capture particles in the 0.2- to 0.4-μm-diameter range, not by impaction but by filtering the air that moves through the snow flakes as they fall to the ground.

Most of the contamination of snow in urban areas likely occurs after it lands on the ground. Table 3-8 shows the flow-weighted mean concentrations of pollutants found in undisturbed falling snow compared to snow found in urban snow cover (Bennett et al., 1981). Pitt and McLean (1986) also measured snowpack contamination as a function of distance from a heavily traveled road passing through a park. The contaminants in the snow were at much greater concentrations near the road (the major source of blown contamination on the snow) than farther away. (The pollutant levels in the fresh fallen snow are generally a small fraction of the levels in the snow collected from urban study areas.) Pierstorff and Bishop (1980) also analyzed freshly fallen snow and compared the quality to snow stored at a snow dump site. They concluded that "pollutant levels at the dump site are the result of environmental input occurring after the snow falls." Some pollutants in snowmelt have almost no atmospheric sources. For example, Oliver et al. (1974) found negligible amounts of chlorides in samples of snow from rooftops, indicating that the high chloride level found in the snowmelt runoff water comes almost entirely from surface sources (i.e., road salting). Similar roadside snowpack observations along city park roads by Pitt and McLean (1986) also indicated the strong association of road salt with snowpack chloride levels.

Runoff and Pollutant Loading from Snowmelt

Snowmelt events can exhibit a first flush, in which there are higher concentrations of contaminants at the beginning compared to the total event averaged concentration. The enrichment of the first portion of a snowmelt event by soluble pollutants may be due to snowpack density changes, where water percolation and melt/freeze events that occur in the snowpack cause soluble pollutants to be flushed from throughout the snowpack to concentrate at the bottom of the pack (Colbeck, 1981). This concentrated layer leaves the snowpack as a highly concentrated pulse, as snow melts from the bottom due to warmth from the ground (Oberts, 1994).

TABLE 3-8 Comparison of Flow-Weighted Pollutant Concentration Means of Snow Samples from Boulder, Colorado

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Note: The units are mg/L. SOURCE: Bennett et al. (1981). Permission pending.

When it rains on snow, heavy pollutant loads can be produced because both soluble and particulate pollutants are melted from the snowpack simultaneously. Also, the large volume of melt plus rain can wash off pollutants that have accumulated on various surfaces such as roads, parking lots, roofs, and saturated soil surfaces. The intensity of runoff from a rain-on-snow event can be greater than a summer thunderstorm because the ground is saturated or frozen and the rapidly melting snowpack provides added runoff volume (Oberts, 1994).

Figure 3-28 compares the runoff volumes associated with snowmelts alone to those associated with snowmelts mixed with rain from monitoring at an industrial area in Toronto (Pitt and McLean, 1986). Rain with snowmelt contributes over 80 percent of the total cold-weather event runoff volume.

Whether pollutant loadings are higher or lower for snowmelt than for rainfall depends on the particular pollutant and its seasonal prevalence in the environment. For example, the high concentrations of dissolved solids found in snowmelt are usually caused by high chloride concentrations that stem from the amount of de-icing salt used. Figure 3-29 is a plot of the chloride concentrations in the influent to the Monroe Street detention pond in Madison, Wisconsin. Chloride levels are negligible in the non-winter months but increase dramatically when road salting begins in the fall, and remain high through the snow melting period, even extending another month or so after the snowpack in the area has melted. Bennett et al. (1981) found that suspended solids and COD loadings for snowmelt runoff were about one-half of those for rainfall. Nutrients were much lower for snowmelt, while the loadings for lead were about the same for both forms of precipitation. Oberts (1994) reports that much of the annual pollutant yields from event flows in Minneapolis is accounted for by end-of-winter major melts. End-of-winter melts yielded 8 to 20 percent of the total phosphorous and total lead annual load in Minnesota. Small midwinter melts accounted for less than 5 percent of the total loads. Box 3-8 shows mass pollutant discharges for a study site in Toronto and emphasizes the significance of snowmelt discharges on the total annual storm drainage discharges.

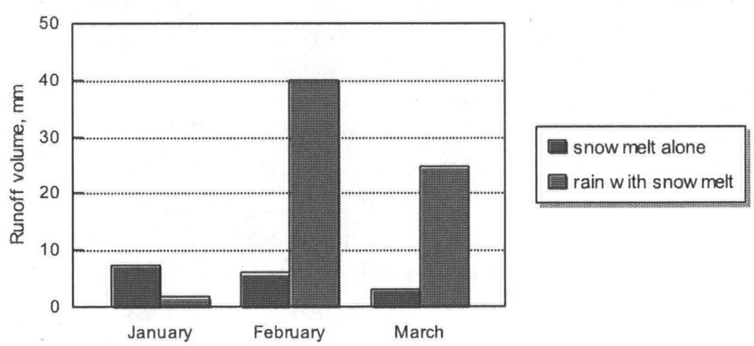

FIGURE 3-28 Runoff volumes for snowmelt events alone and when rain falls on melting snow packs (Toronto industrial area). SOURCE: Pitt and McLean (1986).

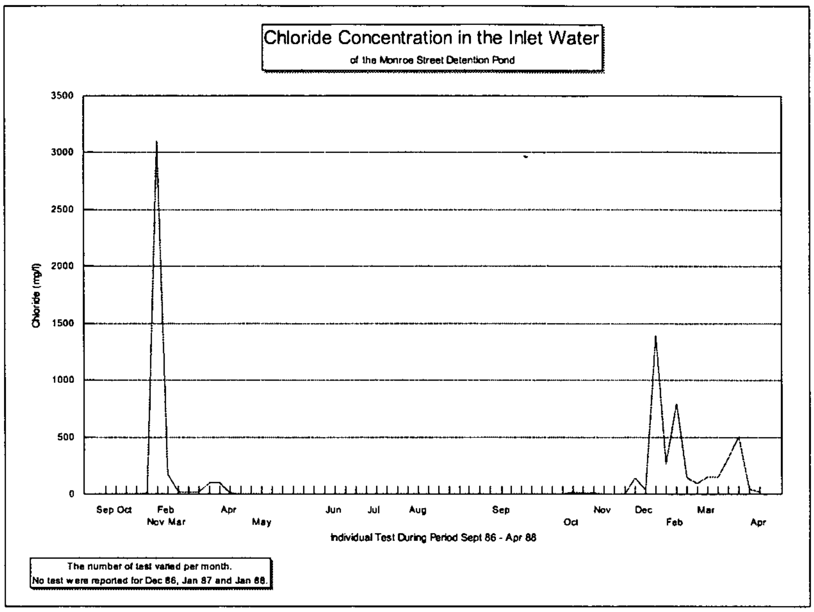

FIGURE 3-29 Monroe Street detention pond chloride concentration of influent (1986–1988). SOURCE: House et al. (1993).

Atmospheric Deposition

The atmosphere contains a diverse array of contaminants, including metals (e.g., copper, chromium, lead, mercury, zinc), nutrients (nitrogen, phosphorus), and organic compounds (e.g., PAHs, polychlorinated biphenyls, pesticides). These contaminants are introduced to the atmosphere by a variety of sources, including local point sources (e.g., power plant stacks) and mobile sources (e.g., motor vehicles), local fugitive emissions (e.g., street dust and wind-eroded materials), and transport from non-local areas. These emissions, composed of gases, small particles (aerosols), and larger particles, become entrained in the atmosphere and subject to a complex series of physical and chemical reactions (Schueler, 1983).

Atmospheric contaminants are deposited on land and water in two ways—termed wet deposition and dry deposition. Wet deposition (or wetfall) involves the sorption and condensation of pollutants to water drops and snowflakes followed by deposition with precipitation. This mechanism dominates the deposition of gases and aerosol particles. Dry deposition (or dryfall) is the direct transfer of contaminants to land or water by gravity (particles) or by diffusion (vapor and particles). Dry deposition occurs when atmospheric turbulence is not sufficient to counteract the tendency of particles to fall out at a rate governed, but not exclusively determined, by gravity (Schueler, 1983).

BOX 3-8
The Contribution of Dry Weather Discharges and Snowmelt to Overall Runoff in Toronto, Ontario

An extensive analysis of all types of stormwater flow—for both dry and wet weather—was conducted in Toronto in the mid-1980s (Pitt and McLean, 1986). The Toronto Area Watershed Management Strategy study included comprehensive monitoring in a residential/commercial area and an industrial area for summer stormwater, warm season dry weather flows, snowmelt, and cold season dry weather flows. In addition to the outfall monitoring, detailed source area sheet flow monitoring was also conducted during rain and snowmelt events to determine the relative magnitude of pollutant sources. Particulate accumulation and wash-off tests were also conducted for a variety of streets in order to better determine their role in contaminant contributions.

Tables 3-9 and 3-10 summarize Toronto residential/commercial and industrial urban runoff median concentrations during both warm and cold weather, respectively. These tables show the relative volumes and concentrations of wet weather and dry weather flows coming from the different land uses. The bacteria densities during cold weather are substantially less than during warm weather, but are still relatively high; similar findings were noted during the NURP studies (EPA, 1983). However, chloride concentrations and dissolved solids are much higher during cold weather. Early spring stormwater events also contain high dissolved solids concentrations. Cold weather runoff accounted for more than half of the heavy metal discharges in the residential/commercial area, while warm weather discharges of zinc were much greater than the cold weather discharges for the industrial area. Warm weather flows were also the predominant sources of phosphorus for the industrial area.

One of the interesting observations is that, at these monitoring locations, warm weather stormwater runoff only contributed about 20 to 30 percent of the total annual flows being discharged from the separate stormwater outfalls. The magnitudes of the base flows were especially surprising, as these monitoring locations were research sites to investigate stormwater processes and were carefully investigated to ensure that they did not have significant inappropriate discharges before they were selected for the monitoring programs.

In comparing runoff from the industrial and residential catchments, Pitt and McLean (1986) observed that concentrations of most constituents in runoff from the industrial watershed were typically greater than the concentrations of the same constituents in the residential runoff. The only constituents with a unit-area yield that were lower in the industrial area were chlorides and total dissolved solids, which was attributed to the use of road de-icing salts in residential areas. Annual yields of several constituents (total solids, total dissolved solids, chlorides, ammonia nitrogen, and phenolics) were dominated by cold weather flows, irrespective of the land use.

A comparison of the Toronto sheet flow data from the different land-use areas indicated that the highest concentrations of lead and zinc were found in samples collected from paved areas and roads during both rain runoff and snowmelt (Pitt and McLean, 1986). Fecal coliform values were significantly higher on sidewalks and on, or near, roads during snowmelt sampling, likely because these areas are where dogs would be walked in winter conditions. In warm weather, dog walking would be less concentrated into these areas. The concentrations for total solids from grass or bare open areas were reduced dramatically during snowmelt compared to rain runoff, an indication of the reduced erosion and the poor delivery of particulate pollutants during snowmelt periods. Cold weather sheet flow median concentrations of particulate solids for the grass and open areas (80 mg/L) were much less than the TSS concentrations observed during warm weather runoff (250 mg/L) for these same areas. Snowmelt total solids concentrations also increased in areas located near roads due to the influence of road salting on dissolved solids concentrations. In the residential areas, streets were the most significant source of snowmelt solids, while yards and open areas were the major sources of nutrients. Parking and storage areas contributed the most snowmelt pollutants in the industrial area. An analysis of snow samples taken along a transect of a snowpack adjacent to an industrial road showed that the pollutant levels decreased as a function of distance from the roadway. At distances greater than 3 to 5 meters from the edge of the

BOX 3-8 Continued

snowpack, the concentrations were relatively constant. Novotny et al. (1986) sampled along a transect of a snowpack by a freeway in Milwaukee. They also found that the concentration of constituents decreased as the distance from the road increased. Most of the measured constituents, including total solids and lead, were at or near background levels at 30 meters or more from the road.

TABLE 3-9 Median Pollutant Concentrations Observed at Toronto Outfalls during Warm Weather[1]

Measured Parameter	Baseflow		Stormwater	
	Residential	Industrial	Residential	Industrial
Stormwater volume (m^3/ha/season)	—	—	950	1500
Baseflow volume (m^3/ha/season)	1700	2100	—	—
Total residue	979	554	256	371
Total dissolved solids	973	454	230	208
Suspended solids	<5	43	22	117
Chlorides	281	78	34	17
Total phosphorus	0.09	0.73	0.28	0.75
Phosphates	<0.06	0.12	0.02	0.16
Total Kjeldahl nitrogen (organic N plus NH_3)	0.9	2.4	2.5	2.0
Ammonia nitrogen	<0.1	<0.1	<0.1	<0.1
Chemical oxygen demand	22	108	55	106
Fecal coliform bacteria (#/100 mL)	33,000	7,000	40,000	49,000
Fecal strep. bacteria (#/100 mL)	2,300	8,800	20,000	39,000
Pseudo. aeruginosa bacteria (#/100 mL)	2,900	2,380	2,700	11,000
Cadmium	<0.01	<0.01	<0.01	<0.01
Chromium	<0.06	0.42	<0.06	0.32
Copper	0.02	0.05	0.03	0.06
Lead	<0.04	<0.04	<0.06	0.08
Zinc	0.04	0.18	0.06	0.19
Phenolics (μg/L)	<1.5	2.0	1.2	5.1
α-BHC (ng/L)	17	<1	1	3.5
γ-BHC (lindane) (ng/L)	5	<2	<1	<1
Chlordane (ng/L)	4	<2	<2	<2
Dieldrin (ng/L)	4	<5	<2	<2
Pentachlorophenol (ng/L)	280	50	70	705

[1]Values are in mg/L unless otherwise indicated. Warm weather samples were obtained during the late spring, summer, and early fall months when the air temperatures were above freezing and no snow was present.

continues next page

BOX 3-8 Continued

TABLE 3-10 Median Pollutant Concentrations Observed at Toronto Outfalls during Cold Weather[1]

Measured Parameter	Base flow		Snow melt	
	Residential	Industrial	Residential	Industrial
Stormwater volume (m^3/ha/season)	—	—	1800	830
Base flow volume (m^3/ha/season)	1100	660	—	—
Total residue	2230	1080	1580	1340
Total dissolved solids	2210	1020	1530	1240
Suspended solids	21	50	30	95
Chlorides	1080	470	660	620
Total phosphorus	0.18	0.34	0.23	0.50
Phosphates	<0.05	<0.02	<0.06	0.14
Total Kjeldahl nitrogen (organic N plus NH_3)	1.4	2.0	1.7	2.5
Ammonia nitrogen	<0.1	<0.1	0.2	0.4
Chemical oxygen demand	48	68	40	94
Fecal coliform bacteria (#/100 mL)	9800	400	2320	300
Fecal strep bacteria (#/100 mL)	1400	2400	1900	2500
Pseudomonas aeruginosa bacteria (#/100 mL)	85	55	20	30
Cadmium	<0.01	<0.01	<0.01	0.01
Chromium	<0.01	0.24	<0.01	0.35
Copper	0.02	0.04	0.04	0.07
Lead	<0.06	<0.04	0.09	0.08
Zinc	0.07	0.15	0.12	0.31
Phenolics (mg/L)	2.0	7.3	2.5	15
α-BHC (ng/L)	NA	3	4	5
γ-BHC (lindane) (ng/L)	NA	NA	2	1
Chlordane (ng/L)	NA	NA	11	2
Dieldrin (ng/L)	NA	NA	2	NA
Pentachlorophenol (ng/L)	NA	NA	NA	40

[1]Values are in mg/L unless otherwise indicated. Cold weather samples were obtained during the winter months when the air temperatures were commonly below freezing. Snowmelt samples were obtained during snowmelt episodes and when rain fell on snow.
NA, not analyzed

As atmospheric contaminants deposit, they can exert an influence on stormwater in several ways. Contaminants deposited by wetfall are directly conveyed to stormwater while those in dryfall can be washed off the land surface. For both processes, the atmospheric load of contaminants is strongly influenced by characteristics such as the amount of impervious surface, the magnitude and proximity of emission sources, wind speed and direction, and precipitation magnitude and frequency (Schueler, 1983). Deposition rates can depend on the type of contaminant and can be site-specific. The relationships between atmospheric deposition and stormwater quality are, however, not well understood and difficult to determine. Following are a few illustrative examples.

Southern California

Several studies have addressed atmospheric deposition in Southern California (e.g., Lu et al., 2003; Harris and Davidson, 2005; Stolzenbach et al., 2007). Stolzenbach et al. and Lu et al. conclude the following *for this region:*

- the major source of contaminants to the atmosphere in this region is associated with resuspended dust, primarily from roads,
- contaminants in resuspended dust may reflect historical as well as current sources and distant as well as local sources,
- atmospheric loadings to the receiving water are primarily the result of chronic daily dry deposition of large particles greater than 10 μm in size on the watershed rather than directly on a waterbody,
- significant spatial variability occurs in trace metal mass loadings and deposition fluxes, particularly along transportation corridors along the coast and the mountain slopes of the airshed,
- significant diurnal and seasonal variations occur in the deposition of trace metals, and
- atmospheric deposition of metals is a significant component of contaminant loading to waterbodies in the region relative to other point and nonpoint sources.

Harris and Davidson (2005) have reported that traditional sources of lead to the south coast air basin of California accounted for less than 15 percent of the lead exiting the basin each year. They resolve this difference by considering that lead particles deposited during the years of leaded gasoline use are resuspended as airborne lead at this time, some decades after their original deposition. This result indicates that lead levels in the soil will remain elevated for decades and that resuspension of this lead will remain a major source of atmospheric lead well into the future.

Sabin et al. (2005) assessed the contribution of trace metals (chromium, copper, lead, nickel, and zinc) from atmospheric deposition to stormwater runoff in a small impervious urban catchment in the Los Angeles area. Dry deposition contributed 90 percent or more of the total deposition inside the catchment, indicating the dominance of dry deposition in semi-arid regions such as Los Angeles. Deposition potentially accounted for from 57 to 90 percent of the total trace metals in stormwater in the study area, demonstrating that atmospheric deposition can be an important source of trace metals in stormwater near urban centers.

San Francisco

Dissolved copper is toxic to phytoplankton, the base of the aquatic food chain. Copper and other metals are released in small quantities when drivers depress their brakes. The Brake Pad Partnership (http://www.suscon.org/brakepad/index/asp) has conducted studies to determine how much copper is released as wear debris, and how it travels through the air and streets to surface waters. A comprehensive and complex model of copper loads to and of transport and reactions in San Francisco Bay was developed (Yee and Franz, 2005). Objectives were to provide daily loadings of flow, TSS, and copper to the bay and to estimate the relative contribution of brake pad wear debris to copper in the bay. The modeling results (Rosselot, 2006a) indicated that an estimated 47,000 kg of copper was released to the atmosphere in the Bay Area in 2003. Of this amount, 17,000 kg Cu/yr was dry-deposited in subwatersheds; 3,200 kg Cu/yr was wet-deposited in subwatersheds; 1,200 kg Cu/yr was dry-deposited directly to bay waters; and 1,300 kg Cu/yr was wet-deposited directly to bay waters. The remaining 24,000 kg Cu/yr remained airborne until it left the Bay Area. The contribution of copper from brake pads to the bay is estimated to range from 10 to 35 percent of the total copper input, with the best estimate being 23 percent (Rosselot, 2006a,b).

Washington, D.C., Metropolitan Area

Schueler (1983) investigated the atmospheric deposition of several contaminants in Washington, D.C., and its surrounding areas in the early 1980s. The contaminants assessed included trace metals (cadmium, copper, iron, lead, nickel, and zinc), nutrients (nitrogen and phosphorus), solids, and organics as measured collectively by BOD and COD. Dryfall solids loading increased progressively from rural to urban sites. A similar trend was observed for total phosphorus, total nitrogen, and trace metal dry deposition rates. Wet deposition rates exhibited few consistent regional patterns.

The relative importance of wet and dry deposition varied considerably with each contaminant and each site. For example, most of the nitrogen was supplied by wet deposition while most of the phosphorus was delivered via dry deposition. If a contaminant is deposited primarily by wet deposition, it is likely that a major fraction of it will be rapidly entrained in urban runoff.

Atmospheric sources were estimated to contribute from 70 to 95 percent of the total nitrogen load to urban runoff and 20 to 35 percent of the total phosphorus load. Overall, atmospheric deposition appeared to be a moderate source of pollutants in urban runoff. However, with the exception of nitrogen, atmospheric deposition was not the major source.

Average annual atmospheric deposition rates suggested a general trend toward greater deposition rates from rural to suburban to urban sites. This pattern was most pronounced for dry deposition. Wet deposition was the most important deposition mechanism for total nitrogen, nitrate, organic nitrogen, COD, copper, and zinc. Dry deposition was most important for most soil-related constituents, such as total solids, iron, lead, total phosphorus, and orthophosphate.

Measurements of rainfall pH showed median values between 4.0 and 4.1 at all stations and during all seasons. Increased mobilization of trace metals from urban surfaces caused by acid rain was noted at several monitoring sites.

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Relationships between atmospheric deposition rates and the quality of urban stormwater are complex and cannot be generalized regionally or temporally. Site-specific measurements or reliable estimates of (1) contaminant sources, (2) atmospheric particle size and contaminant concentrations, (3) deposition rates and mechanisms, (4) land surface characteristics, (5) local and regional hydrology and meteorology, and (6) contaminant concentrations in stormwater are needed to assess management decisions to improve stormwater quality. Transportation is a major source of metals (lead in gasoline, zinc in tires, copper in brake pads). The results of the modeling of copper in San Francisco and its watershed demonstrate the feasibility of modeling the impact of a source, in this case copper input by atmospheric deposition, on water quality in a receiving waterbody.

BIOLOGICAL RESPONSES TO URBANIZATION

As discussed in Chapter 1, the biological integrity of aquatic ecosystems is influenced by five major categories of environmental stressors: (1) chemical, (2) hydrologic, (3) physical (e.g., habitat), (4) biological (e.g., disease, alien species), and (5) energy-related factors (e.g., nutrient dynamics). Recent studies on biological assemblages in urban or urbanizing waters have begun to examine how stormwater stressors limit biological potential along various urban gradients (Horner et al., 2003; Carter and Fend, 2005; Meador et al., 2005; Barbour et al., 2008; Purcell et al., in press). Advances in biological monitoring and assessment over the past two decades have enabled much of this research. Today, many states and tribes use biological data to directly measure their aquatic life beneficial uses and have developed numeric biocriteria that are institutionalized in their water quality standards. Most of these approaches compare biology and stressors to suites of reference sites (Hughes, 1995; Stoddard et al., 2006), which can vary from near-pristine areas to agricultural landscapes. While this section focuses on streams because of the wealth of data, similar work is being performed on other waterbody types such as wetlands (Mack and Micacchion, 2007) and estuaries, both of which are susceptible to stormwater pollutants such as metals because of their depositional nature (Morrisey et al., 2000).

Aquatic life beneficial uses are based on achieving aquatic *potential* given feasible restorative actions. Because such potential may vary substantially across a region depending on land use and other factors, some states have adopted tiered aquatic life uses (see Box 2-1). The potential of many urban streams is likely to be something less than "biological integrity" (the ultimate goal of the CWA) or even "fishable–swimmable" goals, which are the interim goals of the CWA. Indeed, there is a near-universal, negative association between biological assemblages in streams and increasing urbanization, to the extent that it has been termed the "Urban Stream Syndrome" (Walsh et al., 2005). Recent investigations that have quantified the responses of macroinvertebrates and other biological assemblages along multiple measures of urban/stormwater stressors have discussed how best to set aquatic life goals for urban streams (Booth and Jackson, 1997; Bernhardt and Palmer, 2007). One of the most important contributions to this debate has been the development of the Biological Condition Gradient (BCG) concept by EPA. The BCG is an attempt to anchor and standardize interpretations of biological conditions and to unify biological monitoring results across the United States in order to advance the use of tiered aquatic life beneficial uses. This section summarizes the characteristic biological responses to urban gradients, within the framework of the BCG, and it reviews evidence of biological responses within the aforementioned five major categories of environmental stressors.

Biological Condition Gradient

The BCG framework is an ecological model of how structural and functional components of biological assemblages change along gradients of increasing stressors of many kinds (Davies and Jackson, 2006). Ecological systems have some common general attributes related to their structure and function that form the basis for how biological organisms respond to stressors in the environment. Over the past 20 years, development of biological indicators nationwide has taken advantage of these repeatable biological responses to stress; however, state benchmarks often have varied substantially, even between adjacent states. To gain consistency, the EPA convened a national workgroup of EPA Regions, States, and Tribes to develop the BCG—a standardized, nationally applicable model that defines important attributes of biological assemblages and describes how these attributes change along a gradient of increasing stress from pristine environments to severely impaired conditions (Figure 3-30; Davies and Jackson, 2006). The goals of this work were to improve national consistency in the rating and application of biological assessment tools for all types of waterbodies and to provide a baseline for the development of tiered aquatic life uses.

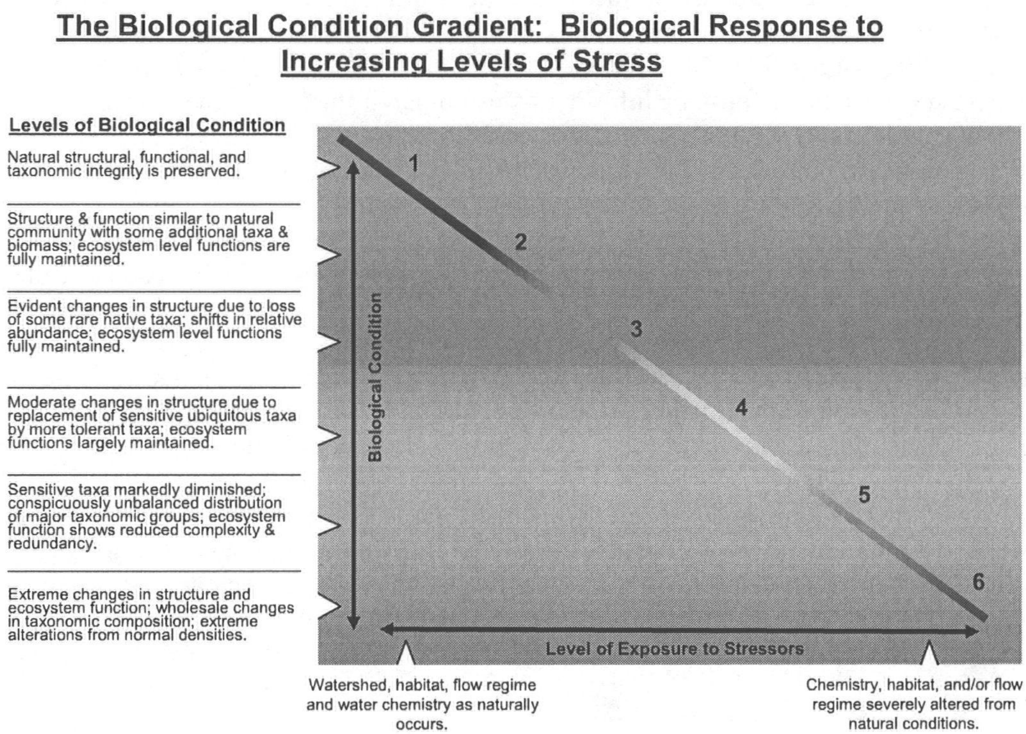

FIGURE 3-30 The Biological Condition Gradient (BCG) and summaries of biological condition along tiers of this gradient. SOURCE: Modified from Davies and Jackson (2006) by EPA.

To date, the BCG has been applied to assemblages including aquatic macroinvertebrates, fish, Unionid mussels, and algae in streams, but it could be applied to any organism group in any type of waterbody. The BCG is derived by applying a suite of ten ecological attributes that allows biological condition to be interpreted independently of assessment method (Table 3-11; Davies and Jackson, 2006). The first five attributes focus on taxa sensitivity, an important component of tools such as multimetric indices (e.g., the Index of Biotic Integrity [IBI], the Invertebrate Community Index [ICI]; see Box 2-3) used in the United States and Europe. Many indicator taxa have been widely studied, and, for groups such as fish, historical data often exist. Most states have established lists of tolerant and intolerant species as part of their use of biological indices (Simon and Lyons, 1995). The relatively large literature on species population and distribution changes in response to stressors and landscape condition offers insight into the mechanisms for population shifts, some of which are summarized in this section.

The first two attributes of the BCG relate to those streams that are closest to natural or pristine, with most taxa "as naturally occur." Attribute 1 and 2 taxa are the most sensitive species that typically disappear with even minor stress. Table 3-12 lists some example attribute 1 taxa for four different regions of the United States. Attribute 3 reflects more ubiquitous, but still sensitive, species that can provide information as human influence on the landscape becomes more obvious, but is not yet severe. Attributes 5 and 6 are taxa that increase in abundance and distribution with increasing stress. The organism condition attribute (7) includes the presence of anomalies (e.g., tumors, lesions, eroded fins, etc.) or the presence of large or long-lived individuals in a population. Most natural streams typically have few or incidental rates of "anomalies" associated with disease and stress. Natural waterbodies typically also have the entire range of life stages present, as would be expected. However, as stress is increased, larger individuals may disappear or emigrate, or reproductive failure may occur. Ecosystem function (attribute 8) is very difficult to measure directly (Davies and Jackson, 2006). However, certain functions can be inferred from structural measures common to various multimetric indices, examples of which are listed in Table 3-13. The last two attributes (9 and 10) may be of particular importance with regard to stormwater and urban impacts. Cumulative impacts are a characteristic of urbanization, and biological organisms typically integrate the effects of many small insults to the landscape. Additionally, most natural systems often have strong "connectance," such that aquatic life often has stages that rely on migrating across multiple types or sizes of waterbodies. Urbanized streams can decrease connectance by creating migration blocks, including vertical barriers at road crossings and small dams (Warren and Pardew, 1998).

TABLE 3-11 Ecological attributes that comprise the basis for the BCG

1. Historically documented, sensitive, long-lived or regionally endemic taxa
2. Sensitive-rare taxa
3. Sensitive-ubiquitous taxa
4. Taxa of intermediate tolerance
5. Tolerant taxa
6. Non-native or introduced taxa
7. Organism condition
8. Ecosystem functions
9. Spatial and temporal extent of detrimental effects
10. Ecosystem connectance

TABLE 3-12 Example of Taxa that *Might Serve* as Attribute 1: "Historically Documented, Sensitive, Long-Lived, Regionally Endemic Taxa for Streams in Four Regions of the United States"

State and taxon	Taxa representative of Attribute I
Maine	
Mollusks	brook floater (*Alasmodonta varicosa*), triangle floater (*Alasmodonta undulata*), yellow lampmussel (*Lampsilis cariosa*),
Fishes	brook stickleback (*Culaea inconstans*), swamp darter (*Etheostoma fusiforme*
Washington	
Fishes	steelhead (*Oncorhynchus mykiss*)
Amphibians	spotted frog (*Rana pretiosa*)
Arizona	
Mollusks	spring snails (*Pyrgulopsis* spp.)
Fishes	Gila trout, (*Oncorhynchus gilae*), Apache trout (*Oncorhynchus apache*), cutthroat trout (endemic strains) (*Oncorhynchus clarki*),
Amphibians	Chihuahua leopard frog (*Rana chiricahuensis*)
Kansas	
Mollusks†	hickorynut (*Obovaria olivaria*), black sandshell (*Ligumia recta*), ponderous campeloma (*Campeloma crassulum*)
Fishes	Arkansas River shiner (*Notropis girardi*), Topeka shiner (*Notropis topeka*), Arkansas darter (*Etheostoma cragini*), Neosho madtom (*Noturus placidus*), flathead chub (*Platygobio gracilisa*)
Other invertebrates	ringed crayfish (*Orconectes neglectus neglectus*), Plains sand-burrowing mayfly (*Homoeoneuria ammophila*)
Amphibians	Plains spadefoot toad (*Spea bombifrons*), Great Plains toad (*Bufo cognatus*), Great Plains narrowmouth toad (*Gastrophryne olivaceae*), Plains leopard frog (*Rana blairi*)

† Although not truly endemic to the central plains, these regionally extirpated mollusks were widely distributed in eastern Kansas prior to the onset of intensive agriculture.

SOURCE: Table 7 from Davies and Jackson (2006). Reprinted, with permission, from Davies and Jackson (2006). Copyright 2006 by Ecological Society of America.

TABLE 3-13 Function Ecological Attributes or Process Rates and Their Structural Indicators

Biotic level and function or process	Structural indicator
Individual level	
Fecundity	Maximum individual size, number of eggs
Growth and metabolism	Length/mass (condition)
Morbidity	Percentage anomalies
Population level	
Growth and fecundity	Density
Mortality	Size- or age-class distribution
Production	Biomass, standing crop, catch per unit effort
Sustainability	Size- or age-class distribution
Migration, reproduction	Presence or absence, density
Community or assemblage level	
Production/respiration ratio, autotrophy vs. heterotrophy	Trophic guilds, indicator species
Primary production	Biomass, ash-free dry mass
Ecosystem level	
Connectivity	Degree of aquatic and riparian fragmentation longitudinally, vertically, and horizontally; presence or absence of diadromous and potadromous species

SOURCE: Table 4 from Davies and Jackson (2006). Reprinted, with permission, from Davies and Jackson (2006). Copyright 2006 by Ecological Society of America.

Construction of a BCG creates a conceptual framework for developing stressor–response gradients for particular urban areas. The initial work done to develop the BCG derived a series of six tiers to describe a gradient of biological condition that is anchored in pristine conditions ("as naturally occurs") and that extends to severely degraded conditions (see Figure 3-30). Exercises done by the national work group to derive such a gradient for macroinvertebrates in wadeable streams showed strong consistency in assigning tiers to datasets using the descriptions of taxa for each attribute along these gradients (Davies and Jackson, 2006). Substantial data already exist to populate many of the attributes of the BCG and to provide mechanistic underpinning for the expected directions of change.

The BCG is not a replacement for assessment tools such as the IBI or multivariate predictive models (e.g., RIVPACS approach), but rather a conceptual overlay for characterizing the anchor point-of-reference conditions and a consistent way to communicate biological condition along gradients of stress. As such, it has strong application to understanding stormwater impacts and to communicating where a goal is located along the gradient of biological condition. While most urban goals may be distant from "pristine" or "natural," the BCG process can dispel misconceptions that alternate urban goals are "dead streams" or unsafe in some manner.

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Factors Limiting Aquatic Assemblages in Urban Waters

A slew of recent investigations have quantified the responses of macroinvertebrates and other biological assemblages to multiple measures of urbanization and to stormwater in particular. One important conclusion of some of this work is that declines in the highest biological condition start with low levels of anthropogenic change (e.g., 5 to 25 percent impervious surface); higher levels of urbanization severely alter aquatic conditions (Horner et al., 2003). This has important consequences for protecting sites with the highest biological integrity, as they may be among the most vulnerable. The non-threshold nature of this aquatic response and the typical wedge-shaped response to multiple stressors by aquatic assemblages are discussed in Box 3-9.

The sections that follow review the evidence underlying biological responses to each of the major categories of stressors: chemical, hydrologic, physical habitat, biological, and energy-related factors. As will be evident in some of the examples, the stressors themselves can interact (e.g., flow can influence habitat, habitat can influence energy processing, etc.), which increases the complexity of understanding how stormwater affects aquatic ecosystems.

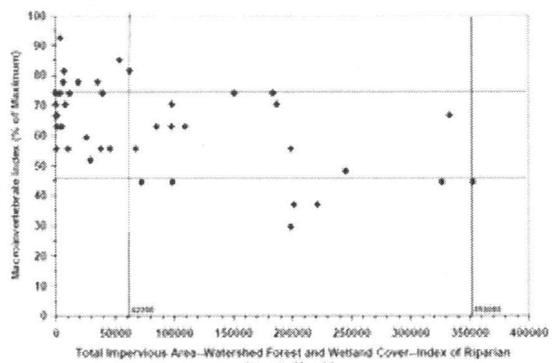
Biological Responses to Toxic Pollutants

The chemical constituents of natural streams vary widely with climatic region, stream size, soil types, and geological setting. Most small natural streams, outside of unique areas wth naturally occurring toxicants, have very low levels of chemicals considered to be toxicants and have relatively low levels of dissolved and particulate materials in general. This applies to chemicals in the water column and in sediments. Increasing amounts of impervious surface in the watershed typically increase the concentrations of many chemical parameters in runoff derived from urban surfaces (e.g., Porcella and Sorenson, 1980; Sprague et al., 2007).

BOX 3-9
Non-threshold Nature of the Decline of Biological Assemblages Along Urban Stressor Gradients

Several recent surveys have demonstrated that biological assemblages begin to decline in condition with even low levels of urban disturbance as measured by various gradients of urbanization (e.g., May, 1996; Horner et al., 1997; May et al., 1997; Horner et al., 2003; Moore and Palmer, 2005; Barbour et al., 2008). This box summarizes the work of Horner et al. (2003) in small streams in three regions: Montgomery County, Maryland; Austin, Texas; and the Puget Sound area of Washington. Geographic Information System (GIS) analyses using information such as land use, total impervious area, and riparian land use were used to develop multi-metric Watershed Condition Indices (WCIs) for each region. These in turn were related to fish and macroinvertebrate indices, e.g., benthic IBIs, (B-IBI, all three regions), a fish IBI (F-IBI for Maryland) and an index that was the ratio of the sensitive coho salmon to the more tolerant cutthroat trout in collections for the Puget Sound lowland area.

In each of these areas, no or extremely low urban development, substantial forest cover, and minimal disturbance of riparian zones characterized sites with the highest biological scores, but these conditions did not guarantee high scores because other impacts could limit biology even with these "natural" characteristics. In all three regions, high urbanization and loss of natural cover always led to biological degradation (Figures 3-31 and 3-32). The results of this study were similar to other recent studies such as Barbour et al. (2008) that identify a "wedge-shaped" relationship or a "polygonal" relationship (Carter and Fend, 2005) between urban gradients and biological condition. These types of relationships have also been termed "factor-ceiling" relationships (Thomson et al., 1996). The outer surface of these wedges or polygons reflects where the urban gradients limit biological assemblages, such that points below this surface typically represent sites affected by other stressors (e.g., combined sewer overflows, discharges, etc.). In all of these studies it is easier to predict loss of biological conditions as the urban gradients (e.g., WCI) worsen than it is to ensure high biological integrity at low proportions of urban stress (because some other stressor may still limit aquatic condition).



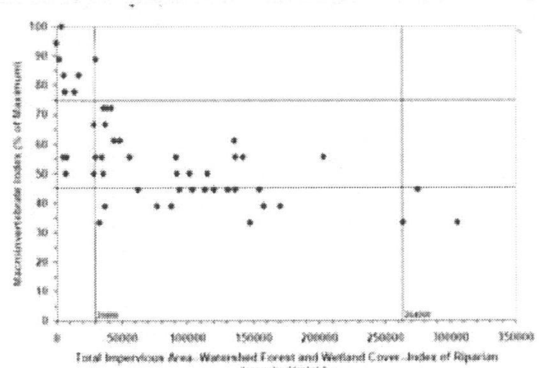


FIGURE 3-31 Plots of a measure of urbanization (TIA + Wetland & Forest Cover + IRI) versus B-IBIs for Austin, Texas (left), and Montgomery County, Maryland (right). SOURCE: Horner et al. (2003).

BOX 3-9 Continued

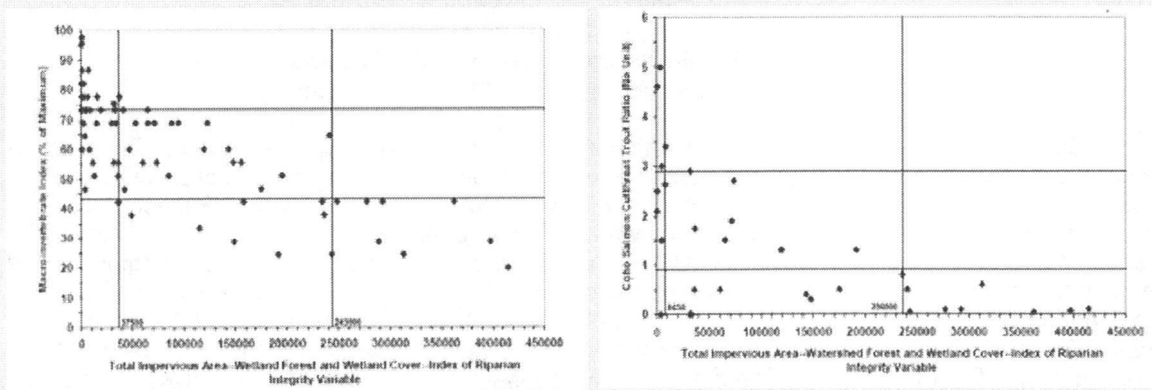


FIGURE 3-32 Plots of a measure of urbanization (TIA + Wetland & Forest Cover + IRI) versus B-IBIs for Puget Sound (left) and versus the ratio of coho salmon to cutthroat trout for Puget Sound (right). SOURCE: Horner et al. (2003).

Horner et al. (2003) also focused on whether structural SCMs could moderate the effects of urbanization on biological assemblages. They made detailed observations of two subbasins in the Puget Sound lowland area, one with a greater degree of stormwater management than the other (although neither had what would be considered comprehensive stormwater management with a focus on water quality issues). As shown in Figure 3-33, at the highest levels of urbanization (triangles), the subbasin with the more extensive use of structural SCMs did have better biological conditions. There was less evidence of biological benefit in the watershed that used SCMs but it had only moderate urbanization and more natural land cover (squares and diamonds). There were no circumstances where high biological condition was observed along with the use of SCMs because high biological condition only occurred where little human alteration was present, and thus SCMs were not used.

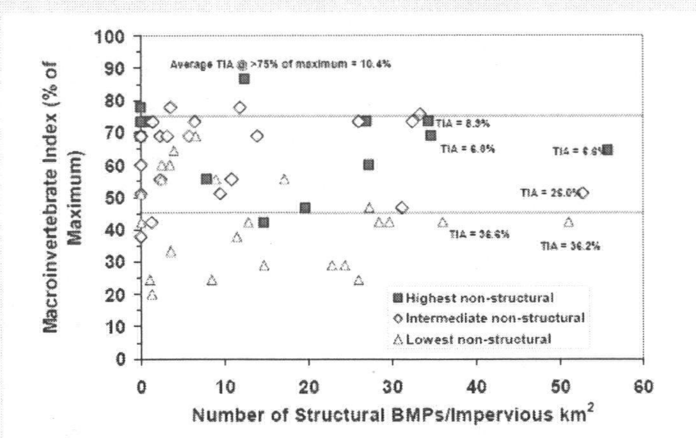


FIGURE 3-33 Macroinvertebrate community index versus structural SCM density with the highest, intermediate, and lowest one-third of natural watershed and riparian cover. The upper and lower horizontal lines represent indices considered to define relatively high and low levels of biological integrity, respectively. SOURCE: Horner et al. (2003).

Stormwater concentrations of these pollutants can be variable and sometimes extreme or "toxic" depending on the timing of flows (e.g., first flush), although concentrations at base flows may not routinely exceed water quality benchmarks (Sprague et al., 2007). Historical deposition of toxics in sediments can also be responsible for extremely high pollutant concentrations within waterbodies, even though the stormwater discharges may no longer be active. These situations have been termed "legacy pollution" and are most commonly associated with urban centers that have a history of industrial production.

Natural constituents such as dissolved materials (e.g., chlorides), particulate material (e.g., fine sediments), nutrients (e.g., phosphorus and nitrogen compounds), as well as a myriad of man-made parameters such as heavy metals and organic chemicals (e.g., hydrocarbons, pesticides and herbicides) have been documented to be increased and at times pervasive in stormwater (Heany and Huber, 1984; Paul and Meyer, 2001; Roy et al., 2003; Gilliom et al., 2006) although specific patterns of concentrations can vary with region and ecological setting (Sprague et al., 2007). Water chemistry impacts can also arise from a complex array of permitted discharges, storm sewer discharges, and combined sewer overflows that are treated to certain limits but at times fail to remove all constituents from flows, especially when associated with storm events (Paul and Meyer, 2001).

Streams in urban settings can have increases in toxicant levels compared to background concentrations. In many instances these cases have been associated with loss of aquatic species and impairment of aquatic life goals (EPA, 2002a), which are usually explained in terms of typical lethal responses. The complexity of urban systems with regard to pathways, magnitude, duration, and timing of toxicity as well as possible synergistic or antagonistic effects of mixtures of pollutants argues for a broad approach to characterizing effects including not only toxicity testing, but also novel approaches and direct monitoring of biological assemblages (Burton et al., 1999). What is problematic from a traditional management perspective is that aquatic communities may decline before exceedances of water quality criteria are evident (May et al., 1997; Horner et al., 2003).

The first three BCG attributes focus on populations of species of high to very high sensitivity, most of which are uncommon or absent in waters with any substantial level of urbanization. Multi-metric indices such as IBI, which reflect loss of these species, decline at least linearly with increasing urbanization (e.g., Miltner et al., 2004; Meador et al., 2005; Walters et al., 2005). Although toxicity to compounds varies with species, many species of federal and state endangered and threatened aquatic species are more sensitive than "commonly" used test species (Dwyer et al., 2005), such that the loss of aquatic species when toxicant levels exceed criteria are readily explained.

The mechanisms of species population declines in response to chemical contaminants are likely complex and not just limited to direct lethality of the pollutant. Indeed, initial chemical changes may have no "toxic" effects, but rather could change competitive and trophic dynamics by changing primary production and energy dynamics in streams. For example, exposures to aromatic and chlorinated organic compounds from sediments derived from urban areas have been found to increase the susceptibility of salmonids to the bacterial pathogen *Vibrio anguillarum* (Arkoosh et al., 2001). Recent work has found that salmonids show substantial behavioral changes from olfactory degradation related to copper at concentrations as low as 2 μg/L, well below copper water quality criteria and above levels measured in most stormwater-affected streams (Hecht et al., 2007; Sandahl et al., 2007). Salmonid and other fish depend extensively on olfactory cues for feeding, emigration, responding to prey and predators, social and spawning interactions, and other behaviors, such that loss or diminution of such cues may have population-level effects on these species (Sandahl et al., 2007). Copper has been shown to cause olfactory effects on other species (Beyers et al., 2001) and to impair the sensory ability of the fish lateral line (Hernandez et al., 2006), which is nearly ubiquitous in fishes and important for most freshwater species in feeding, schooling, spawning, and other behaviors.

Whole effluent toxicity testing or sediment toxicity testing may misclassify the effects of runoff and effluents in urban settings (Burton et al., 1999). Short-term toxicity tests of stormwater often result in no identified toxicity. However, longer studies (e.g., 30 days) have shown increasing toxicity with time (Masterson and Bannerman, 1994; Ramcheck and Crunkilton, 1995). This suggests that the mechanism of toxicity could be through an ingestion pathway, for example, rather than gill uptake. Metals are often in high concentrations where fine sediments accumulate, and their legacy can extend past the time period of active discharge. Metal concentrations in urban stream sediments have been associated with high rates of fish and invertebrate anomalies such as tumors, lesions, and deformities (Burton, 1992; Ingersoll et al., 1997; Smith et al., 2003).

Biological Responses to Non-Toxicant Chemicals

Non-toxic chemical compounds that occur in stormwater such as nutrients, dissolved oxygen (DO), pH, and dissolved solids as well as physical factors such as temperature can have impacts on aquatic life. The effects of some of these compounds (e.g., DO, pH) have been well documented from other impacts (e.g., wastewater, mining), such that nearly all states have developed water quality criteria for these parameters. For example, nutrient enrichment in stormwater runoff has been associated with declines of biological condition in streams (Miltner and Rankin, 1998). Chloride, sulfate, and other dissolved ions that are often elevated in urban areas can have effects on osmoregulation of aquatic organisms and have been associated with loss of species sensitive to dissolved materials such as mayflies (Kennedy et al., 2004). The concentrations of these compounds can vary regionally (Sprague et al., 2007) and with the degree of urbanization.

Water quality criteria for temperature were spurred by the need for thermal permits for industrial and power plant cooling water discharges. There is a very large literature on the importance of water temperature to aquatic organisms; preference, avoidance, and lethal temperature ranges have been derived for many aquatic species (e.g., Brungs and Jones, 1977; Coutant, 1977; Eaton et al., 1995). In addition, temperature is one of the key classification strata for aquatic life, in that streams are routinely classified as cold water, cool water, or warm water based on the geographic and natural settings of waters. The removal of catchment and riparian vegetation and the general increase in surface runoff from impervious, man-made, and heat-capturing surfaces has been associated with increasing water temperatures in urban waterbodies (Wang and Kanehl, 2003; Nelson and Palmer, 2007). A number of researchers have created models to predict in-stream temperatures based on urban characteristics (Krause et al., 2004; Herb et al., 2008).

Hydrologic Influences on Aquatic Life

The importance of "natural" flow regimes on aquatic life has been well documented (Poff et al., 1997; Richter et al., 1997a, 2003). As watersheds urbanize, flow regimes change from little runoff to over 40 to 90 percent of the rainfall becoming surface runoff (Roesner and Bledsoe, 2003). Flow regimes in urban streams typically are very "flashy," with higher and more frequent peak events, compared to undisturbed systems (Poff et al., 1997; Baker et al., 2004) and well as reduced base flows and more frequent desiccation (Bernhardt and Palmer, 2007). Richter et al. (1996) proposed a series of indicators that could be used to measure hydrologic disturbance, many of which have been used in the recent studies identifying the hydrologic effects of stormwater on aquatic biota (Barbour et al., 2008). Pomeroy et al. (2008) did an extensive review of which flow characteristics appear to have the greatest influence on biological metrics and biological integrity. No single measure of flow was found to be significant in all studies; however, important attributes included flow variability and flashiness, flood frequency, flow volume, flow variability, flow timing, and flow duration.

There are a number of mechanisms that may be responsible for the influence of flow characteristics on aquatic assemblages. Aquatic species vary dramatically in their swimming performance and behaviors, and species are generally adapted to undisturbed flow regimes in an area. Many low- to moderate-gradient small streams in the United States, for example, have strong connections with their flood-prone areas and often possess habitat features that insulate poor swimming species from episodic natural high flows. Undercut banks, rootwads, oxbows, and backwater habitats all can act as refugia from high flows. Some aquatic species are more or less mobile within the sediments, like certain macroinvertebrates (meiofauna or hyporheos) and fish species such as sculpins and madtoms. Secondary impacts from hydrologic changes such as bank erosion and aggradation of fines can render substrates embedded and prohibit organisms, particularly the meiofauna, from moving vertically within the bottom substrates (Schmid-Araya, 2000). Substrate fining has been documented to occur with increasing urbanization, especially in the early stages of development, which can embed spawning habitats and eliminate or reduce spawning success of fish such as salmonids and minnows (Waters, 1995).

Flood flows can cause mortality in the absence of urbanization. For example, flood flows in streams under natural conditions have been documented as a cause of substantial mortality in young or larval fish such as smallmouth bass (Funk and Fleener, 1974; Lorantas and Kristine, 2004). Increased flashiness from urbanization is likely to exacerbate this effect. Thus, increases in the frequency of peak flows during spring will increase the probability of spawning failure, such that sensitive species may eventually be locally extirpated. In urban areas, culverts and other flow obstructions can create conditions that may preclude re-colonization of upstream reaches because weak-swimming fishes cannot move past flow constrictions or leap past vertical drops caused by artificial structures.

Hydrologic simplification and stream straightening that occur in urban streams, often as a result of increased peak flows or as a local management response, typically remove habitat used as temporary refuges from high flows, such as backwater areas, undercut banks, and rootwads. There is a large literature relating populations of fish and macroinvertebrates to various habitat features of streams, rivers, and wetlands. The first two attributes of the BCG identify taxa that are historically documented, sensitive, long-lived, or regionally endemic taxa or sensitive-rare taxa. Many of these taxa are endangered because of large-scale changes in flow-influenced habitats; that is, threats of extinction often center on habitat degradation that influence spawning, feeding, or other aspects of a species life history (Rieman et al., 1993). In contrast, many of the fish and macroinvertebrate taxa that compose regional lists of tolerant taxa are tolerant to habitat changes related to flow disturbance as well as chemical parameters. Understanding the life history attributes of certain species and how they may change with multiple stressors (Power, 1997) is an important tool for understanding complex responses of aquatic ecosystems to urban stressors.

Geomorphic and Habitat Influences on Aquatic Life

In natural waters, geomorphic factors and climate, modified by vegetation and land use, constrain the types of physical habitat features likely to occur in streams (Webster and D'Angelo, 1997). For example, very-low-gradient streams may have few riffles and be dominated by woody debris and bank cover, whereas higher gradient waters may have more habitat types formed by rapidly flowing waters (riffles, runs). Aquatic life in streams is influenced directly by the habitat features that are present, such as substrate types, in-stream structures, bank structure, and flow types (e.g., deep-fast vs. shallow-slow).

As discussed previously, human alteration of landscapes, encroachment on riparian areas, and direct channel modifications (e.g., channelization) that acompany urbanization have often resulted in unstable channels, with negative consequences for aquatic habitat. As urbanization has increased, channel density has declined because streams have been piped, dewatered, and straightened (Meyer and Wallace, 2001; Paul and Meyer, 2001). Changes in the magnitude, relative proportions, and timing of sediment and water delivery have resulted in loss of aquatic life and habitat via a wide range of mechanisms, including changes in channel bed materials, increased suspended sediment loads, loss of riparian habitat due to bank erosion, and changes in the variability of flow and sediment transport characteristics relative to aquatic life cycles (Roesner and Bledsoe, 2003). There are still significant gaps in knowledge about how stormwater stressors can affect stream habitat, especially as one moves from the reach scale to the watershed scale. Understanding the stage and trajectory of channel evolution is critical to understanding channel recovery and expected habitat conditions or in choosing effective restoration options (Simon et al., 2007).

Across much of the United States, stream habitats have been altered to the imperilment of aquatic species (Williams et al., 1989; Richter et al., 1997b; Strayer et al., 2004). A study of rapidly urbanizing streams in central Ohio identified the loss of highly and moderately sensitive species as a key factor the decline in the IBI in these streams (Miltner et al., 2004). These streams had historical fish collections when they were primarily influenced by agricultural land use; sampling after the onset of suburban development documented the loss of many of these species attributable to land-use changes and habitat degradation along these urban streams. Along the BCGs that have been developed for streams, most of the species in attributes 1–3 are specialists requiring very specific habitats for spawning, feeding, and refuge. Habitat alteration, either direct or indirect, creates harsh environments that tend to favor tolerant taxa, which would otherwise be in low abundance. Often these tolerant species are characterized by high reproductive potential, generalist feeding behaviors, tolerance to chemical stressors such as low DO, and pioneering strategies that allow rapid recolonization following acute stressful events.

Altered Energy Pathways in Urban Streams

The pathways of energy flow in streams are an important determinant of aquatic species distributions. In most natural temperate streams, headwaters transform and export energy from stream side vegetation and adjacent land uses into aquatic biomass. The types, amount, and timing of delivery of water, organic material, and debris have important consequences for conditions downstream (Dolloff and Webster, 2000). The energy-transforming aspect of stream ecosystems is difficult to capture directly, so most measures are surrogates, such as the trophic characteristics of assemblages and chemical and physical characteristics consistent with natural energy processes.

An increasingly urban landscape can have a complex array of effects on energy dynamics in streams (Allan, 2004). Loss of riparian areas and changes in riparian vegetation can reduce the supply and quality of coarse organic matter that forms the base of aquatic food webs in most small streams. The reduction in the amount of organic matter with riparian loss is obvious; however, changing species of vegetation (e.g., invasion or planting of exotic species) can affect the quality of organic matter and influence higher trophic levels because, for example, exotic species may have different nutrient values (e.g., C/N ratios, trace chemicals) or process nutrients at a different rate (Royer et al., 1999). Furthermore, native invertebrate taxa may not be adapted to utilize the exotic material (Miller and Boulton, 2005). For example, changes in leaf species in a stream may alter the macroinvertebrate community by favoring species that feed on fast-decaying versus slow-decaying leaves (Smock and MacGregor, 1988; Cummins et al., 1989; Gregory et al., 1991).

Other recent work is examining ways that changes in geomorphology with increasing urbanization can influence trophic structure in streams (Doyle, 2006). Groffman et al. (2005) examined nitrogen processing in stream geomorphic structures such as bars, riffles, and debris dams in suburban and forested areas. Although suburban areas had high rates of production in organic-rich debris dams and gravel bars, higher storm flow effects in urban streams may make these features less stable and able to be maintained (Groffman et al., 2005). Changes in habitat and riparian vegetation may greatly alter trophic patterns of energy transport. For example, local nutrient enrichments combined with reduced riparian vegetation can result in nuisance algal growths in waterbodies that are evidence of simpler energy pathways. Corresponding effects are further water chemistry changes from algal decomposition (e.g.., low DO) or very high algal activity (e.g., high pH) (Ehlinger et al., 2004).

The complexity of energy flow through simple ecosystems is illustrated in Figure 3-34, a "simplified" food web of a headwater stream published by Meyer (1994). The forms in which nutrients are delivered to streams may be more important than actual concentrations as well as the availability of carbon sources essential for nutrient transformation. The nutrient components that form the base of the food web in Figure 3-34 are the FPOM and CPOM boxes. In many natural streams, woody and leafy debris are the most common form of nutrient input, and changes to urban landscapes often change this to dissolved and finer forms. Urbanization can also reduce the retention of organic debris of streams (Groffman et al., 2005) and the timing of nutrient delivery. Timing can be of crucial importance since species spawning and growth periods may be specifically timed to take advantage of available nutrients.

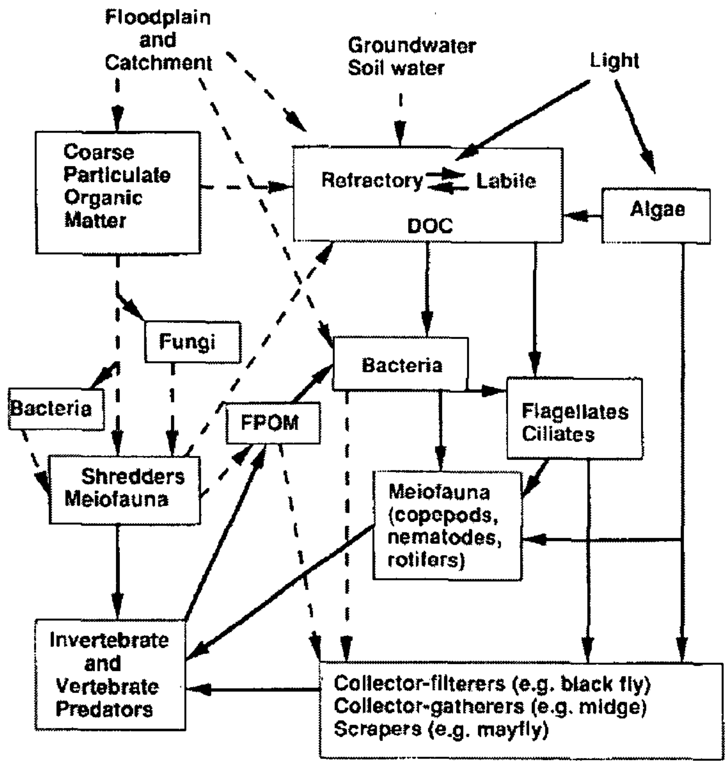


FIGURE 3-34 Simplified diagram of a lotic food web showing sources and major pathways of organic carbon. Dotted lines indicate flows that are a part of the microbial loop in flowing water but not in planktonic systems. SOURCE: Reprinted, with permission, from Meyer (1994). Copyright 1994 by Springer.

As important as energy and nutrient dynamics are to stream function, many of the stream characteristics that determine effective energy flow are not typically considered when characterizing stormwater impacts. The best chance for considering these variables and maximizing ecosystem function is through integrated, biologically based monitoring programs that include urban areas (Barbour et al., 2008) and stressor identification procedures (EPA, 2000) to isolate likely causes of impact and to inform the choices of SCMs.

Biological Interactions in Urban Streams

Streams in urbanized environments often are characterized by fewer native and more alien species than natural streams (DeVivo, 1996; Meador et al., 2005). The influence of exotic species is not always predictable and may be most severe in lentic environments (e.g., wetlands, estuaries) and in riparian zones where various exotic aquatic plants can greatly alter natural systems in both structure and function (Hood and Naiman, 2000). Riley et al. (2005) found that the presence of alien aquatic amphibians was positively related to degree of urbanization, as was the absence of certain native amphibian species. In a review of possible reasons for this observation, he suggested that altered flow regimes were responsible. In the arid California streams they studied, flow became more constant with urbanization (i.e., natural streams were generally ephemeral), which allowed invasion by exotic species that can prey on, compete with, or hybridize with native species (Riley et al., 2005). The alteration of stream habitat that accompanies urbanization can also lead to predation by domestic cats and dogs or collection by humans, especially where species (e.g., California newts) are large and conspicuous (Riley et al., 2005).

The effects of specific exotic species on aquatic systems has been observed to vary geographically, although recent work has found correlations between total invasion rate and the number of high-impact exotic species (Ricciardi and Kipp, 2008). This suggests that overall efforts to reduce the importation or spread of all alien species should be helpful.

The Role of Biological Monitoring

The preceding sections illustrate the importance of biological data to understanding the complexities associated with urban and stormwater impacts to waterbodies. Although categories of urban stressors have been discussed individually, these stressors routinely, if not universally, co-occur in urban waterbodies. Their cumulative impacts are best measured with biological tools because the biota integrate the influence of all of these stressors.

Many programmatic aspects of the CWA arose as a response to rather obvious impacts of chemical pollutants that were occurring in surface waters during this time. The initial focus of water quality standards was on developing chemical criteria that could serve as engineering endpoints for waste treatment systems (e.g., NPDES permits). Rather general aquatic life goals for streams and rivers that were suitable for the initial focus of the CWA are now considered insufficient to deal with the complex suite of stressors limiting aquatic systems. To that end, refined aquatic life goals and improved biological monitoring are essential for effective water quality management, including stormwater issues (NRC, 2001). Practical biological and physical monitoring tools have even been developed for very small headwater streams (Ohio EPA, 2002; Fritz et al., 2006), which are particularly affected by stormwater because of their prevalence (greater than 95 percent of channels), their relatively high surface-to-volume ratio, their role in nutrient and material processing, and their vulnerability to direct modification such as channelization and piping (Meyer and Wallace, 2001).

Surrogate indicators of stormwater impacts to aquatic life (such as TSS concentrations) have been widely used because direct biological measures were poorly developed and these surrogates were assumed to be important to pollutant delivery to urban streams. However, biological assessment has rapidly advanced in many states and can be readily applied or if needed modified to be sensitive to stormwater stressors (Barbour et al., 2008). As Karr and Chu (1999) warned, the management of complex systems requires measures that integrate multiple factors. Stormwater permitting is no different, and care must be taken to ensure that permitting and regulatory actions retain ecological relevance. Surrogate measures have an essential role in the assessment of individual SCMs; however, this needs to be kept in context with the entire suite of stressors likely to be important to the aquatic life goals in streams.

Stormwater management programs should not necessarily bear the burden of biological monitoring; rather, well-conceived biological monitoring should be the prevue of state and local government agencies (as discussed more extensively in Chapter 6). Refined aquatic life goals developed for all waters, including urban waters, measured with appropriate biological measures, should be the final endpoint for management. The collection of biological data needs to be closely integrated across multiple disciplines in order to be effective. Pomeroy et al. (2008)

describe a multidisciplinary approach to study the effects of stormwater in urban settings, and Scholz and Booth (2001) also propose a monitoring approach for urban watersheds. Such efforts are not necessarily easy, and many institutions find pitfalls when trying to integrate scientific information across disciplines (Benda et al., 2002).

EPA water programs, such as the Total Maximum Daily Load (TMDL) program, have been criticized for having too narrow a focus on a limited number of traditional pollutants to the exclusion of important stressors such as hydrology, habitat alteration, and invasive taxa (Karr and Yoder, 2004)—all serious problems associated with stormwater and urbanization. The science has advanced significantly over the past decade so that biological assessment should be an essential tool for identifying stormwater impacts and informing the choice of SCMs in a region or watershed. Although biological responses to stressors in the ambient environment are by their nature correlative exercises, ecological epidemiology principles or "stressor identification" methods can identify likely causative agents of impairment with relatively high certainty in many instances (Suter, 1993, 2006; EPA, 2000). Coupled with other ambient and source monitoring information, biological information can form the basis for an effective stormwater program. As an example, Box 3-10 introduces the Impervious Cover Model (ICM), which was developed using correlative information on the association between impervious cover and biological metrics. The crux of the ICM is that stormwater management is tailored along a readily measureable gradient (impervious cover) that integrates multiple individual stressor categories that would otherwise be overlooked in the traditional pollutant-based approach to stormwater management. Even the form of the ICM (as conceptualized in Figure 3-37) matches that outlined for the BCG (Figure 3-30). Use of the ICM to improve the MS4 stormwater program is discussed in Chapter 6.

BOX 3-10
The Impervious Cover Model: An Emerging Framework for Urban Stormwater Management

The Impervious Cover Model (ICM) is a management tool that is useful for diagnosing the severity of future stream problems in a subwatershed. The ICM defines four categories of urban streams based on how much impervious cover exists in their subwatershed: *high-quality streams*, *impacted streams*, *non-supporting streams*, and *urban drainage*. The ICM is then used to develop specific quantitative or narrative predictions for stream indicators within each stream category (see Figure 3-35). These predictions define the severity of current stream impacts and the prospects for their future restoration. Predictions are made for five kinds of urban stream impacts: changes in stream hydrology, alteration of the stream corridor, stream habitat degradation, declining water quality, and loss of aquatic diversity.

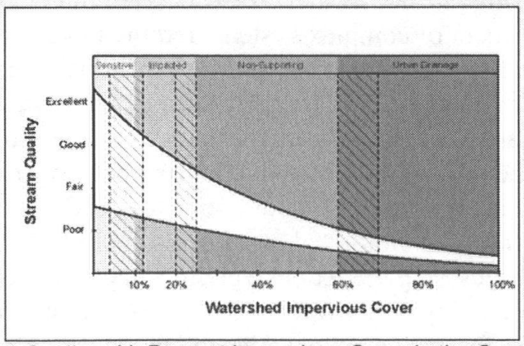


FIGURE 3-35 Changes in Stream Quality with Percent Impervious Cover in the Contributing Watershed. SOURCE: Chesapeake Stormwater Network (2008). Reprinted, with permission, from Schueler (2008). Copyright 2008 by T. Schueler.