediment, with increased effects on the stream's aquatic life. Reducing peak flows would also reduce the flushing of smaller fish and other aquatic organisms from the system.

Many recent studies on urban stream habitats and restoration efforts have been conducted, especially in the Pacific Northwest. In one example, May et al. (1999) found that maintaining natural land cover offers the best protection for maintaining stream ecological integrity and that best management practices have generally been limited in their ability to preserve appropriate conditions for lowland salmon spawning and rearing streams. They found that Puget Sound watersheds having a 10% impervious cover (likely resulting in marginal in-stream conditions) maintained at least 50% forested cover.

Groundwater Impacts from Stormwater Infiltration

There have been some nationwide studies that have shown virtually every agricultural and urban watershed contains elevated levels of nutrients, pesticides, and other organic chemicals in surface and groundwaters, sediments, and fish tissues (e.g., USGS 1999). Since groundwaters are widely used as a drinking water and irrigation source and recharge many surface water bodies, the implications of chemical contamination are quite serious.

Prior to urbanization, groundwater recharge resulted from infiltration of precipitation through pervious surfaces, including grasslands and woods. This infiltrating water was relatively uncontam-

Figure 3.11 Groundwater recharge basin in Long Island, NY, using stormwater. (Courtesy of New York Department of USGS).

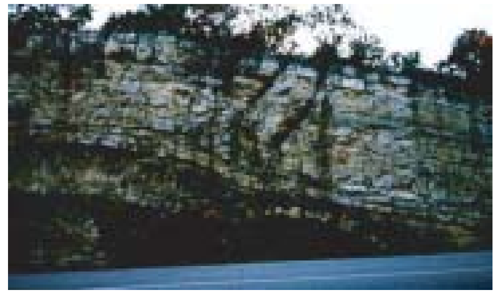

Figure 3.12 Karst geology at an Austin, TX, roadcut showing major channeling opportunities for surface water to enter the Edwards Aquifer.

Figure 3.13 Public education roadside sign in Austin, TX, warning about sensitive recharge zone.

Figure 3.14 Paver blocks for on-site infiltration in Essen, Germany.

inated. Urbanization reduced the permeable soil surface area through which recharge by infiltration could occur. This resulted in much less groundwater recharge and greatly increased surface runoff. In addition, the waters available for recharge generally carried increased quantities of pollutants. With urbanization, new sources of groundwater recharge also occurred, including recharge from domestic septic tanks, percolation basins (Figure 3.11), and industrial waste injection wells, and from agricultural and residential irrigation. Special groundwater contamination problems may occur in areas having Karst geology where surface waters can be easily and quickly directed to the subsurface (Figures 3.12 and 3.13). Of course, there are many less dramatic opportunities for stormwater to enter the groundwater, including areas of porous paver blocks (Figures 3.14 through 3.16), grass swales (Figures 3.17 and 3.18), infiltration trenches (Figure 3.19), biofiltration areas (Figure 3.20), and simply from runoff flowing across grass (Figure 3.21). Many of these infiltration practices are done to reduce surface water impacts associated with stormwater discharges. If the infiltration is conducted through surface soils (such as for grass swales and grass landscaped areas), groundwater contamination problems are significantly reduced. However, if subsurface infiltration is used (especially through the use of injection wells), the risk of groundwater contamination for many stormwater pollutants substantially increases (Pitt et al. 1994, 1996).

Figure 3.15 Paver blocks for emergency and utility vehicle access, Madison, WI (under construction).

Figure 3.16 Paver blocks for occasional access road, Seattle Science Center, WA.

Figure 3.17 Grass swale in residential area, Milwaukee, WI.

Figure 3.18 Grass swale in office park area, Milwaukee, WI.

The Technical University of Denmark (Mikkelsen et al. 1996a,b) has been involved in a series of tests to examine the effects of stormwater infiltration on soil and groundwater quality. It found that heavy metals and PAHs present little groundwater contamination threat if surface infiltration systems are used. However, it expresses concern about pesticides, which are much more mobile. Squillace et al. (1996) along with Zogorski et al. (1996) presented information concerning stormwater and its potential as a source of groundwater MTBE contamination. Mull (1996) stated that traffic areas are the third most important source of groundwater contamination in Germany (after abandoned industrial sites and leaky sewers). The most important contaminants are chlorinated hydrocarbons, sulfate, organic compounds, and nitrates. Heavy metals are generally not an important groundwater contaminant because of their affinity for soils. Trauth and Xanthopoulus (1996) examined the long-term trends in groundwater quality at Karlsruhe, Germany. They found that urban land use is having a long-term influence on the groundwater quality. The concentration of many pollutants has increased by about 30 to 40% over 20 years. Hütter and Remmler (1996)

Figure 3.19 Stormwater infiltration through infiltration trench, office park, Lake Oswego, OR.

Figure 3.20 Biofiltration in parking area (Photo used with permission of Center for Watershed Protection.)

describe a groundwater monitoring plan, including monitoring wells that were established during the construction of an infiltration trench for stormwater disposal, in Dortmund, Germany. The worst problem expected is with zinc if the infiltration water has a pH value of 4.

The following paragraphs (summarized from Pitt et al. 1994, 1996) describe the stormwater pollutants that have the greatest potential of adversely affecting groundwater quality during inadvertent or intentional stormwater infiltration, along with suggestions on how to minimize these potential problems.

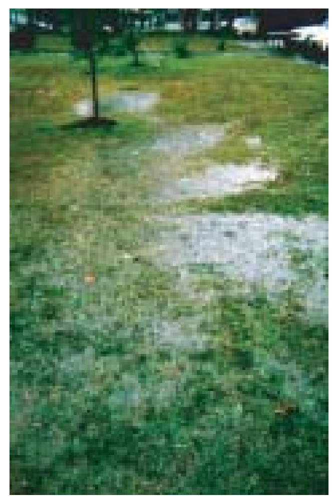

Figure 3.21 Infiltration through grassed areas.

Nutrients

Groundwater contamination with phosphorus has not been as widespread, or as severe, as with nitrogen compounds. Nitrates are one of the most frequently encountered contaminants in groundwater. Whenever nitrogen-containing compounds come into contact with soil, a potential for nitrate leaching into groundwater exists, especially in rapid-infiltration wastewater basins, stormwater infiltration devices, and in agricultural areas. Nitrate has leached from fertilizers and affected groundwaters under various turf grasses in urban areas, including golf courses, parks, and home lawns. Significant leaching of nitrates occurs during the cool, wet seasons. Cool temperatures reduce denitrification and ammonia volatilization, and limit microbial nitrogen immobilization and plant uptake. The use of slow-release fertilizers is recommended in areas having potential groundwater nitrate problems. The slow-release fertilizers include urea formaldehyde (UF), methylene urea, isobutyldiene diurea (IBDU), and sulfur-coated urea. Residual nitrate concentrations are highly variable in soil due to soil texture, mineralization, rainfall and irrigation patterns, organic matter content, crop yield, nitrogen fertilizer/sludge rate, denitrification, and soil compaction. Nitrate is highly soluble (>1 kg/L) and will stay in solution in the percolation water, after leaving the root zone, until it reaches the groundwater.

Nitrate has a low to moderate groundwater contamination potential for both surface percolation and subsurface infiltration/injection practices because of its relatively low concentrations found in most stormwaters. However, if the stormwater nitrate concentration were high, then the groundwater contamination potential would also likely be high.

Pesticides

Pesticide contamination of groundwater can result from agricultural, municipal, and homeowner use of pesticides for pest control and their subsequent collection in stormwater runoff. A wide range of pesticides and their metabolites have been found in watersheds, which include typical urban pesticides in agricultural areas, and vice versa. This cross-contamination of pesticides into areas in which they are not being used is attributed to atmospheric transport. Heavy repetitive use of mobile pesticides on irrigated and sandy soils likely contaminates groundwater. Some insecticides, fungicides, and nematocides must be mobile in order to reach the target pest and, hence, they generally have the highest contamination potential. Pesticide leaching depends on patterns of use, soil texture, total organic carbon content of the soil, pesticide persistence, and depth to the water table.

The greatest pesticide mobility occurs in areas with coarse-grained or sandy soils without a hardpan layer, having low clay and organic matter content and high permeability. Structural voids, which are generally found in the surface layer of finer-textured soils rich in clay, can transmit pesticides rapidly when the voids are filled with water and the adsorbing surfaces of the soil matrix are bypassed. In general, pesticides with low water solubilities, high octanol-water partitioning coefficients, and high carbon partitioning coefficients are less mobile. The slower-moving pesticides have been recommended in areas of groundwater contamination concern. These include the fungicides iprodione and triadimefon, the insecticides isofenphos and chlorpyrifos, and the herbicide glyphosate. The most mobile pesticides include 2,4-D, acenaphthylene, alachlor, atrazine, cyanazine, dacthal, diazinon, dicamba, malathion, and metolachlor.

Pesticides decompose in soil and water, but the total decomposition time can range from days to years. Literature half-lives for pesticides generally apply to surface soils and do not account for the reduced microbial activity found deep in the vadose zone. Pesticides with a 30-day half-life can show considerable leaching. An order-of-magnitude difference in half-life results in a five- to tenfold difference in percolation loss. Organophosphate pesticides are less persistent than organochlorine pesticides, but they also are not strongly adsorbed by the sediment and are likely to leach into the vadose zone and the groundwater. Perhaps a greater concern that has recently emerged is the widespread prevalence of toxic pesticide metabolites (breakdown products) that are not routinely analyzed. The ecological and human health significance of this is not known at present, but will be a future topic of investigation.

Lindane and chlordane have moderate groundwater contamination potentials for surface percolation practices (with no pretreatment) and for subsurface injection (with minimal pretreatment). The groundwater contamination potentials for both of these compounds would likely be substantially reduced with adequate sedimentation pretreatment. Pesticides have mostly been found in urban runoff from residential areas, especially in dry-weather flows associated with landscaping irrigation runoff.

Other Organics

The most commonly occurring organic compounds that have been found in urban groundwaters include phthalate esters (especially bis(2-ethylhexyl)phthalate) and phenolic compounds. Other organics more rarely found, possibly due to losses during sample collection, have included the volatiles: benzene, chloroform, methylene chloride, trichloroethylene, tetrachloroethylene, toluene, and xylene. PAHs (especially benzo(a)anthracene, chrysene, anthracene, and benzo(b)fluoroanthenene) have also been found in groundwaters near industrial sites.

Groundwater contamination from organics, as from other pollutants, occurs more readily in areas with sandy soils and where the water table is near the land surface. Removal of organics from the soil and recharge water can occur by one of three methods: volatilization, sorption, and degradation. Volatilization can significantly reduce the concentrations of the most volatile compounds in groundwater, but the rate of gas transfer from the soil to the air is usually limited by the presence of soil water. Hydrophobic sorption onto soil organic matter limits the mobility of less soluble base/neutral and acid extractable compounds through organic soils and the vadose zone. Sorption is not always a permanent removal mechanism, however. Organic resolubilization can occur during wet periods following dry periods. Many organics can be at least partially degraded by microorganisms, but others cannot. Temperature, pH, moisture content, ion-exchange capacity of soil, and air availability may limit the microbial degradation potential for even the most degradable organic.

1,3-Dichlorobenzene may have a high groundwater contamination potential for subsurface infiltration/injection (with minimal pretreatment). However, it would likely have a lower groundwater contamination potential for most surface percolation practices because of its relatively strong sorption to vadose zone soils. Both pyrene and fluoranthene would also likely have high groundwater contamination potentials for subsurface infiltration/injection practices, but lower contamination potentials for surface percolation practices because of their more limited mobility through the unsaturated zone (vadose zone). Others (including benzo(a)anthracene, bis(2-ethylhexyl) phthalate, pentachlorophenol, and phenanthrene) may also have moderate groundwater contamination potentials if surface percolation with no pretreatment or subsurface injection/infiltration is used. These compounds would have low groundwater contamination potentials if surface infiltration was used with sedimentation pretreatment. Volatile organic compounds (VOCs) may also have high groundwater contamination potentials if present in the stormwater (likely for some industrial and commercial facilities and vehicle service establishments). The other organics, especially the volatiles, are mostly found in industrial areas. The phthalates are found in all areas. The PAHs are also found in runoff from all areas, but they are in higher concentrations and occur more frequently in industrial areas.

Pathogenic Microorganisms

Viruses have been detected in groundwater where stormwater recharge basins are located short distances above the aquifer. Enteric viruses are more resistant to environmental factors than enteric bacteria and they exhibit longer survival times in natural waters. They can occur in potable and marine waters in the absence of fecal coliforms. Enteroviruses are also more resistant to commonly used disinfectants than are indicator bacteria, and can occur in groundwater in the absence of indicator bacteria.

The factors that affect the survival of enteric bacteria and viruses in the soil include pH, antagonism from soil microflora, moisture content, temperature, sunlight, and organic matter. The two most important attributes of viruses that permit their long-term survival in the environment are their structure and very small size. These characteristics permit virus occlusion and protection within colloid-size particles. Viral adsorption is promoted by increasing cation concentration, decreasing pH, and decreasing soluble organics. Since the movement of viruses through soil to groundwater occurs in the liquid phase and involves water movement and associated suspended virus particles, the distribution of viruses between the adsorbed and liquid phases determines the viral mass available for movement. Once the virus reaches the groundwater, it can travel laterally through the aquifer until it is either adsorbed or inactivated.

The major bacterial removal mechanisms in soil are straining at the soil surface and at intergrain contacts, sedimentation, sorption by soil particles, and inactivation. Because they are larger than viruses, most bacteria are retained near the soil surface due to this straining effect. In general, enteric bacteria survive in soil for 2 to 3 months, although survival times up to 5 years have been documented.

Enteroviruses likely have a high groundwater contamination potential for all percolation practices and subsurface infiltration/injection practices, depending on their presence in stormwater (likely, if contaminated with sanitary sewage). Other pathogens, including *Shigella*, *Pseudomonas aeruginosa*, and various protozoa, would also have high groundwater contamination potentials if subsurface infiltration/injection practices are used without disinfection. If disinfection (especially by chlorine or ozone) is used, then disinfection by-products (such as trihalomethanes or ozonated bromides) would have high groundwater contamination potentials. Pathogens are most likely associated with sanitary sewage contamination of storm drainage systems, but several bacterial pathogens are commonly found in surface runoff in residential areas.

Heavy Metals and Other Inorganic Compounds

The heavy metals and other inorganic compounds in stormwater of most environmental concern, from a groundwater pollution standpoint, are chromium, copper, lead, nickel, and zinc. However, the majority of metals, with the consistent exception of zinc, are mostly found associated with the particulate solids in stormwaters and are thus relatively easily removed through sedimentation practices. Filterable forms of the metals may also be removed by either sediment adsorption or organically complexing with other particulates.

In general, studies of recharge basins receiving large metal loads found that most of the heavy metals are removed either in the basin sediment or in the vadose zone. Dissolved metal ions are removed from stormwater during infiltration mostly by adsorption onto the near-surface particles in the vadose zone, while the particulate metals are filtered out near the soil surface. Studies at recharge basins found that lead, zinc, cadmium, and copper accumulated at the soil surface with little downward movement over many years. However, nickel, chromium, and zinc concentrations have exceeded regulatory limits in the soils below a recharge area at a commercial site. Elevated groundwater heavy metal concentrations of aluminum, cadmium, copper, chromium, lead, and zinc have been found below stormwater infiltration devices where the groundwater pH has been acidic. Allowing percolation ponds to go dry between storms can be counterproductive to the removal of lead from the water during recharge. Apparently, the adsorption bonds between the sediment and the metals can be weakened during the drying period.

Similarities in water quality between runoff water and groundwater have shown that there is significant downward movement of copper and iron in sandy and loamy soils. However, arsenic, nickel, and lead did not significantly move downward through the soil to the groundwater. The exception to this was some downward movement of lead with the percolation water in sandy soils beneath stormwater recharge basins. Zinc, which is more soluble than iron, has been found in higher concentrations in groundwater than has iron. The order of attenuation in the vadose zone from infiltrating stormwater is zinc (most mobile) > lead > cadmium > manganese > copper > iron > chromium > nickel > aluminum (least mobile).

Nickel and zinc would likely have high groundwater contamination potentials if subsurface infiltration/injection were used. Chromium and lead would have moderate groundwater contamination potentials for subsurface infiltration/injection practices. All metals would likely have low groundwater contamination potentials if surface infiltration were used with sedimentation pretreatment.

Salts

Salt applications for winter traffic safety is a common practice in many northern areas, and the sodium and chloride, which are collected in the snowmelt, travel down through the vadose zone to the groundwater with little attenuation. Soil is not very effective at removing salts. Salts that are still in the percolation water after it travels through the vadose zone will contaminate the groundwater. Infiltration of stormwater has led to increases in sodium and chloride groundwater concentrations above background concentrations. Fertilizer and pesticide salts also accumulate in urban areas and can leach through the soil to the groundwater.

Studies of depth of pollutant penetration in soil have shown that sulfate and potassium concentrations decrease with depth, while sodium, calcium, bicarbonate, and chloride concentrations increase with depth. Once contamination with salts begins, the movement of salts into the groundwater can be rapid. The salt concentration may not decrease until the source of the salts is removed.

Chloride would likely have a high groundwater contamination potential in northern areas where road salts are used for traffic safety, irrespective of the pretreatment, infiltration, or percolation practice used. Salts are at their greatest concentrations in snowmelt and in early spring runoff in northern areas.

STRESSOR CATEGORIES AND THEIR EFFECTS

There are several ways in which stormwater stressors may be grouped. Overlap between these categories will occur since the ecosystem is comprised of interrelated, interactive components. Attempts at studying single stressors or single categories represents a "reductionist" approach as opposed to a more realistic "holistic" ecosystem approach (Chapman et al. 1992). However, for one to understand the whole system and its response to stormwater stressors, there must first be a basic understanding of single component effects and patterns (see also Chapters 3 through 6). The adverse effect of stormwater runoff has been mainly documented indirectly in NPS effect studies in urban and agricultural watersheds. The aquatic ecosystems in these environments typically show a loss of sensitive species, loss of species numbers (diversity and richness), and increases in numbers of pollution-tolerant organisms (e.g., Schueler 1987; EPA 1987a; Pitt and Bozeman 1982; Pitt 1995). These trends are observed at all levels of biological organization including fish, insects, zooplankton, phytoplankton, benthic invertebrates, protozoa, bacteria, and macrophytes. These alterations tend to change an aquatic ecosystem from a stable system to an unstable one, and from a complex system to an overly simplistic one. As disturbances (e.g., toxic stormwater discharges) increase in frequency and severity, the recovery phase will increase and the ability to cope with a disturbance will decrease. The following categories are but a generalized summary of commonly observed characteristics and effects in previous stormwater and ecotoxicological studies.

Stream Flow Effects and Associated Habitat Modifications

Some of the most serious effects of urban and agricultural runoff are on the aquatic habitat of the receiving waters. A major threat to habitat comes from the rapidly changing flows and the absence of refuge areas to protect the biota during these flow changes. The natural changes in stream hydrology will change naturally at a slow, relatively nondetectable rate in most areas of the United States where stream banks are stabilized by riparian vegetation. In other areas, however, natural erosion and bank slumping will occur in response to high flow events. This "natural" contribution to stream solids is accelerated by hydromodifications, such as increases in stream power due to upstream channelization, installation of impervious drainage networks, increased impervious areas in the watershed (roof tops, roadways, parking areas), and removal of trees and vegetation. All of these increase the runoff volume and stream power, and decrease the time period for stream peak discharge.

In moderately developed watersheds, peak discharges are two to five times those of predevelopment levels (Leopold 1968; Anderson 1970). These storm events may have 50% greater volume, which may result in flooding. The quicker runoff periods reduce infiltration; thus, interflows and baseflows into the stream from groundwater during drought periods are reduced, as are groundwater levels. As stream power increases, channel morphology will change with an initial widening of the channel to as much as two to four times its original size (Robinson 1976; Hammer 1972). Floodplains increase in size, stream banks are undercut, and riparian vegetation lost. The increased sediment loading from erosion moves through the watershed as bedload, covering sand, gravel, and cobble substrates.

The aquatic organism differences found during the Bellevue Urban Runoff Program were probably most associated with the increased peak flows in Kelsey Creek caused by urbanization and the resultant increase in sediment-carrying capacity and channel instability of the creek (Pederson 1981; Perkins 1982; Richey et al. 1981; Richey 1982; Scott et al. 1982). Kelsey Creek had much lower flows than Bear Creek during periods between storms. About 30% less water was available in Kelsey Creek during the summers. These low flows may also have significantly affected the aquatic habitat and the ability of the urban creek to flush toxic spills or other dry-weather pollutants from the creek system (Ebbert et al. 1983; Prych and Ebbert undated). Kelsey Creek had extreme hydrologic responses to storms. Flooding substantially increased in Kelsey Creek during the period of urban development; the peak annual discharges have almost doubled in the last 30 years, and the flooding frequency has also increased due to urbanization (Ebbert et al. 1983; Prych and Ebbert undated). These increased flows in urbanized Kelsey Creek resulted in greatly increased sediment transport and channel instability.

The Bellevue studies (Pitt and Bissonnette 1984) indicated very significant interrelationships among the physical, biological, and chemical characteristics of the urbanized Kelsey Creek system. The aquatic life beneficial uses were found to be impaired, and stormwater conveyance was most likely associated with increased flows from the impervious areas in the urban area. Changes in the flow characteristics could radically alter the ability of the stream to carry the polluted sediments into the other receiving waters.

Stephenson (1996) studied changes in stream flow volumes in South Africa during urbanization. He found increased stormwater runoff, decreases in the groundwater table, and dramatically decreased times of concentration. The peak flow rates increased by about twofold, about half caused by increased pavement (in an area having only about 5% effective impervious cover), with the remainder caused by decreased times of concentration.

Bhaduri et al. (1997) quantified the changes in stream flow and decreases in groundwater recharge associated with urbanization. They point out that the most widely addressed hydrologic effect of urbanization is the peak discharge increases that cause local flooding. However, the increase in surface runoff volume also represents a net loss in groundwater recharge. They point out that urbanization is linked to increased variability in volume of water available for wetlands and small streams, causing "flashy" or "flood-and-drought" conditions. In northern Ohio, urbanization at a study area was found to have caused a 195% increase in the annual volume of runoff, while the expected increase in the peak flow for the local 100-year event was 26% for the same site. Although any increase in severe flooding is problematic and cause for concern, the much larger increase in annual runoff volume, and associated decrease in groundwater recharge, likely has a much greater effect on in-stream biological conditions.

A number of presentations concerning aquatic habitat effects from urbanization were made at the *Effects of Watershed Development and Management on Aquatic Ecosystems* conference held in Snowbird, UT, in August of 1996, and sponsored by the Engineering Foundation and the ASCE. MacRae (1997) presented a review of the development of the common zero runoff increase (ZRI) discharge criterion, referring to peak discharges before and after development. This criterion is commonly met using detention ponds for the 2-year storm. MacRae shows how this criterion has not effectively protected the receiving water habitat. He found that stream bed and bank erosion is controlled by the frequency and duration of the mid-depth flows (generally occurring more often than once a year), not the bank-full condition (approximated by the 2-year event). During monitoring

Table 3.1 Hours of Exceedance of Developed Conditions with Zero Runoff Increase (ZRI) Controls Compared to Predevelopment Conditions

Recurrence Interval (yrs)	Existing Flow Rate (m^3/s)	Exceedance for Predevelopment Conditions (hrs per 5 yrs)	Exceedance for Existing Development Conditions, with ZRI Controls (hrs per 5 yrs)	Exceedance for Ultimate Development Conditions, with ZRI Controls (hrs per 5 yrs)
1.01 (critical mid-bank-full conditions)	1.24	90	380	900
1.5 (bank-full conditions)	2.1	30	34	120

near Toronto, he found that the duration of the geomorphically significant predevelopment mid-bank-full flows increased by a factor of 4.2 times, after 34% of the basin had been urbanized, compared to flow conditions before development. The channel had responded by increasing in cross-sectional area by as much as three times in some areas, and was still expanding. Table 3.1 shows the modeled durations of critical discharges for predevelopment conditions, compared to current and ultimate levels of development with "zero runoff increase" controls in place. At full development and even with full ZRI compliance in this watershed, the hours exceeding the critical mid-bank-full conditions will increase by a factor of 10, with significant effects on channel stability and the physical habitat.

MacRae (1997) also reported other studies that found channel cross-sectional areas began to enlarge after about 20 to 25% of the watershed was developed, corresponding to about a 5% impervious cover in the watershed. When the watersheds are completely developed, the channel enlargements were about five to seven times the original cross-sectional areas. Changes from stable stream bed conditions to unstable conditions appear to occur with basin imperviousness of about 10%, similar to the value reported for serious biological degradation. He also summarized a study conducted in British Columbia that examined 30 stream reaches in natural areas, in urbanized areas having peak flow attenuation ponds, and in urbanized areas not having any stormwater controls. The channel widths in the uncontrolled urban streams were about 1.7 times the widths of the natural streams. The streams having the ponds also showed widening, but at a reduced amount compared to the uncontrolled urban streams. He concluded that an effective criterion to protect stream stability (a major component of habitat protection) must address mid-bank-full events, especially by requiring similar durations and frequencies of stream power (the product of shear stress and flow velocity, not just flow velocity alone) at these depths, compared to satisfactory reference conditions.

Urbanization radically affects many natural stream characteristics. Pitt and Bissonnette (1984) reported that the coho and cutthroat were affected by the increased nutrients and elevated temperatures of the urbanized streams in Bellevue, as studied by the University of Washington as part of the EPA NURP project (EPA 1983). These conditions were probably responsible for accelerated growth of the fry, which were observed to migrate to Puget Sound and the Pacific Ocean sooner than their counterparts in the control forested watershed that was also studied. However, the degradation of sediments, mainly the decreased particle sizes, adversely affected their spawning areas in streams that had become urbanized. Sovern and Washington (1997) reported that, in Western Washington, frequent high flow rates can be 10 to 100 times the predevelopment flows in urbanized areas, but that the low flows in the urban streams are commonly lower than the predevelopment low flows. They have concluded that the effects of urbanization on western Washington streams are dramatic, in most cases permanently changing the stream hydrologic balance, by increasing the annual water volume in the stream, increasing the volume and rate of storm flows, decreasing the low flows during dry periods, and increasing the sediment and pollutant discharges from the watershed. With urbanization, the streams increase in cross-sectional area to accommodate these increased flows, and headwater downcutting occurs to decrease the channel gradient. The gradients of stable urban streams are often only about 1 to 2%, compared to 2 to 10% gradients in natural areas. These changes in width and the downcutting result in very different and changing stream conditions. For example, the common pool/drop habitats are generally replaced by pool/riffle habitats, and the stream bed material is comprised of much finer material. Along urban streams, fewer than 50 aquatic plant and animal species are usually found. Researchers have concluded that once urbanization begins, the effects on stream shape are not completely reversible. Developing and maintaining quality aquatic life habitat, however, is possible under urban conditions, but it requires human intervention and it will not be the same as for forested watersheds.

Increased flows due to urban and agricultural modification obviously cause aquatic life impacts due to destroyed habitat (unstable channel linings, scour of sediments, enlarging stream cross sections, changes in stream gradient, collapsing of riparian stands of mature vegetation, siltation, embeddedness, etc.) plus physical flushing of aquatic life from refuge areas downstream. The increases in peak flows, annual runoff amounts, and associated decreases in groundwater recharge obviously cause decreased dry-weather flows in receiving streams. Many small and moderate-sized streams become intermittent after urbanization, causing extreme aquatic life impacts. Even with less severe decreased flows, aquatic life impacts can be significant. Lower flows are associated with increased temperatures, increased pollutant concentrations (due to decreased mixing and transport), and decreased mobility and forage opportunities.

Safety Concerns with Stormwater

There are many aspects of safety associated with urban and agricultural waters, including:

- Exposure to pathogens and toxicants
- Flows (rapidly changing and common high flows)
- Steep banks/cut banks/muddy/slippery banks
- Mucky sediments
- Debris (sharps and strainers)
- Habitat for nuisance organisms (e.g., mosquitoes, rats, snakes)

Most urban receiving waters having direct storm drainage outfalls are quite small and have no formally designated beneficial uses. Larger receiving waters typically have basic uses established, but few urban receiving waters have water contact recreation as a designated beneficial use. Unfortunately, these small waters typically attract local children who may be exposed to some of the hazards associated with stormwater, as noted above. Conditions associated with pathogens and toxicants are likely a serious problem, but the other hazards listed are also very serious. Obviously, drowning should be a concern to all and is often a topic of heated discussion at public meetings where wet detention ponds for stormwater treatment are proposed. However, drowning hazards may be more common in typical urban streams than in well-designed wet detention ponds. These hazards are related to rapidly changing water flows, high flow rates, steep and muddy stream banks, and mucky stream deposits. These hazards are all increased with stormwater discharges and are typically much worse than in predevelopment times when the streams were much more stable. This can be especially critical in newly developing areas where the local streams are thought to be relatively safe from prior experience, but rapidly degrade with increased development and associated stormwater discharges. Other potentially serious hazards are related to debris thrown into streams or trash dumped along stream banks. In unstable urban streams, banks are often continuously cut away, with debris (bankside trees, small buildings, trash piles, and even automobiles) falling into the waterway.

Many people also see untidy urban stream corridors as habitat for snakes and other undesirable creatures and like to clearcut the riparian vegetation and plant grass to the water's edge. Others see creeks as convenient dumping grounds and throw all manner of junk (yard wastes, old appliances, etc.) over their back fences or off bridges into stream corridors. Both of these approaches greatly hinder the use of streams. In contrast, residents of Bellevue, WA, have long accepted the value of small urban streams as habitat for fish. As an example, they have placed large amounts of gravel into streams to provide suitable spawning habitat. In other Northwest area streams, large woody debris is carefully placed into urban streams (using large street-side cranes, and sometimes even helicopters) to improve the aquatic habitat. In these areas, local residents are paying a great deal of money to improve the habitat along the streams and are obviously much more careful about creating hazards associated with trash and other inappropriate debris or discharges.

Drowning Hazards

Marcy and Flack (1981) state that drownings in general most often occur because of slips and falls into water, unexpected depths, cold water temperatures, and fast currents. Four methods to minimize these problems include eliminating or minimizing the hazard, keeping people away, making the onset of the hazard gradual, and providing escape routes.

Jones and Jones (1982) consider safety and landscaping together because landscaping should be used as an effective safety element. They feel that appropriate slope grading and landscaping near the water's edge can provide a more desirable approach than widespread fencing around wet detention ponds. Fences are expensive to install and maintain and usually produce unsightly pond edges. They collect trash and litter, challenge some individuals who like to defy barriers, and impede emergency access if needed. Marcy and Flack (1981) state that limited fencing may be appropriate in special areas. When the side slopes of a wet detention pond cannot be made gradual (such as when against a railroad right-of-way or close to a roadway), steep sides with submerged retaining walls may be needed. A chain-link fence located directly on the top of the retaining wall very close to the water's edge may be needed (to prevent human occupancy of the narrow ledge on the water side of the fence). Another area where fencing may be needed is at the inlet or outlet structures of wet detention ponds. However, fencing usually gives a false sense of security, because most can be easily crossed (Eccher 1991).

Common recommendations to maximize safety near wet detention ponds include suggestions that the pond side slopes be gradual near the water's edge, with a submerged ledge close to shore. Aquatic plants on the ledge would decrease the chance of continued movement to deeper water, and thick vegetation on shore near the water's edge would discourage access to the water and decrease the possibility of falling accidentally. Pathways should not be located close to the water's edge, or turn abruptly near the water. Marcy and Flack (1981) also encourage the placement of escape routes in the water whenever possible. These could be floats on cables, ladders, hand-holds, safety nets, or ramps. They should not be placed to encourage entering the water.

The use of inlet and outlet trash racks and antivortex baffles is also needed to prevent access to locations with dangerous water velocities. Several types are recommended by the NRCS (SCS 1982). Racks need to have openings smaller than about 6 in, to prevent people from passing through them, and they need to be placed where water velocities are less than 3 ft/s, to allow people to escape (Marcy and Flack 1981). Besides maintaining safe conditions, racks also help keep trash from interfering with the operation of the outlet structure.

Eccher (1991) lists the following pond attributes to ensure maximum safety, while having good ecological control:

1. There should be no major abrupt changes in water depth in areas of uncontrolled access.
2. Slopes should be controlled to ensure good footing.

3. All slope areas should be designed and constructed to prevent or restrict weed and insect growth (generally requiring some form of hardened surface on the slopes).
4. Shoreline erosion needs to be controlled.

Obviously, many of these suggestions to improve safety near wet detention ponds may also be applicable to urban stream corridors. Of course, streams can periodically have high water velocities, and steep banks may be natural. However, landscaping and trail placement along urban stream corridors can be carefully done to minimize exposure to the hazardous areas.

Aesthetics, Litter/Floatables, and Other Debris Associated with Stormwater

One of the major problems with the aesthetic degradation of receiving waters in urban areas is a general lack of respect for the local water bodies. In areas where stormwater is considered a beneficial component of the urban water system, these problems are not as severe, and inhabitants and visitors enjoy the local waterscape. The following list indicates the types of aesthetic problems that are common for neglected waters:

- Low flows
- Mucky sediments
- Trash from illegal dumping
- Floatables from discharges of litter
- Unnatural riparian areas
- Unnatural channel modifications
- Odiferous water and sediment
- Rotting vegetation and dead fish
- Objectionable sanitary wastes from CSOs and SSOs

The above list indicates the most obvious aesthetic problems in receiving waters. Many of these problems are directly associated with poor water quality (such as degraded sediments, eutrophication, and fish kills). Other direct problems associated with runoff include massive modifications of the hydrologic cycle with more severe and longer durations of low flow periods due to reduced infiltration of rainwater. Many of the other problems on the above list are related to indirect activities of the inhabitants of the watershed, namely, illegal dumping of trash into streams, littering in the drainage area, and improper modifications. In many areas, separate sewer overflows (SSOs) and combined sewer overflows (CSOs) also contribute unsightly and hazardous debris to urban receiving waters.

Floatable Litter Associated with Wet-Weather Flows

As previously indicated, aesthetics is one of the most important beneficial uses recognized for urban waterways. Floatable litter significantly degrades the aesthetic enjoyment of receiving waters. The control of floatables has therefore long been a goal of most communities.

In coastal areas, land-based sources of beach debris and floatable material have generally been found to originate from wet-weather discharges from the land, and not from marine sources (such as shipping). Of course, in areas where solid wastes (garbage or sewage sludge, for example) have been (or are still being) dumped in the sea, these sources may also be significant beach litter sources. In CSO areas, items of sanitary origin are found in the receiving waters and along the beaches, but stormwater discharges are responsible for most of the bulk litter material, including much of the hazardous materials. In inland areas, marine contributions are obviously not an issue. Therefore, with such direct linkages to the drainage areas, much of the floatable material control efforts have focused on watershed sources and controls (including being part of the “nine minimum” controls

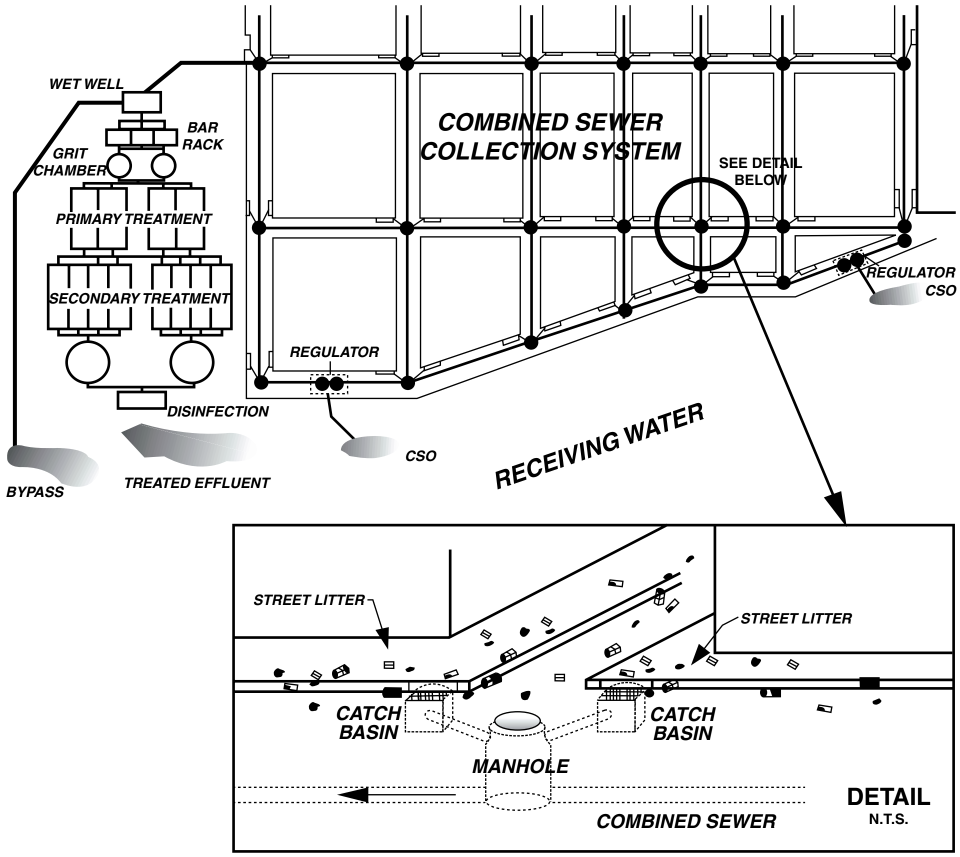

Figure 3.22 Schematic of transport of street and sidewalk litter into receiving waters. (From HydroQual, Inc. *Floatables Pilot Program Final Report: Evaluation of Non-Structural Methods to Control Combined and Storm Sewer Floatable Materials.* City-Wide Floatables Study, Contract II. Prepared for New York City, Department of Environmental Protection, Bureau of Environmental Engineering, Division of Water Quality Improvement. NYDP2000. December 1995.)

for CSOs required by the EPA). Figure 3.22 shows a schematic of how street and sidewalk litter enter the receiving waters (HydroQual 1995).

An example of an investigation of beach litter sources was conducted by Williams and Simmons (1997) along the Bristol Channel in the U.K. They concluded that most of the litter accumulating on the beaches originated from river discharges, and not from litter being deposited directly on the beaches by visitors or from shipping or other oceanic sources. The sources of the litter into the major rivers were the many combined sewer overflows in the area. About 3000 CSOs exist in Wales, and 86 of the 126 CSOs discharging into the study area receive no treatment. They summarized previous studies that have concluded that about half of Britain's coastline is contaminated, with an average of 22 plastic bottles, 17 cans, and 20 sanitary items occurring per km of coast. In some areas, the beach litter can exceed 100 items per category per kilometer. Their survey found that low energy (relatively flat) sandy beaches collected the most debris. Winter litter loadings were generally higher than during the summer, further indicating that storm-related sources were more important than visitor-related sources. They concluded that the linear strip development in South Wales' valleys had led to rivers being used as open sewers and as general dumping grounds.

One of the largest and most comprehensive beach litter and floatable control investigations and control efforts in the United States has been conducted by New York City. At the beginning of their description of this floatable control program, Grey and Olivieri (1998) stated that "one of the major

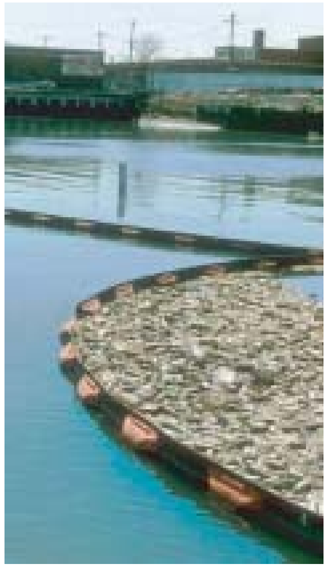

Figure 3.23 Trash boom, New York City.

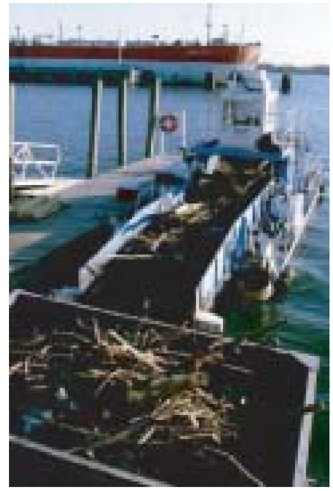

Figure 3.24 New York booms and skimmers for the control of floatable discharges.

issues of urban wet-weather pollution is the control of floatable pollution." The comprehensive New York City program included investigations of the sources of the litter contributing to the floatable discharges (mostly street and sidewalk litter) and the effectiveness of many floatable control practices (including public education, enhanced street cleaning, catchbasin hoods, floatable capture nets, and booming and skimmer boats) (Figures 3.23 through 3.26).

New York City used in-line net boxes installed below catchbasin inlets to capture the discharge of floatables for identification and quantification. Much of the work was directed at the capture efficiency of the floatable material in catchbasins. It was found that it was critical that hoods (covers over the catchbasin outlets that extended below the standing water) be used in the catchbasins to help retain the captured material. The hoods increased the capture of the floatables by 70 to 85%. Unhooded catchbasins were found to discharge about 11 g/100 ft of curb length per day, while

Figure 3.25 TrashTrap™ at Fresh Creek, Brooklyn, NY.

Figure 3.26 New York City's use of end-of-pipe TrashTrap systems.

Table 3.2 Floatable Litter Characteristics Found on New York City Streets

	No. of Items (%)	Weight of Items (%)	Density of Items (lb/ft^3)
Plastic	57.2	44.3	2.8
Metal	18.9	12.0	3.8
Paper (coated/waxed)	5.9	4.0	2.0
Wood	5.9	5.3	7.7
Polystyrene	5.4	1.3	0.7
Cloth/fabric	2.5	12.5	8.3
Sensitive items	1.7	0.4	na
Rubber	1.1	1.1	10.5
Misc.	1.0	3.6	9.8
Glass	0.4	15.6	13.8

From HydroQual, Inc. *Floatables Pilot Program Final Report: Evaluation of Non-Structural Methods to Control Combined and Storm Sewer Floatable Materials.* City-Wide Floatables Study, Contract II. Prepared for New York City, Department of Environmental Protection, Bureau of Environmental Engineering, Division of Water Quality Improvement. NYDP2000. December 1995.

hooded catchbasins reduced this discharge to about 3.3 g/100 ft of curb length per day. It was also found that the hoods greatly extended the period of time between cleanings and the depth of accumulated litter that could be captured in the catchbasins without degraded capture performance.

There are about 130,000 stormwater inlet structures in New York City's 190,000 acres served by combined and separate sewers, or about 1.5 acres served by each inlet. They are surveying all of these inlet structures, replacing damaged or missing hoods, and accurately measuring their dimensions and indicating their exact locations for a citywide GIS system. Catchbasin cleaning costs are about \$170 per inlet, while the inspection and mapping costs are about \$45 per inlet. Replacement hoods cost about \$45 per inlet.

Litter surveys conducted by the New York City Department of Sanitation (DOS) in 1984 and 1986 found that 70% of the street litter items consisted of food and beverage wrappers and containers (60%) and the paper and plastic bags (10%) used to carry these items. The early studies also found that litter levels on the streets and sidewalks were about 20 to 25% higher in the afternoon than in the morning. The DOS conducted similar surveys in 1993 at 90 blockfaces throughout the city (HydroQual 1995). Each litter monitoring site was monitored several times simultaneously when the surveys were conducted with the floatable litter separated into 13 basic categories. They found that twice as much floatable litter was located on the sidewalks compared to the streets (especially glass) and that land use had little effect on the litter loadings (except in the special business districts where enhanced street cleaning/litter control was utilized, resulting in cleaner conditions). Their baseline monitoring program determined that an average of 2.3 floatable litter items were discharged through the catchbasin inlets per day per 100 ft of curb. This amount was equivalent to about 6.2 in^2 and 0.013 lb (8.5 g) of material. The total litter load discharged was about twice this floatable amount. Table 3.2 summarizes the characteristics of the floatable litter found on the streets.

Solids (Suspended, Bedded, and Dissolved)

The detrimental effects of elevated suspended and dissolved solids and increases in siltation and fine-grained bedded sediments have been well documented (EPA 1987b). The sources of these solids are primarily from dry deposition, roadways, construction, and channel alteration and have significant effects on receiving-system habitats. Solids concentrations are directly related to watershed use characteristics and watershed hydrology.

In the United States, 64% of the land is dominated by agriculture and silviculture from which the major pollutant is sediment (approximately 1.8 billion metric tons per year) (EPA 1977). The suspended sediments transport toxicants, nutrients, and lower the aesthetic value of the waterways

Table 3.3 Classification of Suspended and Dissolved Solids and Their Probable Major Impacts on Freshwater Ecosystems

	Chemical and Physical Effects	Biochemical and Biological Effects
	Suspended Solids	
Clays, silts, sand	Sedimentation, erosion, and abrasion turbidity (light reduction), habitat change	Respiratory interference habitat restriction, light limitation
Natural organic matter	Sedimentation, DO utilization	Food sources, DO effects
Wastewater organic particles	Sedimentation, DO utilization	DO effects, eutrophication, nutrient source
Toxicants sorbed to particles	All of the above	Toxicity
	Dissolved Solids	
Major inorganic salts	Salinity, buffering, precipitation, element ratios	Nutrient availability, succession, salt effects
Important nutrients		Eutrophication, DO production
Natural organic matter		DO effects and utilization
Wastewater organic matter		DO effects and utilization
Toxicants		Toxicity and effects on DO

From EPA (U.S. Environmental Protection Agency). *Suspended and Dissolved Solids Effects on Freshwater Biota: A Review,* Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, OR, EPA 600/3-77/042. 1977.

(EPA 1977). Suspended sediments decrease light penetration and photosynthesis, clog gills and filtering systems of aquatic organisms, reduce prey capture, reduce spawning, reduce survival of sensitive species, and carry adsorbed pollutants (Tables 3.3 through 3.5). Acute effects of suspended solids are commonly observed at 80,000 mg/L with death at 200,000 mg/L. Recovery is quick at lower exposures (EPA 1977). As the suspended sediments settle, they cover silt-free spawning substrates, suffocating embryos, and alter the sediment environment. Suspended solids reduce primary productivity and alter temperatures, thus affecting summer stratification. Solids should not reduce photosynthesis by more than 10% of the seasonal average, using the "light–dark" bottle method (APHA 1992). Reduced productivity may then reduce zooplankton populations. Desirable benthic species may be smothered, and tolerant species, such as oligochaetes, will increase in numbers. The sediment environment plays a major role in aquatic ecosystem functioning and overlying water quality (Wetzel 1975). These new bedded sediments may possess different chemical, physical, and biological characteristics from pre-impact sediments. So any alteration to the micro-, meio-, and macrobenthic communities, sorption and desorption dynamics of essential and toxic chemical species, and organic matter and nutrient cycling processes may profoundly influence the aquatic ecosystem (Power and Chapman 1992). As the rate of bedload sediment movement increases and the frequency of occurrence of bedload movement increases, the stress to the system increases.

Dissolved solids concentrations can often be very high in stormwaters and baseflows. The associated dissolved constituents consist primarily of road salts and salts from exposed soils. Though the major cations and anions are nontoxic to most species in relatively high concentrations, stormwaters may exceed threshold levels (EPA 1977) and alter ion ratios, which may cause chronic toxicity effects. In addition, toxic trace metal-metalloids such as selenium may be dissolved from natural soil matrices (as dramatically demonstrated in the San Joaquin Valley's Kesterson Reservoir of California), or dissolved zinc may be discharged from roof runoff components of urban runoff. Long-term and repeated exposures result as the dissolved species accumulate in interstitial water, bacteria, macrophytes, phytoplankton, and other food chain components (Burton et al. 1987; EPA 1977) and result in increased mortality, teratogenicity and other adverse effects (EPA 1977).

Table 3.4 Summary of Suspended Solids Effects on Aquatic Macroinvertebrates

Organisms	Effect	Suspended Solid Concentration	Source of Suspended Solids	Comment
Mixed populations	Lower summer populations		Mining area	
	Reduced populations to 25%	261–390 NTU (turbidity)	Log dragging	
	Densities 11% of normal	1000–6000 mg/L		Normal populations at 60 mg/L
	No organisms in the zone of setting	>5000 mg/L	Glass manufacturing	Effect noted 13 miles downstream
Chironomus and Tubificidae	Normal fauna replaced by species selection		Colliery	Reduction in light-reduced submerged plants
Chematopsyche (net spinners)	Number reduced	(High concentrations)	Limestone quarry	Suspended solids as high as 250 mg/L
Tricorythoides	Number increased		Limestone quarry	Due to preference for mud or silt
Mixed populations	90% increase in drift	80 mg/L	Limestone quarry	
	Reduction in numbers	40–200 NTU	Manganese strip mine	Also caused changes in density and diversity
Chironomidae	Increased drift with suspended sediment		Experimental sediment addition	
Ephemoptera, Simuliidae, Hydracarina	Inconsistent drift response to added sediment		Experimental sediment addition	

From EPA (U.S. Environmental Protection Agency). *Suspended and Dissolved Solids Effects on Freshwater Biota: A Review,* Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, OR, EPA 600/3-77/042. 1977.

Dissolved Oxygen

Historically, dissolved oxygen has received much attention when researchers investigate biological receiving water effects of pollutant discharges. Therefore, the earliest efforts to evaluate the potential problems caused by urban runoff included investigations of dissolved oxygen conditions in urban receiving waters.

Bacteria respond rapidly (within minutes) in temperate streams and lakes to their surrounding environment. Due to the low level of nutrients normally present, most of the indigenous bacteria are dominant. During a storm event, however, micro- to submicrogram levels of organic nutrients (e.g., carbon, nitrogen, phosphorus, and sulfur-containing compounds) suddenly increase by orders of magnitude. Consequently, bacterial reproduction and respiration rates increase dramatically; thus exerting biochemical oxygen demand (BOD). Oxygen depletion problems may occur during the high flow event, but it is likely more serious days later when associated with organic material affecting the sediment oxygen demand (Pitt 1979). BOD_5 levels may exceed 20 mg/L during storm events, which may result in anoxia in downstream receiving waters (Schueler 1987). Predicting this problem is complicated by toxicants that may be present and interfere with the BOD test (OWML 1982). Sediment resuspension contributes to both BOD and chemical oxygen demand (COD). BOD_5 values were elevated tenfold (10 to 20 days after a storm event) related to sediment oxygen demand (SOD). Stormwater dissolved oxygen (DO) levels less than 5 mg/L are common (Keefer et al. 1979).

Aquatic macrofauna are cold-blooded and sensitive to temperature changes. In cold water systems, sustained temperatures in excess of 21°C are stressful to resident biota. Many agricultural and urban watersheds contribute to thermal pollution by removing shade canopies over streams, and runoff temperatures increase rapidly as water flows over impervious surfaces (Schueler 1987).

Table 3.5 Summary of Suspended Solids Effects on Fish[a]

Fish (Special)	Effect	Concentration of Suspended Solids (mg/L)	Source of Suspended Materials
Rainbow trout (*Salmo gairdneri*)	Survived 1 day	80,000	Gravel washing
	Killed in 1 day	160,000	Gravel washing
	50% Mortality in 3½ weeks	4250	Gypsum
	Killed in 20 days	1000–2500	Natural sediment
	50% mortality in 16 weeks	200	
	1/5 mortality in 37 days	1000	Spruce fiber
	No deaths in 4 weeks	553	Cellulose fiber
	No deaths in 9–10 weeks	200	Gypsum
	20% mortality in 2–6 months	90	Coal washery waste
	No deaths in 8 months	100	Kaolin and diatomaceous earth
	No deaths in 8 months	50	Spruce fiber
	No increased mortality	30	Coal washery waste
	Reduced growth	50	Kaolin or diatomaceous earth
	Reduced growth	50	Wood fiber
	Fair growth	200	Coal washery waste
	"Fin-rot" disease	270	Coal washery waste
	"Fin-rot" disease	100	Diatomaceous earth
	No "fin-rot"	50	Wood fiber
	Reduced egg survival	(Siltation)	Wood fiber
	Total egg mortality in 6 days	1000–2500	
	Reduced survival of eggs	(Silting)	Wood fiber
	Supports populations	(Heavy loads)	Mining operations
	Avoid during migration	(Muddy waters)	Glacial silt
Brown trout (*Salmo trutta*)	Do not dig redds	(Sediment in gravel)	
	Reduced populations to 1/7 of clean streams	1000–6000	China-clay waste
Cutthroat trout (*Salmo clarkii*)	Abandon redds	(If silt is encountered)	
	Sought cover and stopped feeding	35	
Brook trout (*Salvelinus fontinalis*)	No effect on movement	(Turbidity)	
Golden shiner (*Notemigonus crysoleucas*)	Reaction	20,000–50,000	
	Death	50,000–100,000	
Carp (*Cyrinus carpio*)	Reaction	20,000	
	Death	175,000–250,000	
Largemouth black bass (*Micropterus salmoides*)	Reaction	20,000	
	Death	101,000 (average)	
Smallmouth bass (*Micropterus dolomieu*)	Successful nesting, spawning, hatching	(Sporadic periods of high turbidity)	

[a] See EPA 1977 for additional species-specific effect information.

From EPA (U.S. Environmental Protection Agency). *Suspended and Dissolved Solids Effects on Freshwater Biota: A Review,* Environmental Research Laboratory, U.S. Environmental Protection Agency, Corvallis, OR, EPA 600/3-77/042. 1977.

Acid precipitation and acid mine drainage cause NPS pollution problems in some parts of the United States which are, at times, aggravated by storm events. During the spring in areas where snows have accumulated, rain events intensify the snowmelt process. This results in pulses of low pH runoff and snowmelts which may be stressful or lethal to aquatic macrofauna, particularly the sensitive life stages of fish occurring during the spring spawning period.

Keefer et al. (1979) examined the data from 104 water quality monitoring sites near urban areas throughout the country for DO conditions. These stations were selected from more than 1000 nationwide monitoring stations operated by various federal and state agencies. They conducted analyses of daily DO data for 83 of these sites. About one half of the monitoring stations examined showed a 60% or greater probability of a higher than average dissolved oxygen deficit occurring at times of higher than average stream flow, or on days with rainfall. This result was based on daily data for entire water years; not all years at any given location exhibited this 60% probability condition. They found that the DO levels fell to less than 75% saturation at most of the stations that had this 60% or greater probability condition. They also found that DO concentrations of less than 5 mg/L were common. Keefer et al. (1979) examined hourly DO data at 22 nationwide sites to find correlations between flows and DO deficit. They found that for periods of steady low flows, the DO fluctuated widely on a daily cycle, ranging from 1 to 7 mg/L. During rain periods, however, the flow increased, of course, but the diurnal cycle of this DO fluctuation disappeared. The minimum DO dropped from 1 to 1.5 mg/L below the minimum values observed during steady flows, and remained constant for periods ranging from 1 to 5 days. They also reported that as the high flow conditions ended, the DO levels resumed diurnal cyclic behavior. About 50% of the stations examined in detail on an hour-by-hour basis would not meet a 5 mg/L DO standard, and about 25% of these stations would not even meet a 2.0 mg/L standard for 4-hour averages. The frequency of these violations was estimated to be up to five times a year per station.

Ketchum (1978) conducted another study in Indiana that examined DO depletion on a regional basis. Sampling was conducted at nine cities, and the project was designed to detect significant DO deficits in streams during periods of rainfall and runoff. The results of this study indicated that wet-weather DO levels generally appeared to be similar or higher than those observed during dry-weather conditions in the same streams. They found that significant wet-weather DO depletions were not observed, and due to the screening nature of the sampling program, more subtle impacts could not be measured.

Heaney et al. (1980), during their review of studies that examined continuous DO stations downstream from urbanized areas, indicated that the worst DO levels occurred after the storms in about one third of the cases studied. This lowered DO could be due to urban runoff moving downstream, combined sewer overflows, and/or resuspension of benthic deposits. Resuspended benthic deposits could have been previously settled urban runoff solids.

Pitt (1979) found that the BOD of urban runoff, after a 10- to 20-day incubation period, can be more than five to ten times the BOD of a 1- to 5-day incubation period (Figure 3.27). Therefore, urban runoff effects on DO may occur at times substantially different from the actual storm period and be associated with interaction between sediment and the overlying water column. It is especially important to use acclimated microorganisms for the BOD test seed for stormwater BOD analyses. The standard activated sludge seed may require substantial acclimation periods. Even in natural waters, several-day acclimation periods may be needed (see Lalor and Pitt 1998; P/R *in situ* test descriptions in Chapter 6).

Temperature

In-stream temperature increases have been noted in many studies as being adversely affected by urbanization. Rainwater flowing across heated pavement can significantly elevate stormwater temperatures. This temperature increase can be very detrimental in steams having sensitive cold-water fisheries. Removal of riparian vegetation can also increase in-stream water temperatures. Higher water temperatures increase the toxicity of ammonia and also affect the survival of pathogens. The temperature increases in urban streams are most important during the hot summer months when the natural stream temperatures may already be nearing critical conditions and when the stream flows are lowest. Pavement is also the hottest at this time and stormwater temperature increases are therefore the highest. Much of the habitat recovery efforts in urban streams focus on restoring an overstory for the streams to provide shading, refuge areas, and bank stability. Wet detention ponds in urban areas have also been shown to cause significant temperature increases. Grass-lined channels, however, provide some relief, compared to rock-lined or asphalt-lined drain-

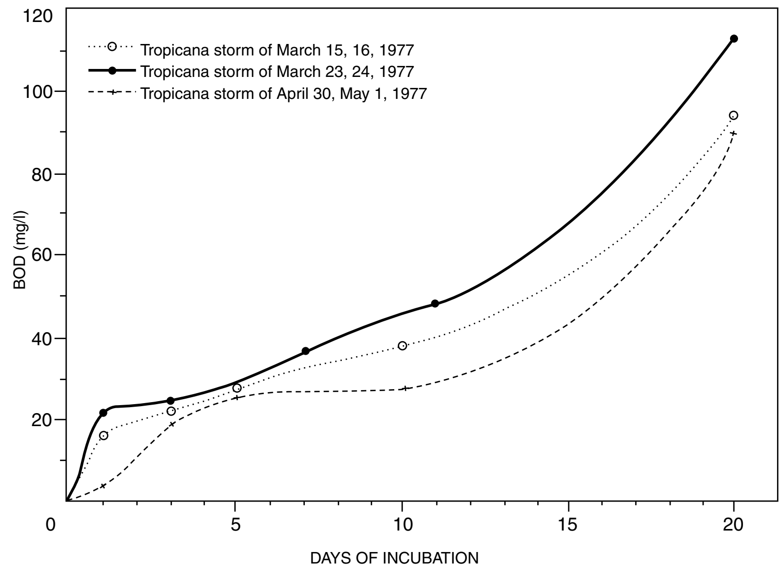

Figure 3.27 BOD rate curve for stormwater, showing dramatic increase after 10 days of incubation. (From Pitt, R. *Demonstration of Nonpoint Pollution Abatement through Improved Street Cleaning Practices,* EPA-600/2-79-161, U.S. Environmental Protection Agency, Cincinnati, OH. 270 pp. 1979.)

age channels. Since temperature is simple to monitor and is a critical stressor for many aquatic organics, it should be included in most monitoring efforts.

Nutrients

In general, urban stormwater is relatively low in organic matter and nutrients and high in toxicants. However, the nutrient levels in stormwaters can periodically be high and produce large mass discharges of nitrogen and phosphorus compounds (e.g., EPA 1977, 1983; Schueler 1987). Single spring storm events have been shown to contribute 90% of the annual phosphorus input into receiving impoundments. However, urban and agricultural runoff may contain nutrient concentrations which exceed the normal (predevelopment) ranges, and result in adverse responses such as cyanobacterial (blue-green algae) and green algal blooms. Many of the nutrients present in urban runoff are soluble and thus readily assimilated by planktonic organisms (Schueler 1987). Sources include rain, dry deposition, soils, fertilizers, and animal wastes. Impoundments receiving contaminated runoff, with retention times of 2 weeks or longer, may develop symptoms of eutrophication. Blue-green algal blooms can produce hepato- and neurotoxins implicated in cattle deaths, human liver cancer, and allergic responses (Zhang et al. 1991). As algal blooms eventually decompose, bacterial respiration may result in DO sags and anoxia, with associated fish kills.

A large amount of the nutrients enter receiving waters adsorbed to suspended solids (Lin 1972; Middlebrooks 1974; Carlile et al. 1974). These fractions will largely end up as bedded sediments which may or may not be subsequently released to overlying waters. The sediment nutrients may stimulate bacterial activity, ammonia production, and rooted macrophyte growth.

Toxicants

Heavy Metals

Stormwater runoff commonly contains elevated levels of metals and metalloids, particularly in urban areas (EPA 1983; Pitt et al. 1995; Schueler 1987). Some of these constituents are very toxic at relatively low concentrations (Table 3.6). The metals of principal concern that often occur in

Table 3.6 U.S. EPA Trace Metal Criteria for Human Health and Aquatic Life Beneficial Uses

Trace Metal Contaminant	Water Hardness (mg/L as $CaCO_3$)	Human[a] Ingestion (food/drink) (µg/L)	Ambient Life Criteria for Intermittent Exposure (µg/L)[b]	
			Threshold[c] Effect	Significant[d] Mortality
Copper	50	—	20	50–90
	100	—	35	90–150
	200	—	80	120–350
Cadmium	50	10	3	7–160
	100	10	6.6	15–350
	300	10	20	45–1070
Lead	50	50	150	350–3200
	100	50	360	820–7500
	200	50	850	1950–17850
Zinc	50	—	380	870–3200
	100	—	680	1550–4500
	200	—	1200	2750–8000
Nickel	—	13.4	—	—

[a] Derived from EPA drinking water criteria.
[b] EPA estimate of toxicity under intermittent, short-duration exposure (several hours once every several days).
[c] Concentration causing mortality to the most sensitive individual of the most sensitive species.
[d] Significant mortality shown as a range: 50% mortality in the most sensitive species, and mortality of the most sensitive individual in the species in the 25th percentile of sensitivity.

From EPA (U.S. Environmental Protection Agency). *Quality Criteria for Water.* EPA 440/5-86-001. U.S. Environmental Protection Agency, Washington, D.C. May 1986.

urban runoff are arsenic, cadmium, copper, lead, mercury, and zinc (EPA 1983). Metal bioavailability is reduced in waters of higher hardness (Table 3.6) by sorption to solids and by stormwater dilution. However, acute and chronic effects have been attributed to stormwater metals (Ray and White 1979; Ellis 1992). The highest metal concentrations are not always associated with the "first flush," but are better correlated with the peak flow period (Heaney 1978). Most metals are bound to street and parking area particulates and subsequently deposited in stream and lake sediments (Pitt et al. 1995). Sediment metal concentrations are dependent on particle size (Wilber and Hunter 1980). Wilber and Hunter (1980) suggest that larger particle sizes are better indicators of urban inputs since they are less affected by scouring. Zinc and copper are often present in runoff as soluble forms (Schueler 1987; Pitt et al. 1995).

Predicting detrimental effects from water or sediment metal concentration or loading data is difficult due to the myriad of processes which control bioavailability and fate. Speciation, availability, and toxicity are affected by pH, redox potential, temperature, hardness, alkalinity, solids, iron and manganese oxyhydroxides, sulfide fractions, and other organic-inorganic chelators. These constituents and conditions are often rapidly changing during a storm event and processes which increase and decrease bioavailability (e.g., loss of sulfide complexes and formation of oxyhydroxide complexes) may occur simultaneously. This makes accurate modeling of toxicity difficult, if not impossible.

Episodic exposures of organisms to stormwaters laden with metals can produce stress and lethality (see also Chapter 6). Ray and White (1976) observed fish death days after exposure and miles downstream after metals were diluted to nondetectable levels. Ellis et al. (1992) showed amphipods bioaccumulated zinc from episodic, *in situ* exposures. Repeated exposures increased their sensitivity, and mortality was observed 3 weeks after the storm event.

Toxic Organic Compounds

The types and concentrations of toxic organic compounds that are in stormwaters are driven primarily by land use patterns and automobile activity in the watershed. Most nonpesticide organic compounds originate as washoff from impervious areas in commercial areas having large numbers of automobile startups and/or other high levels of vehicle activities, including vehicle maintenance operations and heavily traveled roads. The compounds of most interest are the polycyclic aromatic hydrocarbons (PAHs). Other organics include phthalate esters (plasticizers) and aliphatic hydrocarbons. Other compounds frequently detected in residential and agricultural areas are cresol constituents (and other wood preservatives), herbicides, and insecticides. Many of these organic compounds are strongly associated with the particulate fraction of stormwater. Volatile organic compounds (VOCs) are rarely found in urban runoff. While most organics are not detected or are detected at low µg/L concentrations, some are acutely toxic, including freshly applied pesticides and photoactivated PAHs (Skalski 1991; Oris and Giesy 1986). The extent of detrimental impact from these constituents has not been well documented, but likely is significant in some areas.

Environmental Fates of Runoff Toxicants

The fate of runoff toxicants after discharge significantly determines their associated biological effects. If the pollutants are discharged in a soluble form and remain in solution, they may have significant acute toxicity effects on fish, for example. However, if discharged soluble pollutants form insoluble complexes or sorb onto particulates, chronic toxicity effects associated with contaminated sediments are more likely. For many of the metallic and organic toxicants discharged in urban runoff, the particulate fractions are much greater than the soluble fractions (Pitt et al. 1995). Particulate forms of pollutants may remain in suspension, if their settling rates are low and the receiving water is sufficiently turbulent. However, polluted sediments are common in many urban and agricultural streams, indicating significant accumulations of runoff particulate pollutants (Pitt 1995).

Tables 3.7 through 3.9 summarize the importance of various environmental processes for the aquatic fates of some runoff heavy metals and organic priority pollutants, as described by Callahan et al. (1979). Photolysis (the breakdown of the compounds in the presence of sunlight) and volatilization (the transfer of the materials from the water into the air as a gas or vapor) are not nearly as important as the other mechanisms for heavy metals. Chemical speciation (the formation of chemical compounds) is very important in determining the solubilities of the specific metals. Sorption (adsorption is the attachment of the material onto the outside of a solid, and absorption is the attachment of the material within a solid) is very important for all of the heavy metals shown. Sorption can typically be the controlling mechanism affecting the mobility and the precipitation of most heavy metals. Bioaccumulation (the uptake of the material into organic tissue) can occur for all of the heavy metals shown. Biotransformation (the change of chemical form of the metal by organic processes) is very important for some of the metals, especially mercury, arsenic, and lead. In many cases, mercury, arsenic, or lead compounds discharged in forms that are unavailable can be accumulated in aquatic sediments. They are then exposed to various benthic organisms that can biotransform the material through metabolization to methylated forms, which can be highly toxic and soluble.

Tables 3.8 and 3.9 also summarize various environmental fates for some of the toxic organic pollutants found in typical runoff from human-modified watersheds, mainly various phenols, polycyclic aromatic hydrocarbons (PAHs), and phthalate esters. Photolysis may be an important fate process for phenols and PAHs but is probably not important for the phthalate esters. Oxidation or hydrolysis may be important for some phenols. Volatilization may be important for some phenols and PAHs. Sorption is an important fate process for most of the materials, except for phenols. Bioaccumulation, biotransformation, and biodegradation are important for many of these organic materials.

Pathogens

Water Environment & Technology (1996) reported that the latest National Water Quality Inventory released by the EPA only showed a slight improvement in the attainment of beneficial uses in

Table 3.7 Importance of Environmental Processes on the Aquatic Fates of Selected Urban Runoff Heavy Metals

Environmental Process	Arsenic	Cadmium	Copper	Mercury	Lead	Zinc
Photolysis	Not important	Not important	Not important	May be important in some aquatic environments	Determines the form of lead entering the aquatic system	Not important
Chemical speciation	Important in determining distribution and mobility[a]	Complexation with organics; most important in polluted waters	Complexation with organics; most important in polluted waters	Conversion to complex species; HgS will precipitate in reducing sediments	Determines which solid phase controls solubility	Complexation predominates in polluted waters
Volatilization	Important when biological activity or highly reducing conditions produce AsH_3 or methyl-arsenic	Not important	Not important	Important	Not important	Not important
Sorption	Sorption onto clays, oxides, and organic material important	Sorption onto organic materials, clays, hydrous iron and manganese oxides most important	Can reduce Cu mobility and enrich suspended and bed sediments; sorption onto organics in polluted waters, clay minerals or hydrous iron and manganese oxides	Strongest onto organic material, results in partitioning of mercury into suspended and bed sediments	Adsorption to inorganic solids, organic materials and hydrous iron and manganese oxides control mobility of lead	Strong affinity for hydrous metal oxides, clays, and organic matter; adsorption increases with pH
Bioaccumulation	Most important at lower trophic levels; toxicity limits bioaccumulation	Biota strongly bioaccumulate cadmium	Biota strongly bioaccumulate copper	Occurs by many mechanisms, most connected to methylated forms of mercury	Biota strongly bioaccumulates lead	Zinc is strongly bioaccumulated
Biotransformation	Arsenic can be metabolized to organic arsenicals	Not methylized biologically, organic ligands may affect solubility and adsorption	Source Cu complexes may be metabolized; organic ligands are important in sorption and complexation processes	Can be metabolized by bacteria to methyl and dimethyl forms which are quite mobile	Biomethylation of lead in sediments can remobilize lead	Not evident; organic ligands of biological origin may affect solubility and adsorption

[a] Conversion of As^{3+} and As^{5+} and organic complexation most important.

From Callahan, M.A. et al. *Water Related Environmental Fates of 129 Priority Pollutants.* U.S. Environmental Protection Agency, Monitoring and Data Support Division, EPA-4-79-029a and b. Washington, D.C. 1979.

Table 3.8 Importance of Environmental Processes on the Aquatic Fates of Various Polycyclic Aromatic Hydrocarbons and Phthalate Esters

Environmental Process[a]	Anthracene	Fluoranthene	Phenanthrene	Diethyl Phthalate (DEP)	Di-n-Butyl Phthalate (DBP)	Bis (2-Ethyl-hexyl) Phthalate (DEHP)	Butyl Benzyl Phthalate (BBP)
Photolysis	Dissolved portion may undergo rapid photolysis	Dissolved portion may undergo rapid photolysis	Dissolved portion may undergo rapid photolysis	Not important	Not important	Not important	Not important
Volatilization	May be competitive with adsorption	May be competitive with adsorption	May be competitive with adsorption	Not important	Not important	Not important	Not important
Sorption	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Adsorbs onto suspended solids; movement by suspended solids is important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process	Sorbed onto suspended solids and biota; complexation with humic substances most important transport process
Bioaccumulation	Short-term process; is readily metabolized	Short-term process; is readily metabolized	Short-term process; is readily metabolized	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)	Variety of organisms accumulate phthalates (lipophilic)
Biotransformation	Readily metabolized by organisms and biodegradation, probably ultimate fate mechanisms	Readily metabolized by organisms and biodegradation, probably ultimate fate mechanism	Readily metabolized by organisms and biodegradation, probably ultimate fate mechanisms	Can be metabolized	Can be metabolized	Can be metabolized	Can be metabolized

[a] Oxidation and hydrolysis are not important fate mechanisms for any of these compounds.

From Callahan, M.A. et al. *Water Related Environmental Fates of 129 Priority Pollutants.* U.S. Environmental Protection Agency, Monitoring and Data Support Division, EPA-4-79-029a and b. Washington, D.C. 1979.

Table 3.9 Importance of Environmental Processes on the Aquatic Fates of Various Phenols and Pyrene

Environmental Process	Phenol	Pentachlorophenol (PCP)	2,4,6-Trichlorophenol	2,4-Dimethyl Phenol (2,4-Xylenol)	Pyrene
Photolysis	Photooxidation may be important degradation process in aerated, clear, surface waters	Reported to occur in natural waters; important near water surface	Reported, but importance is uncertain	May be important degradation process in clear aerated surface waters	Dissolved portion may undergo rapid photolysis
Oxidation	Metal-catalyzed oxidation may be important in aerated surface waters	Not important	Not important	Metal-catalyzed oxidation may be important in aerated surface waters	Not important
Volatilization	Possibility of some phenol passing into the atmosphere	Not important	Not important	Not important	Not as important as adsorption
Sorption	Not important	Sorbed by organic litter in soil and sediments	Potentially important for organic material; not important for clays	Not important	Adsorption onto suspended solids important; movement by suspended solids important
Bioaccumulation	Not important	Bioaccumulates in numerous aquatic organisms	Not important	Not important	Short-term process not significant; metabolized over long term
Biotransformation	Very significant	Can be metabolized to other phenol forms	Reported in soil and sewage sludge; uncertain for natural surface waters	Inconclusive information	Readily metabolized; biodegradation probably ultimate fate process

From Callahan, M.A. et al. *Water Related Environmental Fates of 129 Priority Pollutants.* U.S. Environmental Protection Agency, Monitoring and Data Support Division, EPA-4-79-029a and b. Washington, D.C. 1979.

the nation's waters. Urban runoff was cited as the leading source of problems in estuaries, with nutrients and bacteria as the primary problems. Problems in rivers and lakes were mostly caused by agricultural runoff, with urban runoff the third ranked source for lakes and the fourth ranked source for rivers. Bacteria, siltation, and nutrients were the leading problems in the nation's rivers and lakes.

Pathogens in stormwater are a significant concern potentially affecting human health. The use of indicator bacteria is controversial for stormwater, as is the assumed time of typical exposure of swimmers to contaminated receiving waters. However, recent epidemiological studies have shown significant health effects associated with stormwater-contaminated marine swimming areas. Proto8 zoan pathogens, especially associated with likely sewage-contaminated stormwater, are also a public health concern.

Fecal indicators (i.e., fecal coliforms, fecal streptococci, *Escherichia coli*, and enterococci) are usually found in elevated concentrations in stormwater runoff, greatly exceeding water quality criteria and standards for primary and secondary contact (MWCOG 1984). This suggests that fecal pathogen levels are also elevated, though significant correlations with fecal coliforms are tenuous (EPA 1986). Die-off of fecal organisms in receiving waters during summer months is relatively rapid, with 99% dying within 24 to 48 hours (Burton 1985). However, fecal microorganisms also accumulate in sediments where survival is extended for weeks to months (Burton et al. 1987). Recent sediment bacteriological analyses conducted by UAB in local Birmingham (AL) area urban lakes have found elevated pore water concentrations (several hundred to several thousand organisms/100 mL) of *E. coli* and enterococci extending to at least 0.1 m into the sediments. Also, when gently disturbed, the water layer over the sediments is also found to significantly increase in microorganism concentrations. *In situ* die-off studies also indicated that bacteria sedimentation may be a more important fate mechanism of stormwater bacteria than die-off (Easton 2000).

Good correlations between the incidence of gastroenteritis in swimmers and *E. coli* and entero8 cocci concentrations in water have resulted in new recreational water criteria (EPA 1986). High fecal microorganism concentrations in stormwaters originate from wastes of wildlife, pets, livestock, septic systems, and combined sewer overflows (CSOs). The ecological effects of these inputs of fecal organisms are unknown; however, public health is at risk in swimming areas that receive stormwaters.

Urban Bacteria Sources

The Regional Municipality of Ottawa–Carleton (1972) recognized the importance of rooftop, street surface, and field runoff in contributing bacteria contaminants to surface waters in the Ottawa area. Gore & Storrie/Proctor and Redfern (1981) also investigated various urban bacteria sources affecting the Rideau River. They examined dry-weather continuous coliform sources, the resuspen8 sion of contaminated river bottom sediments, exfiltration from sanitary sewers, and bird feces. These sources were all considered in an attempt to explain the relatively high dry-weather coliform bacteria concentrations found in the river. They concluded, however, that stormwater runoff is the most probable source for the wet-weather and continuing dry-weather bacteria concentrations in the Rideau River. The slow travel time of the river water usually does not allow the river to recover completely from one rainstorm before another begins.

The Regional Municipality of Ottawa–Carleton (1972) noted the early Ottawa activities in correcting stormwater and sanitary sewage cross-connections. Since that time, many combined sewer overflows have also been eliminated from the Rideau River. Loijens (1981) stated that, as a result of sewer separation activities, only one overflow remained active by 1981 (Clegg Street). During river surveys in 1978 and 1979 in the vicinity of this outfall, increased bacteria levels were not found. Gore & Storrie/Proctor and Redfern (1981) stated that there was no evidence that combined sewer overflows are causing the elevated fecal coliform bacteria levels in the river. Environment Canada (1980), however, stated that high dry-weather bacteria density levels, especially when considering the fecal coliform to fecal streptococci ratio, constitutes presumptive evidence of low-volume sporadic inputs of sanitary sewage from diverse sources into the downstream Rideau River sectors.

Street surfaces have been identified as potential major sources of urban runoff bacteria. Pitt and Bozeman (1982) found that parking lots, street surfaces, and sidewalks were the major con8 tributors of indicator bacteria in the Coyote Creek watershed in California. Gupta et al. (1981) found high concentrations of fecal coliforms at a highway runoff site in Milwaukee. This site was entirely impervious and located on an elevated bridge deck. The only likely sources of fecal coliforms at this site were atmospheric deposition, bird droppings, and possibly feces debris falling from livestock trucks or other vehicles.

Several studies have found that the bacteria in stormwater in residential and light commercial areas were from predominantly nonhuman origins. Geldreich and Kenner (1969) stated that the fecal coliforms in stormwater are from dogs, cats, and rodents in city areas, and from farm animals and wildlife in rural areas. Qureshi and Dutka (1979) found that there may be an initial flush of animal feces when runoff first develops. The most important source, however, may be feces bacteria that are distributed in the soil and not the fresh feces washing off the impervious surfaces.

Some studies have investigated vegetation sources of coliform bacteria. For example, Geldreich (1965) found that the washoff of bacteria from vegetation does not contribute significant bacteria to the runoff. They also found that most of the bacteria on vegetation is of insect origin. Geldreich et al. (1980) found that recreation activities in water bodies also increase the fecal coliform and fecal streptococci concentrations. These organisms of intestinal origin will concentrate in areas near the shore or in areas of stratification. Fennell et al. (1974) found that open dumps containing domestic refuse can be a reservoir of *Salmonella* bacteria that can be spread to nearby water bodies by foraging animals and birds.

When a drainage basin has much of its surface paved, the urban runoff bacteria concentrations can be expected to peak near the beginning of the rainfall event and then decrease as the event continues. Initial high levels of bacteria may be associated with direct flushing of feces from paved surfaces. These feces are from dogs defecating on parking lots and street areas and from birds roosting on rooftops. When a drainage area has a lot of landscaped areas or open land, relatively high bacteria concentrations in the urban runoff may occur throughout the rain event associated with continuous erosion of contaminated soils.

Fecal Coliform to Fecal Streptococci Bacteria Ratios

Geldreich (1965) found that the ratio of fecal coliform to fecal streptococci bacteria concen8 trations may be indicative of the probable fecal source. In fresh human fecal material and domestic wastes, he found that the fecal coliform densities were more than four times the fecal streptococcal densities. However, this ratio for livestock, poultry, dogs, cats, and rodents was found to be less than 0.6. These ratios must be applied carefully because of the effects of travel time and various chemical changes (especially pH) on the die-off rates of the component bacteria. This can result in the ratio changing, as the fecal coliform organisms tend to die faster than the fecal streptococcal bacteria. As a generality, he stated that fecal coliform to fecal streptococci ratios greater than 4 indicate that the bacteria pollution is from domestic wastes, which are composed mostly of human fecal material, laundry wastes, and food refuse. If the ratio is less than 0.6, the bacteria are probably from livestock or poultry in agricultural areas or from stormwater runoff in urban areas. He found that agricultural and stormwater runoff can be differentiated by studying the types of fecal strep8 tococci bacteria found in the water samples. Geldreich and Kenner (1969) further stressed the importance of using this ratio carefully. They stressed that samples must be taken at the wastewater outfalls. At these locations, domestic waste, meat packing wastes, stormwater discharges, and feedlot drainage contain large numbers of fecal organisms recently discharged from warm-blooded animals. Once these organisms are diffused into the receiving stream, however, water temperature,

Table 3.10 Fecal Coliform to Fecal Streptococci Bacteria Population Ratios in Study Area

Source Areas	FC/FS Ratio
Rooftop runoff	0.5
Vacant land sheetflow	0.3
Parking lot sheetflow	0.2
Gutter flows	0.2
Average of source area values	0.3
Rideau River segment	
A	1.2
B	0.6
C	0.5
D	0.5
E	1.0
Average of river segment values	0.7
River swimming beaches	
Strathcona	2.8
Brantwood	2.3
Brighton	2.1
Mooney's Bay	1.7
Average of swimming beach values	2.2

From Pitt, R. *Urban Bacteria Sources and Control by Street Cleaning in the Lower Rideau River Watershed.* Rideau River Stormwater Management Study Technical Report. Prepared for the Ontario Ministry of the Environment, Environment Canada, Regional Municipality of Ottawa-Carleton, City of Ottawa, and Nepean. 1983.

organic nutrients, toxic metals, and adverse pH values may alter the relationship between the indicator organisms. This ratio should only be applied within 24 hours following the discharge of the bacteria.

Feachem (1975) examined how these ratios could be used with bacteria observations taken over a period of time. Because the fecal coliform and fecal streptococci bacteria die-off rates are not the same, the ratio gradually changes with time. He found that bacteria are predominantly from human sources if the FC/FS ratios are initially high (greater than 4) and then decrease with time. Nonhuman bacteria sources would result in initially low FC/FS ratios (less than 0.7), which then rise with time.

Pitt (1983) examined the FC/FS bacteria population ratios observed in the Rideau River study area in Ottawa, as shown in Table 3.10. These ratios were divided into groups corresponding to source area samples, Rideau River water samples, and water samples collected at the swimming beaches farther downstream. The source area sheet-flow samples contained the most recent pollu8 tion, while the river segment and beach samples contained "older" bacteria. The initial source area samples all had ratios of less than 0.7. However, the river averages ranged from 0.5 to 1.2, and the beach samples (which may be "older" than the river samples) ranged from 1.7 to 2.8. These ratios are seen to start with values less than 0.7 and increase with time. Based on Feachem's (1975) work, this would indicate that the major bacteria sources in the Rideau River are from nonhuman sources. Periodic high bacteria ratios in the river and at the beaches could be caused by the greater die-off ratio of fecal streptococci as compared to fecal coliform. The observed periodic high Rideau River FC/FS ratios (which can be greater than 4) may therefore be from old, nonhuman fecal discharges and not from fresh human fecal discharges.

Human Health Effects of Stormwater

There are several mechanisms whereby stormwater exposure can cause potential human health problems. These include exposure to stormwater contaminants at swimming areas affected by stormwater discharges, drinking water supplies contaminated by stormwater discharges, and the consumption of fish and shellfish that have been contaminated by stormwater pollutants. Understanding the risks associated with these exposure mechanisms is difficult and not very clear. Receiving waters where human uses are evident are usually very large, and the receiving waters are affected by many sanitary sewage and industrial point discharges, along with upstream agricultural nonpoint discharges, in addition to the local stormwater discharges. In receiving waters having only stormwater discharges, it is well known that inappropriate sanitary and other wastewaters are also discharging through the storm drainage system. These "interferences" make it especially difficult to identify specific cause-and-effect relationships associated with stormwater discharges alone, in contrast to the many receiving water studies that have investigated ecological problems that can more easily study streams affected by stormwater alone. Therefore, much of the human risk assessment associated with stormwater exposure must use theoretical evaluations relying on stormwater characteristics and laboratory studies in lieu of actual population studies. However, some site investigations, especially related to swimming beach problems associated with nearby stormwater discharges, have been conducted and are summarized (from Lalor and Pitt 1998) in the following discussion.

Contact recreation in pathogen-contaminated waters has been studied at many locations. The sources of the pathogens are typically assumed to be sanitary sewage effluent, or periodic industrial discharges from certain food preparation industries (especially meat packing and fish and shellfish processing). However, several studies have investigated pathogen problems associated with stormwater discharges. It has generally been assumed that the source of pathogens in stormwater are from inappropriate sanitary connections. However, stormwater unaffected by these inappropriate sources still contains high counts of pathogens that are also found in surface runoff samples from many urban surfaces. Needless to say, sewage contamination of urban streams is an important issue that needs attention during a receiving water investigation.

Inappropriate Sanitary Sewage Discharges into Urban Streams

Urban stormwater runoff includes waters from many other sources that find their way into storm drainage systems, besides from precipitation. There are cases where pollutant levels in storm drainage are much higher than they would otherwise be because of excessive amounts of contam8 inants that are introduced into the storm drainage system by various non-stormwater discharges. Additionally, baseflows (during dry weather) are also common in storm drainage systems. Dry-weather flows and wet-weather flows have been monitored during numerous urban runoff studies. These studies have found that discharges observed at outfalls during dry weather were significantly different from wet-weather discharges and may account for the majority of the annual discharges for some pollutants of concern from the storm drainage system.

In many cases, sanitary sewage was an important component (although not necessarily the only component) of the dry-weather discharges from the storm drainage systems. From a human health perspective (associated with pathogens), it may not require much raw or poorly treated sewage to cause a receiving water problem. However, at low discharge rates, the DO receiving water levels may be minimally affected. The effects these discharges have on receiving waters is therefore highly dependent on many site-specific factors, including frequency and quantity of sewage discharges and the creek flows. In many urban areas, the receiving waters are small creeks in completely developed watersheds. These creeks are the most at risk from these discharges as dry baseflows may be predominantly dry-weather flows from the drainage systems. In Tokyo (Fujita 1998), for example, numerous instances were found where correcting inappropriate sanitary sewage discharges resulted in the urban streams losing all of their flow. In cities adjacent to large receiving waters, these discharges likely have little impact (such as DO impacts from Nashville, TN, CSO discharges on the Cumberland River, as studied by Cardozo et al. 1994). The presence of pathogens from raw or poorly treated sewage in urban streams, however, obviously presents a potentially serious public health threat. Even if the receiving waters are not designated as water contact recreation, children are often seen playing in small city streams.

There have been a few epidemiology studies describing the increased health risks associated with contaminated dry-weather flows affecting public swimming beaches. The following discussion presents an overview of the development of water quality criteria for water contact recreation, plus the results of a recent epidemiological study that specifically examined human health problems associated with swimming in water affected by stormwater. In most cases, the levels of indicator organisms and pathogens causing increased illness were well within the range found in urban streams.

Runoff Pathogens and Their Sanitary Significance

The occurrence of *Salmonella* biotypes is typically low, and their reported density is less than one organism/100 mL in stormwater. *Pseudomonas aeruginosa* are frequently encountered at densities greater than 10 organisms/100 mL, but only after rains. The observed ranges of concen8 trations and percent isolations of bacterial biotypes vary significantly from site to site and at the same location for different times. Many potentially pathogenic bacteria biotypes may be present in urban runoff. Because of the low probability of ingestion of urban runoff, many of the potential human diseases associated with these biotypes are not likely to occur. The pathogenic organisms of most concern in urban runoff are usually associated with skin infections and body contact. The most important biotype causing skin infections would be *P. aeruginosa*. This biotype has been detected frequently in most urban runoff studies in concentrations that may cause infections. However, there is little information associating the cause and effect of increased *P.* concentrations with increased infections. *Shigella* may be present in urban runoff and receiving waters. This pathogen, when ingested in low numbers, can cause dysentery.

Salmonella

Salmonella has been reported in some, but not all, urban stormwaters. Qureshi and Dutka (1979) frequently detected *Salmonella* in southern Ontario stormwaters. They did not find any predictable patterns of *Salmonella* isolations; they were found throughout the various sampling periods. Olivieri et al. (1977a) found *Salmonella* frequently in Baltimore runoff, but at relatively low concentrations. Typical concentrations were from 5 to 300 *Salmonella* organisms/10 L. The concentrations of *Salmonella* were about ten times higher in the stormwater samples than in the urban stream receiving the runoff. The researchers also did not find any marked seasonal variations in *Salmonella* concen8 trations. Almost all of the stormwater samples that had fecal coliform concentrations greater that 2000 organisms/100 mL had detectable *Salmonella* concentrations, while about 275 of the samples having fecal coliform concentrations less than 200 organisms/100 mL had detectable *Salmonella*.

Quite a few urban runoff studies have not detected *Salmonella*. Schillinger and Stuart (1978) found that *Salmonella* isolations were not common in a Montana subdivision runoff study and that the isolations did not correlate well with fecal coliform concentrations. Environment Canada (1980) stated that *Salmonella* were virtually absent from Ottawa storm drainage samples in 1979. It concluded that *Salmonella* are seldom present in significant numbers in Ottawa urban runoff. The types of *Salmonella* found in southern Ontario were *S. thompson* and *S. typhimurium* var. *copenhagen* (Qureshi and Dutka 1979).

Olivieri et al. (1977b) stated that the primary human enteric disease producing *Salmonella* biotypes associated with the ingestion of water include *S. typhi* (typhoid fever), *S. paratyphi* (paratyphoid fever), and *Salmonella* species (salmonellosis). These biotypes are all rare except for

Salmonella sp. The dose of *Salmonella* sp. required to produce an infection is quite large (approx8 imately 10^5 organisms). The salmonellosis health hazard associated with water contact in urban streams is believed to be small because of this relatively large infective dose. If 2 L of stormwater having typical *Salmonella* concentrations (10 *Salmonella* organisms/10 L) is ingested, less than 0.001 of the required infective dose would be ingested. If a worst-case *Salmonella* stormwater concentration of 10,000 organisms/10 L occurred, the ingestion of 20 L of stormwater would be necessary for an infective dose. They stated that the low concentrations of *Salmonella*, coupled with the unlikely event of consuming enough stormwater, make the *Salmonella* health hazard associated with urban runoff small.

Staphylococcus

Staphylococcus aureus is an important human pathogen it can cause boils, carbuncles, abscesses, and impetigo on skin on contact. Olivieri et al. (1977b) stated that the typical concentrations of *Staphylococci* are not very high in urban streams. They also noted that there was little information available relating the degree of risk of staph infections with water concentrations. They concluded that *Staphylococcus aureus* appears to be the most potentially hazardous pathogen associated with urban runoff, but there is no evidence available that skin, eye, or ear infections can be caused by the presence of this organism in recreational waters. They concluded that there is little reason for extensive public health concern over recreational waters receiving urban storm runoff containing staph organisms.

Shigella

Olivieri et al. (1977b) stated that there is circumstantial evidence that *Shigella* is present in urban runoff and receiving waters and could present a significant health hazard. *Shigella* species causing bacillary dysentery are one of the primary human enteric disease-producing bacteria agents present in water. The infective dose of *Shigella* necessary to cause dysentery is quite low (10 to 100 organisms). Because of this low required infective dose and the assumed presence of *Shigella* in urban waters, it may be a significant health hazard associated with urban runoff.

Streptococcus

Streptococcus faecalis and atypical *S. faecalis* are of limited sanitary significance (Geldreich 1976). *Streptococcus* determinations on urban runoff are most useful for identifying the presence of *S. bovis* and *S. equinus,* which are specific indicators of nonhuman, warm-blooded animal pollution. However, it is difficult to interpret fecal streptococcal data when their concentrations are lower than 100 organisms/100 mL because of the ubiquitous occurrence of *S. faecalis* var. *liquifaciens.* This biotype is generally the predominant streptococcal biotype occurring at low fecal streptococcal concentrations.

Pseudomonas aeruginosa

Pseudomonas is reported to be the most abundant pathogenic bacteria in urban runoff and streams (Olivieri et al. 1977b). This pathogen is associated with eye and ear infections and is resistant to antibiotics. Oliveri et al. also stated that past studies have failed to show any relationships between *P. aeruginosa* concentrations in bathing waters and ear infections. However, *Pseudomonas* concentrations in urban runoff are significantly higher (about 100 times) than the values associated with past bathing beach studies. Cabelli et al. (1976) stated that *P. aeruginosa* is indigenous in about 15% of the human population. Swimmer's ear or other *Pseudomonas* infections may, therefore, be caused by trauma to the ear canals associated with swimming and diving, and not exposure to *Pseudomonas* in the bathing water.

Environment Canada (1980) stated that there is preliminary evidence of the direct relationship between very low levels of *P. aeruginosa* and an increase in incidents of ear infections in swimmers. It stated that a control level for this *Pseudomonas* biotype of between 23 and 30 organisms/100 mL was considered. Cabelli et al. (1976) stated that *P. aeruginosa* densities greater than 10 organisms/100 mL were frequently associated with fecal coliform levels considerably less than 200 organisms/100 mL. *Pseudomonas aeruginosa* densities were sometimes very low when the fecal coliform levels were greater than 200 organisms/100 mL. An average estimated *P. aeruginosa* density associated with a fecal coliform concentration of 200 organisms/100 mL is about 12/100 mL. It further stated that *P. aeruginosa* by itself cannot be used as a basis for water standards for the prevention of enteric diseases during recreational uses of surface waters. The determinations of this biotype should be used in conjunction with fecal coliform or other indicator organism concentrations for a specific location. It recommended that bathing beaches that are subject to urban runoff be temporarily closed until the *P. aeruginosa* concentrations return to a baseline concentration.

Campylobacter

Koenraad et al. (1997) investigated the contamination of surface waters by *Campylobacter* and its associated human health risks. They reported that campylobacteriosis is one of the most frequently occurring acute gastroenteritis diseases in humans. Typical investigations have focused on the consumption of poultry, raw milk, and untreated water as the major sources of this bacterial illness. Koenraad et al. (1997) found that human exposures to *Campylobacter*-contaminated surface waters is likely a more important risk factor than previously considered. In fact, they felt that *Campylobacter* infections may be more common than *Salmonella* infections. The incidence of campylobacteriosis due to exposure to contaminated recreational waters has been estimated to be between 1.2 to 170 per 100,000 individuals. The natural habitat of *Campylobacter* is the intestinal tract of warm-blooded animals (including poultry, pigs, cattle, gulls, geese, pigeons, magpies, rodents, shellfish, and even flies). It does not seem to multiply outside of its host, but it can survive fairly well in aquatic environments. It can remain culturable and infective for more than 2 months under ideal environ8 mental conditions. Besides runoff, treated wastewater effluent is also a major source of *Campylobacter* in surface waters. Sanitary wastewater may contain up to 50,000 MPN of *Campylobacter* per 100 mL, with 90 to 99% reductions occurring during typical wastewater treatment.

Cryptosporidium, Giardia, *and* Pfiesteria

Protozoa became an important public issue with the 1993 *Cryptosporidium*-caused disease outbreak in Milwaukee when about 400,000 people become ill from drinking contaminated water. Mac Kenzie et al. (1994) prepared an overview of the outbreak, describing the investigation of the causes of the illness and the number of people affected. They point out that *Cryptosporidium*8 caused disease in humans was first documented in 1976, but had received little attention and no routine monitoring. *Cryptosporidium* is now being monitored routinely in many areas and is the subject of much research concerning its sources and pathways. At the time of the Milwaukee outbreak, both of the city's water treatment plants (using water from Lake Michigan) were operating within acceptable limits, based on required monitoring. However, at one of the plants (which delivered water to most of the infected people), at the time of the outbreak the treated water underwent a large increase in turbidity (from about 0.3 NTU to about 1.5 NTU) that was not being well monitored (the continuous monitoring equipment was not functioning, and values were obtained only every 8 hours). More than half of the residents receiving water from this plant became ill. The plant had recently changed its coagulant from polyaluminum chloride to alum, and equip8 ment to assist in determining the correct chemical dosages was not being used. The finished water had apparently relatively high levels of *Cryptosporidium* because some individuals became ill after drinking less than 1 L of water.

Cryptosporidium oocysts have often been found in untreated surface waters, and it was thought that *Cryptosporidium* oocysts entered the water treatment supply before the increase in turbidity was apparent. MacKenzie et al. (1994) point out that monitoring in the United Kingdom has uncovered sudden, irregular, community-wide increases in cryptosporidiosis that were likely caused by waterborne transmission. They also stated that the source of the *Cryptosporidium* oocysts was speculative, but could have included cattle feces contamination in the Milwaukee and Menomonee Rivers, slaughterhouse wastes, and human sewage. The rivers were also swelled by high spring rains and snowmelt runoff that may have aided the transport of upstream *Cryptosporidium* oocysts into the lake near the water intakes.

The *Journal of the American Water Works Association* has published numerous articles on protozoa contamination of drinking water supplies. Crockett and Haas (1997) describe a watershed investigation to identify sources of *Giardia* and *Cryptosporidium* in the Philadelphia watershed. They describe the difficulties associated with monitoring *Cryptosporidium* and *Giardia* in surface waters because of low analytical recoveries and the cost of analyses. Large variations in observed protozoa concentrations made it difficult to identify major sources during the preliminary stages of their investigations. They do expect that wastewater treatment plant discharges are a major local source, although animals (especially calves and lambs) are likely significant contributors. Combined sewer overflows had *Giardia* levels similar to raw sewage, but the CSOs had much less *Cryptosporidium* than the raw sewage. LeChevallier et al. (1997) investigated *Giardia* and *Cryptosporidium* in open reservoirs storing finished drinking water. This gave them an opportunity to observe small increases in oocyst concentrations associated from nonpoint sources of contamination from the highly controlled surrounding area. They observed significantly larger oocyst concentrations at the effluent (median values of 6.0 *Giardia*/100 L and 14 *Cryptosporidium*/100 L) in the reservoirs than in the influents (median values of 1.6 *Giardia*/100 L and 1.0 *Cryptosporidium*/100 L). No human wastes could influence any of the tested reservoirs, and the increases were therefore likely caused by wastes from indigenous animals or birds, either directly contaminating the water or through runoff from the adjacent wooded areas.

A Management Training Audioconference Seminar on *Cryptosporidium* and Water (MTA 1997) was broadcast in May of 1997 to familiarize state and local agencies about possible *Cryptosporidium* problems that may be evident as a result of the EPA's Information Collection Rule which began in July of 1997. This regulation requires all communities serving more than 100,000 people to monitor their source water for *Cryptosporidium* oocysts. If the source water has more than 10 *Cryptosporidium* oocysts/L, the finished water must also be monitored. It is likely that many source waters will be found to be affected by *Cryptosporidium*. The researchers reviewed one study that found the percentage of positive samples of *Cryptosporidium* in lakes, rivers, and springs was about 50 to 60% and about 5% in wells. In contrast, the percentage of samples testing positive for *Giardia* was about 10 to 20% in lakes and rivers, and very low in springs and wells.

Special human health concerns have also been recently expressed about *Pfiesteria piscicida*, a marine dinoflagellate that is apparently associated with coastal eutrophication caused by runoff nutrients (Maguire and Walker 1997). Dramatic blooms and resulting fish kills have been associated with increased nutrient loading from manure-laden runoff from large livestock feedlot operations. This organism has garnered much attention in the popular press, usually called the "cell from hell" (Zimmerman 1998). It has been implicated as causing symptoms of nausea, fatigue, memory loss, and skin infections in south Atlantic coastal bay watermen. *Pfiesteria* and *Pfiesteria*-like organisms have also been implicated as the primary cause of many major fish kills and fish disease events in Virginia, Maryland, North Carolina, and Delaware. In August 1997, hundreds of dead and dying fish were found in the Pocomoke River, near Shelltown, MD, in the Chesapeake Bay, prompting the closure of a portion of the river. Subsequent fish kills and confirmed occurrences of *Pfiesteria* led to further closures of the Manokin and Chicamacomico Rivers. The Maryland Department of Health and Mental Hygiene also presented preliminary evidence that adverse public health effects could result from exposure to the toxins released by *Pfiesteria* and *Pfiesteria*-like organisms. The increasing numbers of fish kills of Atlantic menhaden (an oily, non-game fish) motivated Maryland's governor to appoint a Citizens *Pfiesteria* Action Commission. The commission convened a forum of noted scientists to examine the existing information on *Pfiesteria.* The results of the State of Maryland's *Pfiesteria* monitoring program are available on the Maryland Department of Natural Resources' Web site: http://www.dnr.state.md.us/pfiesteria/.

Pfiesteria has a complex life cycle, including at lease 24 flagellated, amoeboid, and encysted stages. Only a few of these stages appear to be toxic, but their complex nature makes them difficult to identify by non-experts (Maguire and Walker 1997). *Pfiesteria* spends much of its life span in a nontoxic predatory form, feeding on bacteria and algae, or as encysted dormant cells in muddy sediment. Large schools of oily fish (such as the Atlantic menhaden) trigger the encysted cells to emerge and excrete toxins. These toxins make the fish lethargic, so the fish remain in the area where the toxins attack the fish skin, causing open sores to develop. The *Pfiesteria* then feed on the sloughing fish tissue. Unfortunately, people working in the water during these toxin releases may also be affected (Zimmerman 1998).

Researchers suggest that excessive nutrients (causing eutrophication) increase the algae and other organic matter that the *Pfiesteria* and Atlantic menhaden use for food. The increased concentrations of *Pfiesteria* above natural background levels increase the likelihood of toxic problems. Maguire and Walker (1997) state that other factors are also apparently involved, including stream hydraulics, water temperature, and salinity. They feel that *Pfiesteria* is only one example of the increasing threats affecting coastal ecosystems that are experiencing increased nutrient levels. Most of the resulting algal blooms only present nuisance conditions, but a small number can result in human health problems (mostly as shellfish poisonings). The increased nutrient discharges are mostly associated with agricultural operations, especially animal wastes from large poultry and swine operations. In the Pocomoke River watershed, the Maryland Department of Natural Resources estimates that about 80% of the phosphorus and 75% of the nitrogen load is from agricultural sources. Urban runoff may also be a causative factor of eutrophication in coastal communities, especially those having small enclosed coastal lagoons or embayments, or in rapidly growing urban areas. Zimmerman (1998) points out that the Chesapeake Bay area is one of the country's most rapidly growing areas, with the population expected to increase by 12% by the year 2010.

Viruses

It is believed that approximately half of all waterborne diseases are of viral origin. Unfortunately, it is very difficult and time-consuming to identify viruses from either environmental samples or sick individuals. When the EPA conducted its extensive epidemiological investigations of freshwater and marine swimming beaches in the 1980s, two viruses common to human gastrointestinal tracts (coliphage and enterovirus) were evaluated as potential pathogen indicators. These two indicators did not show good correlations between their presence and the incidence of gastroenteritis. Viruses tend to survive for slightly longer periods in natural waters than do Gram-negative bacteria. It is believed that the high correlation observed between gastroenteritis and the presence of enterococci may be because the Gram-positive enterococci's longer survival more closely mimics viral survival. Therefore, enterococci may serve as a good recreational water indicator for the presence of viral pathogens.

RECEIVING WATER EFFECT SUMMARY

Recent studies have combined chemical-physical characterizations of water and sediment with biosurveys and laboratory/*in situ* toxicity surveys (low and high flow) to effectively characterized major water column and sediment stressors (Burton and Rowland 1999; Burton et al. 1998; Dyer and White 1996; Burton and Moore 1999). Suspended solids, ammonia, sediments, temperature, PAHs, and/or stormwater runoff were observed to be primary stressors in these test systems. These primary stressors could not have been identified without low and high flow and sediment quality assessments both in the laboratory and field. It is apparent that to determine the role of chemicals as stressors in the receiving waters, the role of other stressors (both natural and anthropogenic) must be assessed (see also Chapters 6 and 8).

Johnson et al. (1996) and Herricks et al. (1996a,b) describe a structured tier testing protocol to assess both short-term and long-term wet-weather discharge toxicity. The protocol recognizes that the test systems must be appropriate to the time-scale of exposure during the discharge. Therefore, three time-scale protocols were developed, for intra-event, event, and long-term exposures.

There is a natural tendency in the popular "weight-of-evidence" or "sediment quality triad" approaches to look for "validation" of one assessment tool with another (see also Chapters 6 and 8). For example, matching a toxic response in a WET test with that of an impaired community gives a greater weight of evidence. This does not, however, necessarily "validate" the results (or invalidate, if there are differences) (Chapman 1995). Natural temporal changes in aquatic populations at different sites within a study system need not be the same (Power et al. 1988; Resh 1988; Underwood 1993); therefore, predictions of effect or no-effect from WET testing of reference sites may be in error. Each monitoring tool (i.e., chemical, physical, and indigenous biota characterizations, laboratory and field toxicity, and bioaccumulation) provides unique and often essential information (Burton 1995; Chapman et al. 1992; Burton et al. 1996; Baird and Burton 2001). If the responses of each of the biological tools disagree, it is likely due to species differences or a differing stressor exposure dynamic/interaction. These critical exposures issues can be characterized through a systematic process of separating stressors and their respective dynamics into low and high flow and sediment compartments using both laboratory and field exposures. Then, a more efficient and focused assessment can identify critical stressors and determine their ecological significance with less uncertainty than the more commonly used approaches. The chronic degradation potential of complex ecosystems receiving multiple stressors cannot be adequately evaluated without a comprehensive assessment that characterizes water, sediment, and biological dynamics and their interactions.

Because most sites have multiple stressors (physical, chemical, and biological), it is essential that the relative contributions of these stressors be defined to design effective corrective measures. The integrated laboratory and field approach rigorously defines the exposures of organisms (media of exposure and contaminant concentration), separating it into overlying water, surficial sediment, historical sediment, and interstitial water. The degree of contaminant-associated toxicity can best be assessed using a combination of laboratory and field screening methods which separate stressors (i.e., a Stressor Identification Evaluation (SIE) approach) (Burton et al. 1996), into different, major stressor categories, including metals, nonpolar organics, photoinduced toxicity from PAHs, ammonia, suspended solids, predators, dissolved oxygen, and flow. There is much research to be done to refine these approaches, but the tools are there to make ecologically relevant assessments of aquatic ecosystem contamination with reasonable certainty.

The effects of urban runoff on receiving water aquatic organisms or other beneficial uses is also very site specific. Different land development practices may create substantially different runoff flows. Different rain patterns cause different particulate washoff, transport, and dilution conditions. Local attitudes also define specific beneficial uses and desired controls. There are also a wide variety of water types receiving urban and agricultural runoff, and these waters all have watersheds that are urbanized to various degrees. Therefore, it is not surprising that runoff effects, though generally dramatic, are also quite variable and site specific.

Previous attempts to identify runoff problems using existing data have not generally been conclusive because of differences in sampling procedures and the common practice of pooling data from various sites or conditions. It is therefore necessary to carefully design comprehensive, long-term studies to investigate runoff problems on a site-specific basis. Sediment transport, deposition, and chemistry play key roles in receiving waters and need additional research. Receiving water aquatic biological conditions, especially compared to unaffected receiving waters, should be studied in preference to laboratory bioassays.

These specific studies need to examine beneficial uses directly, and not rely on published water quality criteria and water column measurements alone. Published criteria are usually not applicable to urban runoff because of the sluggish nature of runoff and the unique chemical speciation of its components.

The long-term aquatic life effects of runoff are probably more important than short-term effects associated with specific events. The long-term effects are probably related to the deposition and accumulation of toxic sediments, or the inability of the aquatic organisms to adjust to repeated exposures to high concentrations of toxic materials or high flow rates.

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