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Hydrologic Changes in Urban Streams and Their Ecological Significance

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Abstract.—Urban development modifies the production and delivery of runoff to streams and the resulting rate, volume, and timing of streamflow. Given that streamflow demonstrably influences the structure and composition of lotic communities, we have identified four hydrologic changes resulting from urban development that are potentially significant to stream ecosystems: increased frequency of high flows, redistribution of water from base flow to storm flow, increased daily variation in streamflow, and reduction in low flow. Previous investigations of streamflow patterns and biological assemblages provide a scale of ecological significance for each type of streamflow pattern. The scales establish the magnitude of changes in streamflow patterns that could be expected to produce biological responses in streams. Long-term streamflow records from eight streams in urbanizing areas of the United States and five additional reference streams, where land use has been relatively stable, were analyzed to assess if streamflow patterns were modified by urban development to an extent that a biological response could be expected and whether climate patterns could account for equivalent hydrologic variation in the reference streams. Changes in each type of streamflow pattern were evident in some but not all of the urban streams and were nearly absent in the reference streams. Given these results, hydrologic changes are likely significant to urban stream ecosystems, but the significance depends on the stream's physiographic context and spatial and temporal patterns of urban development. In urban streams with substantially altered hydrology, short-term goals for urban stream rehabilitation may be limited because of the difficulty and expense of restoring hydrologic processes in an urban landscape. The ecological benefits of improving physical habitat and water quality may be tempered by persistent effects of altered streamflow. In the end, the hydrologic effects of urban development must be addressed for restoration of urban streams.

Introduction

Streamflow is the dominant physical process distinguishing rivers and streams from other ecosystems. The structure and composition of lotic communities in fluvial ecosystems depend on source, timing, and rate of streamflow as they regulate both habitat conditions and disturbance regimes. Because of the important roles of streamflow in fluvial ecosystems, hydrologic changes in urban streams pose plausible and potentially significant ecological risks that warrant evaluation, along with other recognized changes (e.g., water chemistry, physical habitat, riparian conditions). Although previous investigations have associated degradation in the biological condition of streams with urban development (Klein 1979; Wang et al. 2000; Morley and Karr 2002; Booth et al. 2004), streamflow patterns per se are seldom implicated as a primary cause of urban stream degradation. Failure to address the significant hydrologic changes in urban areas may limit the success of restoration efforts in rivers and streams.

"Urbanization" is not a single condition; instead, it is a collection of actions that lead to recognizable landscape forms and, in turn, to changes in stream conditions. No single change defines urbanization,

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but the cumulative effect of human activities in urban basins profoundly influences streams and their biota. Karr and Yoder (2004) provide a conceptual framework for assessing biological degradation in which the human actions that constitute urbanization modify many aspects of stream ecosystems, which elicit biological responses (Figure 1). Among the many aspects of stream ecosystems modified by urbanization, changes in hydrologic regimes (i.e., the timing and rate of streamflow) and potential biological responses are the focus of this paper. The influence of hydrologic modification, however, is likely to depend on a broader ecological context, and so the importance of hydrologic modification is likely to vary from stream to stream and region to region.

We focus on hydrology as a primary link between urban development and biological responses in urban streams because of the well-established influence of streamflow on the structure, composition, and productivity of lotic communities. In this paper, we briefly review the literature on relations between streamflow patterns and lotic communities to identify the types of streamflow patterns that have demonstrable effects on stream biota. We then extend the results of two investigations, Poff and Allan (1995) and Clausen and Biggs (1997), that associated variation of biological assemblages with variation in a variety of streamflow statistics. Our extended analyses identify the magnitude of variation in four streamflow patterns (high flow frequency, flow distribution, daily variation, and low flow magnitude) that was associated with biological differences within their respective groups of streams. The magnitude of hydrologic variation associated with biological differences establishes the scale of ecological significance for assessing changes in each streamflow pattern. We then analyze long-term streamflow records from eight urban streams and five reference streams in the United States, where land use has been relatively stable, to identify streamflow patterns that changed in association with urban development. Finally, we discuss our results in the context of managing urban streams.

Hydrologic Effects of Urban Development

Hydrologic effects of urbanization can be characterized as the redistribution of water once intercepted by

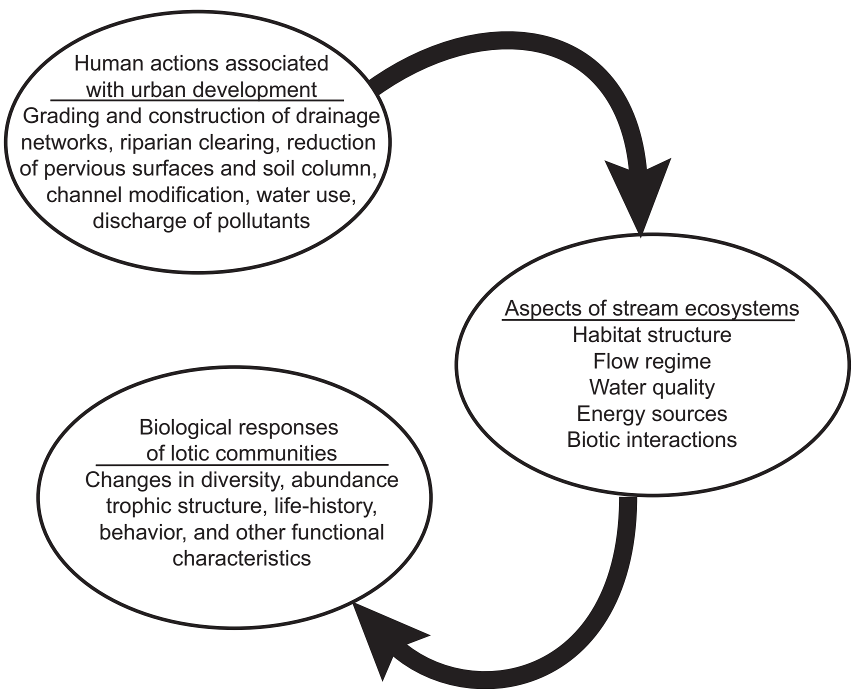

FIGURE 1. Urban development represents many distinct human actions that affect each of five major aspects of stream ecosystems. Changes in the ecosystem, in turn, elicit biological responses (modified from Karr and Yoder 2004).

vegetation or stored in the soil column, from which it drained slowly into streams or aquifers or was taken up by vegetation and transpired. Following development, this water runs off impervious surfaces or saturates thin soils and then runs off as overland flow. Drainage networks collect runoff and allow it to flow quickly as open-channel flow to receiving water bodies, greatly reducing the length of subsurface or overland flow paths. Aquifer recharge and evapotranspiration may be reduced. The hydrologic effects of urban development were evident in a comparison of runoff from two headwater catchments in western Washington (Burges et al. 1998). Novelty is a 37-ha, forested catchment. Klahanie is a 17-ha catchment with residential land use. Although much of the landscape in Klahanie is "pervious" lawn, the thin soils are quickly saturated during storms producing runoff patterns distinct from Novelty, including higher peak unit-area discharge, higher runoff volume, and more frequent peaks (Figure 2).

Changes in runoff patterns resulting from urban development manifest in streams as an increase in storm flow rates and volumes and a corresponding reduction in recessional flow and base flow rates and volumes (Leopold 1968; Konrad 2003a). There are also higher-order hydrologic effects of urban development: streamflow rises more rapidly during storms and recedes more rapidly after storms, which is typically described as "flashy" streamflow. Storm flow is more frequent and the seasonal period for storm flow may lengthen as storms produce higher streamflow in urban areas, even when antecedent conditions are dry. Because runoff is redistributed from base flow periods to storm flow periods, the total water balance and, thus, central measures of streamflow distributions such as annual mean streamflow may not change in response to urban development (Konrad and Booth 2002). The water balance for a stream basin may change when changes in vegetation alter interception of precipitation and evapotranspiration, surface or groundwater is used consumptively, or water is imported into a basin for irrigation.

Base flow changes less consistently in response to urban development. Annual base flow volume for two streams draining sewered areas of Long Island, New York was reduced from more than 60% to approximately 20% during urban development (Simmons and Reynolds 1982). Unit-area base flow during the wet season in western Washington streams clearly decreases with increasing road density, in contrast to unit-area base flow during the dry season (Figure 3). Minimum annual 7-d streamflow, however, did not change consistently in response to urban development in western Washington (Konrad and Booth 2002). Thus, urban development appears to reduce shallow subsurface flow that supports wet-season base flow but has a less evident effect on deeper groundwater recharge that supports summertime discharges.

Low flow in urban streams can increase as a result

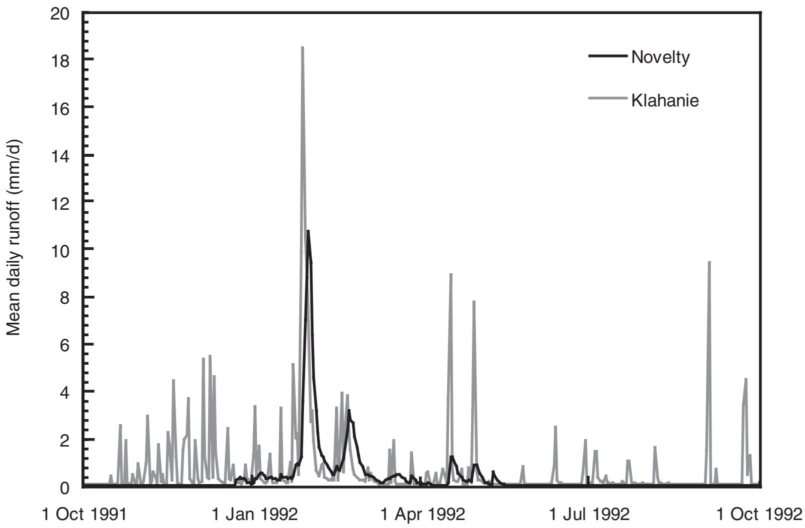

FIGURE 2. Runoff during water year 1992 for two headwater catchments in western Washington. At the time of data collection, Novelty was a fully forested 37-ha catchment; Klahanie was a fully built-out 17-ha residential catchment.

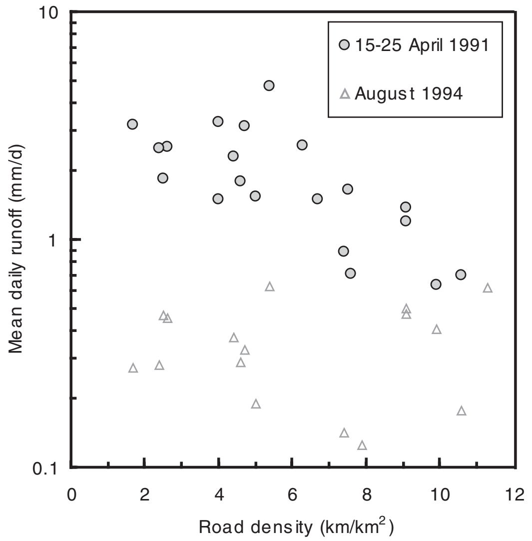

FIGURE 3. A comparison of mean unit-area base flow in the wet season and dry season for 21 streams in western Washington.

of imported water used for landscape irrigation that subsequently drains through shallow soil layers and aquifers (Harris and Rantz 1964) or used in residences with on-site septic systems. In contrast, decreases in low flow occur from increased surface drainage (storm sewers and ditches), surface water diversions, and pumping shallow groundwater. For example, annual minimum streamflow in Big Soos Creek in Washington have declined significantly in response to residential development and water use in the basin over the past few decades, while no such trends are evident in nearby rural streams (Konrad and Booth 2002).

In headwater ephemeral streams, periods of continuous flow are likely to be briefer with frequent intermittency (Figure 2). For example, during the period from water year (WY) 1991–1993, there was an average of 24 periods of flow intermittency per year in the residential Klahanie catchment, including periods of no flow during spring, compared to five times per year for the forested Novelty catchment. This difference is not reflected by the percentage of days with flow in these streams, which was 62% of the 3-year period in Klahanie compared to 54% in Novelty.

Given the magnitude of hydrologic changes resulting from urban development, urban streamflow patterns are likely to affect the biological conditions of streams, although these patterns frequently have been quantified in ways that are not directly relevant to their ecological effects but, instead, represent only their social impacts. Few studies definitively make the link between hydrologic alteration and biological responses in urban streams, in part because urban development affects nearly all aspects of fluvial ecosystems. Orser and Shure (1972) documented lower population densities of dusky salamander *Desmognathus fuscus fuscus* with increasing levels of urban development for five streams near Atlanta, Georgia. They indicated that floods had likely transported individuals downstream, reducing salamander populations and creating unstable age structures. In 27 streams in Maryland, Klein (1979) found an association between biological conditions (macroinvertebrate diversity and fish taxa) and the percent of watershed imperviousness, which in turn was linked to unit-area base flow. Likewise, an index of the diversity and structure of macroinvertebrate assemblages was correlated with streamflow patterns in western Washington streams (Booth et al. in press). As in any study of urban streams, however, many other factors, including channel morphology, streambed material, nutrients, migration barriers, water temperature, and water chemistry, are likely to have influenced the biological conditions of these urban streams.

Biological Variation Associated with Streamflow Patterns

Although the biological effects of streamflow modification are difficult to isolate because of other changes in urban streams, many ecological investigations have documented how the rate, timing, and sources of streamflow affect the structure, composition, and productivity of lotic assemblages by regulating habitat conditions, availability of food sources, or natural disturbance regimes (Shelford and Eddy 1929; Horwitz 1978; Fisher et al. 1982; Schlosser 1985; Newbury 1988; Power et al. 1988; Resh et al. 1988; Townsend 1989; Power et al. 1999). Physical characteristics of water (e.g., velocity, depth, temperature, turbidity, and nutrient availability) and geomorphic features of the channel (e.g., width, bank height, and bed material) depend on streamflow. These characteristics and features define habitat units that compose the physical template for stream ecosystems in which distinct hydraulic conditions and substrates favor specific types of organisms or provide refugia during disturbance (Stehr and Branson 1938; Minshall 1984, 1988; Statzner and Higler 1986; Statzner et al. 1988; Townsend 1989; Townsend et al. 1997). The area, volume, and diversity of habitats generally increase with streamflow, which may account for the increasing diversity and productivity of lotic assemblages in larger streams and rivers (Harrel et al. 1967; Platts 1979; Allan 1995; Fairchild et al. 1998; Wiberg-Larsen et al. 2000). Hydrologic variability generally decreases with increasing stream size, which promotes habitat stability and may also promote increased diversity of lotic assemblages with stream size (Horwitz 1978). A relatively uniform distribution of streamflow is generally associated with relatively high groundwater inflow, which produces more stable seasonal temperatures, less ice cover in cold regions, higher dissolved oxygen during summer, and stable habitat volume (Power et al. 1999).

Disturbances in the form of high flows or hydrologic droughts (i.e., streamflow very low or absent) reduce periphyton biomass, macrophyte and fish populations, and the diversity of lotic assemblages (Stehr and Branson 1938; Douglas 1958; Anderson and Lehmkuhl 1968; Fisher et al. 1982; McAuliffe 1984; Schlosser 1985; Erman et al. 1988; McElravy et al. 1989; McCormick and Stevenson 1991; Boulton et al. 1992; Bayley and Osborne 1993; Closs and Lake 1994; Dieterich and Anderson 1995; Wootton et al. 1996). High flows, in particular, have a direct effect on stream biota by scouring streambeds, killing organisms or transporting them downstream. High flows also modify the trophic structure of streams by transporting dissolved nutrients and particulate organic material (Biggs and Close 1989; Anderson and Lehmkuhl 1968) or by selecting for benthic insects with traits (e.g., morphology) that affect their suitability as fish prey (Wootton et al. 1996; Perry et al. 2003). Lotic communities quickly recover after an individual flood or period of no flow (Shelford and Eddy 1929; Stehr and Branson 1938; Fisher et al. 1982; Power and Stewart 1987; Scrimgeour and Winter-bourn 1989; DeBrey and Lockwood 1990; Boulton et al. 1992; Bayley and Osborne 1993; Jones et al. 1995); however, rivers and streams with frequent high flows or no-flow periods have relatively simple trophic structure, low taxonomic diversity, and high dominance by a few taxa (Gorman and Karr 1978; Schlosser 1985; White and Pickett 1985; Robinson and Minshall 1986; Power and Stewart 1987; Death and Winterbourn 1995; Rabeni and Wallace 1998). While periodic disturbance is readily accommodated in river and stream ecosystems without degrading biological productivity, diversity, or structure, frequent disturbances may have persistent biological effects.

The biological influence of disturbance is complex; it depends on the biological attribute or organism of interest and is mediated by the ecological context at a site. For example, the lack of disturbance can allow a few taxa to dominate an assemblage (Connell 1978; McAuliffe 1984; Wootton et al. 1996), as can a high frequency of disturbance. Different attributes of lotic communities do not respond consistently to disturbance. In an investigation of 11 streams in New Zealand, Death and Winterbourn (1995) found the density of invertebrate species and species richness increased with site stability, but species evenness peaked at sites with intermediate stability. The biological effects of high flows also depend on seasonal timing in relation to the life histories of aquatic organisms (Bickerton 1995; Dieterich and Anderson 1995).

The relative magnitude of streamflow is an important component of the ecological context. To account for differences in stream size in ecological investigations, high flows are often assessed with respect to the mean or median flow (Horwitz 1978; Poff and Ward 1989; Clausen and Biggs 1997). Nonhydrologic factors such as habitat diversity (Gorman and Karr 1978; Gurtz and Wallace 1984; Townsend et al. 1997) and biotic interactions (McAuliffe 1984; Feminella and Resh 1990; McCormick and Stevenson 1991; Wootton et al. 1996) also help form the broad ecological context that mediates effects of single extreme streamflow events on stream biota. Thus, the specific biological responses to changes in streamflow patterns are difficult to assess without information about the biological feature of interest and the other components of the ecosystem. Nonetheless, investigations of groups of streams representing a range of biological and physical conditions provide evidence of systematic biological responses to streamflow patterns.

Previous investigations have provided frameworks for ecological analyses of streamflow patterns. Poff and Ward (1989) developed a classification system of regional hydrologic regime. Their conceptual model included a hierarchy of four flow characteristics: duration of intermittency, high flow frequency, high flow predictability, and overall flow variability. Richter et al. (1996) assessed flow alteration of regulated rivers in terms of five characteristics of flow: magnitude, duration, frequency, timing, and rate of change. Clausen and Biggs (2000) used principal component analysis to identify four distinct categories of streamflow statistics for New Zealand streams: central tendency (stream size), flow variability, volume of high flows, and frequency of high-flow events. Clausen and Biggs (2000) did not include any statistics for the timing of flows or rate of change. Each of these frameworks include streamflow statistics that are likely to be modified by urban development. Unlike the effects of regulation by a dam or other control structure, the hydrologic consequences of urban development are incremental. Streamflow patterns are produced by runoff processes, not by controlled releases through penstocks and spillways. As a result, urban streamflow does not necessarily have clearly artificial patterns such as truncated peaks, stepped ramping of flows, or high frequency (diurnal) fluctuations. Instead, urban development causes broad changes in the rate and timing of runoff that should be evident from a limited set of streamflow statistics representing ecologically significant streamflow patterns.

Ecologically Significant Variation in Streamflow Patterns

Comparative analyses of rivers and streams provide evidence for the strongest associations between streamflow patterns and biological conditions, though not necessarily the causative mechanisms. Two studies—Poff and Allan (1995) and Clausen and Biggs (1997)—are reviewed to define ranges of streamflow variables associated with attributes of lotic assemblages. These studies did not attribute streamflow patterns to differences in land use, though it is likely that land use had an influence in some basins. Instead, they examined rivers and streams from wide geographic regions where various physiographic factors such as drainage area, climate, and geology produced different streamflow patterns.

We assume that natural physiographic factors produce differences in streamflow among the Poff and Allan (1995) and Clausen and Biggs (1997) streams comparable to the hydrologic changes resulting from urban development. Thus, these streams provide a natural correlate of an urban gradient to assess the biological response to differences in streamflow (e.g., streamflow is typically flashier in smaller streams than in large rivers). This approach avoids the covarying and, thus, confounding effects from water quality, channel form, and riparian zones that also vary with streamflow patterns across an urban gradient. Hydrologic changes in urban streams, however, may not be strictly analogous to hydrologic differences between streams. In urban streams, hydrologic changes occur over decadal time scales, so species do not have the time to evolve to modified streamflow regimes that they have had in natural systems. Thus, the biological response to changes in streamflow patterns over decades in a stream may be greater than the biological variation between streams that have long standing natural differences in streamflow pattern.

Poff and Allan (1995) analyzed relations between four streamflow statistics and fish assemblages at 34 sites in rivers in Wisconsin and Minnesota. Streamflow statistics were calculated from records collected over various time periods spanning at least 20 years. Two functional groups of fish were identified based on species presence/absence at the sites. Canonical discriminant analysis was used to identify hydrologic characteristics of the sites associated with the two groups. A canonical variate composed of daily flow predictability, base flow stability, coefficient of variation (CV) of daily flows, and frequency of spates distinguished the two functional groups of fishes based on hydrologically variable or hydrologically stable sites. Daily flow predictability and base flow stability were significantly different ($P < 0.05$, Wilcoxon rank-sum test) between the two groups of sites, but the CV of daily flows and frequency of daily flow with a 1.67-year return interval were not (Table 1). A comparison of species traits for the two functional groups indicated that the hydrologically variable group had more trophic and habitat generalists tolerant of silt than the hydrologically stable group.

Clausen and Biggs (1997) analyzed relations between 34 streamflow statistics and benthic (periphyton and invertebrate) assemblages in 83 rivers in New Zealand. Streamflow statistics were calculated from contemporaneous records spanning 1–7-year periods. They found that periphyton biomass was inversely related to mean flow, median flow, and the frequency of high flows (daily streamflow equal to or greater than three times the median daily streamflow). Periphyton biomass was directly related to 90th-percentile flow. Periphyton species richness was inversely related to flood volume/base flow volume, frequency of high flows, and the product of frequency and duration of high flow. Periphyton species diversity was related to high flow and variation (CV) statistics but not to central-tendency or low-flow statistics. Invertebrate density and richness were inversely related to mean, median, and low flows and directly related to flow variability and the frequency of high flows. The frequency of high flows provided the single most useful measure to account for the variation in measures of benthic assemblages among the rivers.

Clausen and Biggs (1997) proposed that flood disturbance (scour, deposition, abrasion of bed mate-

TABLE 1. Median values of streamflow statistics for 34 sites in Wisconsin and Minnesotta classified by Poff and Allan (1995) as hydrologically stable or hydrologically variable based on fish assemblages.

Streamflow statistics	Sites with hydrologically stable fish assemblage	Sites with hydrologically variable fish assemblages
Frequency of daily flow with 1.67 year return interval	0.69	0.77
Daily coefficient of variation	1.3	1.9
Daily flow predictability[a]	70	46
Baseflow stability[a]	0.36	0.05
Annual frequency of flows > 10th percentile[a]	4.9	6.0
Annual frequency of flows > three times median flow[a]	3.9	6.5
T_{Qmean} (fraction of time that flow exceeds mean flow)	0.26	0.25
CV (LN transformed data)[a, b]	0.7	2.1
Median daily percent change in streamflow[a]	5%	7%
90th percentile flow/median flow[a]	0.52	0.21

[a] Values were significantly different between the two groups ($P < 0.05$, two-tailed Wilcoxon rank sum test).
[b] Coefficient of variation of natural log transformed daily flows.

rial) and relative flow stability were the dominant hydrologic mechanisms affecting the benthic assemblages. They also found that mean and median flows were negatively correlated with periphyton and invertebrate density and invertebrate richness because of decreasing bed stability and heterogeneity of bed material with stream size. Thus, the biological influence of stream size was not solely hydrologic but instead also depended on sediment supply and the size of structural features (wood, cobbles, valley width) relative to the stream channel.

Clausen and Biggs (2000) also suggested additional reasons for lower density and richness of benthic assemblages in larger rivers. These include lower velocity, lower gradient, finer bed material, reduced allochothonous inputs, and increased light availability. Inorganic nitrogen and phosphorous concentrations were also negatively correlated with specific yield (a measure of stream size) and high flow magnitude, and were likely to control autotrophic production between high flows. None of these reasons for a biological response are solely a function of streamflow patterns.

Four types of streamflow patterns examined by Poff and Allan (1995) and Clausen and Biggs (1997) were thus associated with variation of fish and benthic assemblages: high flow frequency, streamflow distribution, daily variation, and low flow magnitude (Table 2). The biological response of lotic assemblages to variation in each of these patterns has a mechanistic basis. Increased high flow frequency could result in shorter periods between disturbances and increased seasonal periods with episodic high flows. Change in high flow frequency is particularly an issue in regions where the reproduction and growth cycles of aquatic and benthic organisms are tied to stable flow periods. The likelihood and spatial extent of bed disturbance increase with storm flow frequency (Konrad et al. 2002). The distribution of streamflow, as characterized by its central tendency (i.e., mean or median) provides a gross indication of the amount (volume and area) of habitat available to aquatic organisms. Measures of variability provide information on habitat heterogeneity over time. Stream ecosystems where storm flow, rather than base flow, constitutes much of the total distribution of streamflow will have short periods when hydraulic conditions, nutrient transport, and amount of habitat are at high levels (though not necessarily optimal) and extended periods of time when these conditions and processes are at low levels. Movement of nutrients and particulate organic material is also likely to be skewed to short periods of rapid transport and limited retention. Increased daily variation in flow may reduce the time that a given location is suitable habitat for an organism and cause the organism to expend energy moving to suitable locations. Reduced low flows can reduce the area of benthic habitat and volume of aquatic habitat, leading to lower population sizes and diversity in lotic communities.

Methods for Assessing Hydrologic Changes in Urban Streams

We examine the effects of urban development on the four types of streamflow patterns (high flow frequency,

TABLE 2. Streamflow patterns with biological responses to variation, statistics, and ranges of values for streams in each source. Statistics in italics were analyzed in this chapter for selected reference and urban streams.

Biological responses to variable:	High-flow frequency statistics	Range of values
Variation in high-flow frequency		
Dominance of trophic and habitat generalists in fish assemblages, lower periphyton density and diversity with increasing high flow frequency.	*Frequency of daily flows that exceeded three times median flow*	0 to 34 events per year[b] 1.1 to 10 events per year[c]
	Frequency of events greater than 10th percentile flow	2.4 to 10 events per year[c]
	Frequency of daily flow corresponding to annual peak flow with 1.67 return interval	0.4 to 1.1 events per year[a]
Variation in streamflow distribution		
Dominance of trophic and habitat specialists in fish assemblages with increasing mean or median flows; lower periphyton density, richness, and diversity with increasing mean and median flows or increasing stormflow relative to baseflow; lower invertebrate density and richness with increasing mean or median flow or increasing stormflow to baseflow.	*Mean streamflow*	0.4 to 520 m^3/s[b] 0.6 to 37 m^3/s[c]
	Median daily streamflow	0.3 to 468 m^3/s[c] 0.1 to 30 m^3/s[c]
	T_{Qmean}	0.17 to 0.37[c]
	Flood flow index	0.03 to 2.8[b]
Variation in daily streamflow		
Dominance of trophic and habitat generalists in fish assemblages, lower periphyton diversity, lower invertebrate density and evenness with increasing daily variation.	*Coefficient of variation of logarithms of daily streamflow*	0.4 to 27[c]
	Median daily percent change	3% to 12%[c]
	Coefficient of variation	0.4 to 3.2[a] 0.09 to 3.7[b]
Variation in low flow		
Dominance of trophic and habitat specialists in fish assemblages, higher periphyton and invertebrate density with increasing low flow	*90th percentile flow*	0 to 20.4 m^3/s[c]
	90th percentile flow/median daily flow	0.17 to 0.98[b] 0 to 0.72[c]
	Baseflow stability	0 to 0.6[a]

Sources: [a] Poff and Allan (1995), [b] Clausen and Biggs (1997), [c] reanalysis of sites examined by Poff and Allan (1995).

streamflow distribution, daily variation, low flow/flow intermittency) in 13 streams in the United States that have streamflow records of at least 30 years (Table 3). Five streams are part of the U.S. Geological Survey (USGS) Hydrologic Benchmark Network (HBN), which represents reference streams relatively unaffected by local land uses, and eight urban streams were in counties with population densities greater than 200 people/km^2 in 2000 (Figure 4).

A host of streamflow statistics can be used to demonstrate the hydrologic effects of urbanization and their relations to stream biota, but many streamflow statistics are correlated with each other because they represent the same underlying streamflow pattern. For our analysis, we selected nine statistics representing four distinct streamflow patterns (Table 2) to assess which statistics consistently change in response to urban development while remaining stationary in reference streams. Annual values of the streamflow statistics were analyzed for changes in urban streams between WY 1957–1968 and WY 1991–2000 using the Wilcoxon rank-sum test, with rejection of the null hypothesis when $P < 0.05$. For the urban streams, the specific direction of change was hypothesized for each parameter except mean, median, and low-flow statistics. The hypotheses are described below and allow a one-tailed test (i.e., reject the null hypotheses for no change or a change in the opposite direction). Two-

TABLE 3. Selected Hydrologic Benchmark Network stations and urban streams with U.S. Geological Survey station number, county and state, and drainage area.

Stream	Station number	County and state	Drainage area
Hydrologic Benchmark Network			
Elder Creek	11475560	Mendicino, CA	17 km^2
Holiday Creek	02038850	Appomattox, VA	22 km^2
Andrews Creek	12447390	Okanogan, WA	57 km^2
Mogollon Creek	09430600	Grant, NM	177 km^2
Popple River	04063700	Florence, WI	356 km^2
Urban			
Valley Stream	01311500	Nassau, NY	12 km^2
Mercer Creek	12120000	King, WA	31 km^2
Poplar Creek	05550500	Cook, IL	90 km^2
San Francisquito Creek	11164500	Santa Clara, CA	96 km^2
Morrison Creek	11336580	Sacramento, CA	137 km^2
Northeast Branch Anacostia River	01649500	Prince George's, MD	186 km^2
Peachtree Creek	02336300	Fulton, GA	222 km^2
Salt Creek	05531500	Cook, IL	294 km^2

tailed tests were used for mean streamflow, median streamflow and the two low-flow statistics because we did not have specific hypotheses regarding the direction of change for these statistics

Streamflow records for most HBN sites began later than the urban streams, so statistics for WY 1969–1978 were compared to statistics for WY 1991–2000 for the HBN sites. For the HBN sites, periods were compared with a two-tailed Wilcoxon rank-sum test. The null hypothesis (no change over time) was rejected when $P < 0.10$ for the HBN streams to increase the power of the test to detect changes in each statistic. By reducing the likelihood of false rejection of the null hypothesis for the HBN sites, there is greater certainty for concluding that a statistic is relatively stationary despite decadal-scale variation in weather patterns.

Streamflow data available for the period of analysis include annual maxima and daily mean streamflows. Daily mean streamflow does not capture the variability of urban streamflow, which can rise and fall rapidly during an hour or two. Thus, hydrologic measures based on daily streamflow underrep-resent the hydrologic changes associated with urban development and their biological consequences. Nonetheless, these data are consistent with those used by Clausen and Biggs (1997) and Poff and Allan (1995).

Frequency of High-Flow Events

Urban development was expected to increase the volume and peak rate of runoff from storms, which results in increased flood magnitude and frequency. Various approaches account for the relative magnitude of a flood in a particular stream, including scaling the flow by drainage area or channel width, converting the discharge to a depth and assessing it relative to bankfull depth, or using a reference discharge as a threshold for identifying a high flow. Clausen and Biggs (1997, 2000) selected a discharge three times the median streamflow as a threshold for defining "high" flows because of its association with variation in benthic assemblages.

In our reanalysis of streamflow data from sites examined by Poff and Allan (1995), the frequency of events greater than three times the median flow was significantly different between the hydrologically variable and hydrologically stable sites (Table 1). Poff and Allan (1995) used the daily discharge on the day when there was a peak discharge with a 1.67-year recurrence interval as the threshold for high flows because it referenced a significant, but not rare, disturbance to the fish assemblage. Although such an event may represent a disturbance, it does not provide much resolution between streams insofar as the annual frequency of a given streamflow is related to its recurrence interval. Indeed, the frequency of high flows was not significantly different between hydrologically stable and hydrologically variable sites using this frequency-based threshold (Table 1).

The cumulative exceedence distribution of streamflow (flow-duration curve) provides a mechanistic basis for selecting a geomorphically effective flow (Wolman and Miller 1960) capable of disturb-

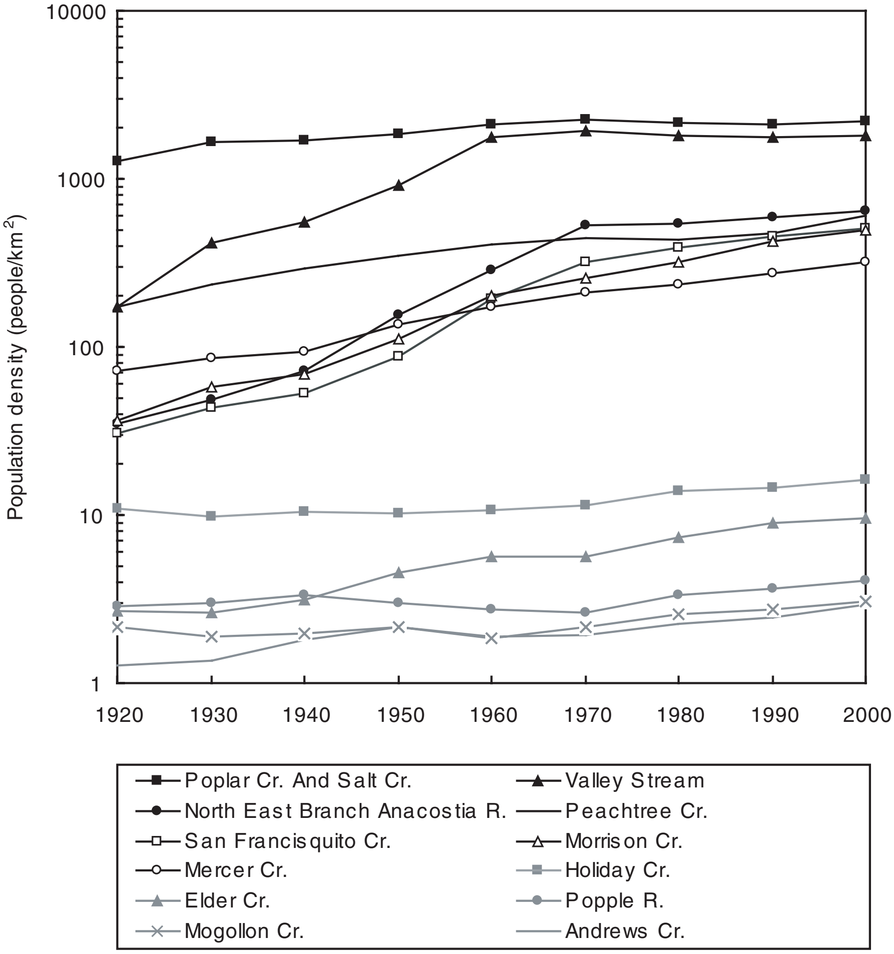

FIGURE 4. Decadal population densities from 1920 to 2000 for the counties where the selected streamflow gauges are located. Refer to Table 3 for stream locations. Data from the U.S. Census Bureau (2004).

ing a streambed. The 5th- to 10th-percentile flows represent an approximate threshold of streambed stability for many river channels (e.g., Helley 1969; Milhous 1973; Pickup and Warner 1976; Andrews 1984; Sidle 1988; Carling 1988; Konrad et al. 2002). The 5th- or 10th-percentile flows are very highly correlated; for simplicity, we use the 10th-percentile flow as the high flow threshold. When high flow frequency was reanalyzed at the sites investigated by Poff and Allan (1995), hydrologically variable sites had more frequent high-flow events than hydrologically stable sites (Table 1). Changes in the frequency of high-flow events in the urban and HBN streams were analyzed using as thresholds both the 10th-percentile streamflow and three times the median streamflow.

Flow Distribution

We did not expect central measures of streamflow distribution to change consistently in response to urban development, but both annual mean and annual median flows were tested for changes between the urban and reference streams because of the associations of these measures with characteristics of fish and benthic assemblages, and with habitat area/volume and streamflow. Streamflow was expected to be redistributed from base flow periods to storm flow periods in response to urban development. Clausen and Biggs (1997) used the ratio of flood-flow volume to base flow volume as a "flood-flow index" (FFI). However, the flood-flow index requires a stream-specific specification of base flow for each stream and reference changes in streamflow volume. These data were not available for the streams we analyzed.

As an alternative measure of the distribution of streamflow, we analyzed the fraction of time that streamflow exceeds mean streamflow ($T_{Q\mathrm{mean}}$) in the urban and reference streams. $T_{Q\mathrm{mean}}$ provides a relative measure of the distribution of storm flow and base flow comparable to the flood-flow index, but it is based on the duration rather than the volume of streamflow. In response to urban development, $T_{Q\mathrm{mean}}$ is expected to decrease because there is a shorter period of the year when streamflow is greater than mean streamflow (Konrad and Booth 2002).

In the reanalysis of the Poff and Allan (1995) sites, $T_{Q\mathrm{mean}}$ was not significantly different between the hydrologically variable and hydrologically stable sites (Table 1). Four sites with hydrologically variable fish assemblages were associated with hydrologic stability (higher $T_{Q\mathrm{mean}}$) possibly because of either low base flow stability or a short spate-free period. When these sites were excluded from the analysis, $T_{Q\mathrm{mean}}$ was significantly lower (P = 0.02, Wilcoxon rank-sum test) for the hydrologically variable group (median = 0.22), than the hydrologically stable group (median = 0.26). Annual values of $T_{Q\mathrm{mean}}$ were tested to assess whether there was a significant redistribution of streamflow from base flow to storm flow in urban streams and whether the distribution of streamflow between base flow and storm flow was relatively stationary year to year in the reference streams.

Daily Variation of Streamflow

Daily variation of streamflow was analyzed to represent variability, distinct from high-flow disturbances and the gross distribution of storm flow and base flow. Two statistics describing daily variation were examined: the CV of the logarithms of daily streamflow and the percentage daily change in streamflow. The values of both variability statistics were expected to increase in response to urban development. The logarithmic transformation reduced the sensitivity of CV to high flows (Stedinger et al. 1993). Because of zero values, 1 m^3/s was added to each streamflow value before taking the logarithm.

Although the CV of untransformed daily streamflow was not significantly different between the hydrologically variable and hydrologically stable sites analyzed by Poff and Allan (1995), a reanalysis of these sites showed significantly higher values of CV of log-transformed streamflow for hydrologically variable sites (median = 2.1) compared to hydrologically stable sites (median = 0.7) (Table 1). The CV of log transformed daily streamflow was included in the analysis of urban and reference streams to represent routine variation in hydraulic conditions.

The absolute value of the relative change in daily streamflow, $|Q_d - Q_{d-1}|/Q_d$, where Q_d is daily discharge for day d (expressed as the daily percent change in daily flow), was used as an alternative measure of daily flow variability. This statistic weights the relative changes in high flows and low flows equally and, thus, is less sensitive to changes in high flows than is CV. In a reanalysis of the sites in Poff and Allan (1995), the median daily percentage change in streamflow was significantly higher at hydrologically variable sites (median = 7%) compared to hydrologically stable sites (median = 5%) (Table 1).

Low Flow Magnitude

As noted earlier, low flows are not consistently affected by urban development, but we assessed changes in low flows because of their ecological significance. Two low-flow statistics were tested for significant changes over time in the HBN and urban streams: 90th-percentile flow and 90th-percentile flow divided by median streamflow. The 90th-percentile flow provides an absolute measure of streamflow during relatively common low-flow periods but does not account for extremely low flows. The 90th-percentile flow divided by the median flow, from Clausen and Biggs (1997), provides a relative measure that should be less sensitive to annual variation in low flows. In our reanalysis of the Poff and Allan (1995) sites, the values of 90th-percentile flow divided by the median flow were significantly lower at hydrologically variable sites (median = 0.21) compared to hydrologically stable sites (median = 0.52) (Table 1).

Temporal Changes in Streamflow Patterns at Urban and Reference Streams

High Flow Frequency

The frequency of high flows increased from 1958–1967 to 1991–2000 in most of the urban streams, but not in any of the HBN streams (Table 4). The frequency of events greater than the 10th-percentile flow increased in four of the urban streams, while the frequency of events greater than three times the me-

TABLE 4. Median annual frequency of high-flow events.

Stream	Frequency of events exceeding the 10th-percentile flow (events per year)		Frequency of events exceeding the three times median (events per year)	
	Period 1[a]	Period 2[b]	Period 1[a]	Period 2[b]
Hydrologic Benchmark Network				
Elder Cr.	10	9	7	9
Holiday Cr.	20	22	20	22
Andrews Cr.	3	3	2	4
Mogollon Cr.	6	10	12	15
Popple R.	5	8	7	11
Urban				
Valley Stream	26	30	25	42[c]
Mercer Cr.	18	27[c]	19	32[c]
Poplar Cr.	9	19[c]	14	26[c]
San Francisquito Cr.	7	10	15	10
Morrison Cr.	12	17	12	23[c]
Northeast Branch Anacostia R.	25	34[c]	28	41[c]
Peachtree Cr.	27	33	36	48[c]
Salt Cr.	13	19[c]	20	24

[a] Period 1 is 1969 to 1978 for Hydrologic Benchmark Network streams and 1958 to 1967 for urban streams.
[b] Period 2 is 1991 to 2000.
[c] Probability < 0.05 that frequency is the same for both periods or lower in period 2 based on a one-tailed Wilcoxon rank-sum test.

dian flow increased in six of the urban streams. Only one urban stream, San Francisquito Creek in central California, did not have a significant increase in either high-flow statistic. Streamflow in San Francisquito Creek was intermittent and the high flow regime is likely to reflect storm patterns driven by decadal-scale oscillations in ocean conditions. In this case, it appears that the frequency of events greater than the 10th-percentile flow may be less sensitive to climatic conditions than the frequency of events greater than three times the median flow.

Both of these statistics represent relatively common high-flow events with median annual values for 1991–2000 ranging from 10 to 48 events per year for urban streams and from 3 to 22 events in the HBN streams. The frequency of events greater than the 10th-percentile flow was higher in the urban streams than the streams analyzed by Poff and Allan (1995) (2.4–10.3 events per year) and, thus, all represent hydrologically variable streams by their criterion even during the earlier period from 1958 to 1967. Likewise, the change in the frequency of events exceeding three times the median flow in the urban streams was outside the range for the streams examined by Poff and Allan (1995) (1.1–10 events per year) but spanned part of the range found by Clausen and Biggs (1997) (0–34 events per year) (Table 2).

Streamflow Distribution

The distribution of streamflow changed between 1958–1967 and 1991–2000 in many of the urban streams but not between 1969 and 1978 and 1991–2000 in the HBN streams. Mean streamflow increased significantly between 1958–1967 and 1991–2000 in two urban streams (Table 5). The increases in mean and median streamflow, however, were much less than the three to four order-of-magnitude ranges associated with responses in either fish or benthic assemblages (Table 2).

The redistribution of runoff from base flow to storm flow periods resulted in significant decreases in T_{Qmean} from 1958–1967 to 1991–2000 in three urban streams. In Valley Stream, Long Island, T_{Qmean}, decreased from 0.24 for 1958–1967 to 0.15 for 1991–2000 (Table 5). The decrease in T_{Qmean} corresponded to redistribution of annual streamflow volume from more than 60% base flow (FFI = 0.67) prior to 1960 to less than 20% base flow (FFI = 4) after 1966 (Simmons and Reynolds 1982). The urban streams without significant changes in T_{Qmean} had

TMABLE 5. Median annual values of streamflow distribution statistics.

Stream	Mean daily streamflow (m^3/s)		Median daily streamflow (m^3/s)		Fraction of time mean streamflow is exceeded	
	Period 1[a]	Period 2[b]	Period 1[a]	Period 2[b]	Period 1[a]	Period 2[b]
Hydrologic Benchmark Network						
Elder Cr.	0.85	0.85	0.19	0.20	0.25	0.25
Holiday Cr.	0.28	0.25	0.17	0.19	0.23	0.28
Andrews Cr.	0.91	1.07	0.15	0.18	0.20	0.22
Mogollon Cr.	0.32	0.98	0.10	0.41	0.29	0.30
Popple R.	3.30	3.07	1.95	2.05	0.28	0.31
Urban						
Valley Stream	0.07	0.04	0.04	0.01	0.24	0.15[d]
Mercer Cr.	0.57	0.64	0.37	0.40	0.33	0.26[d]
Poplar Cr.	0.57	0.83[c]	0.21	0.44[c]	0.26	0.28
San Francisquito Cr.	0.18	0.91	0.00	0.06	0.15	0.16
Morrison Cr.	0.40	0.78	0.17	0.15	0.14	0.15
Northeast Branch Anacostia R.	2.03	2.79	1.08	1.33	0.24	0.21[d]
Peachtree Cr.	3.40	4.11	1.67	1.66	0.20	0.20
Salt Cr.	2.75	5.03[c]	1.29	3.04[c]	0.27	0.28

[a] Period 1 is 1969 to 1978 for Hydrologic Benchmark Network streams and 1958 to 1967 for urban streams.
[b] Period 2 is 1991 to 2000.
[c] Probability < 0.05 that value was the same for both periods based on a two-tailed Wilcoxon rank-sum test.
[d] Probability < 0.05 that value was the same for both periods or higher for period 2 based on a one-tailed Wilcoxon rank-sum test.

low values (0.2 or lower) during the initial period of the analysis or had significant increases in base flow (Salt and Poplar Creeks). The changes in T_{Qmean} spanned much of the total range of values (0.17–0.37) from the reanalysis of streams examined by Poff and Allan (1995) (Table 3). Thus, the redistribution of runoff from base flow to storm flow in many, but not all, urban streams is likely to be ecologically significant.

Daily Flow Variability

Daily flow variability increased in many urban streams from 1958–1967 to 1991–2000. Although the CV of the logarithm of daily flows increased in five urban streams (Table 6), the changes were significant in only three of the streams because of interannual variability. No differences between 1969–1978 and 1991–2000 were significant for the HBN streams.

The median daily percentage change in streamflow appeared to be more sensitive to urban development, increasing in five urban streams between 1958–1967 and 1991–2000 and generally having higher values in urban streams than in HBN streams (Table 6). One HBN stream had a significant increase in the median daily percentage change in streamflow, and one HBN stream had a significant decrease from 1969–1978 to 1991–2000. The median daily percentage change for urban streams was more variable (10–40%), than in the HBN streams (3–12%) or the reanalyzed streams of Poff and Allan (1995) (3–12%).

Low Flow

The 90th-percentile streamflow increased significantly in four of the urban streams from 1958–1967 to 1991–2000 and increased significantly one HBN stream from 1969–1978 to 1991–2000 (Table 7). Two of the urban streams had significant increases in 90th-percentile flow normalized by median discharge, but none of the HBN streams had significant changes (Table 7). The increase in low flow may be a result of many factors, including wastewater discharges, or uses of deep groundwater or water imported from other basins that contribute to base flow (e.g., landscape irrigation or septic system effluent). These changes in low flow were small relative to the range of value reported by Clausen and Biggs (1997) and for the reanalysis of the streams examined by Poff and Allan

TABLE 6. Median annual values of streamflow-variability statistics.

Stream	Daily percentage change in streamflow		Coefficient of variation of logarithms of daily streamflow	
	Period 1[a]	Period 2[b]	Period 1[a]	Period 2[b]
Hydrologic Benchmark Network				
Elder Cr.	6%	6%	1.72	1.62
Holiday Cr.	10%	10%	0.77	0.74
Andrews Cr.	4%	3%	1.36	1.25
Mogollon Cr.	10%	12%[c]	1.17	1.43
Popple R.	5%	5%	0.78	0.76
Urban				
Valley Stream	18%	40%[d]	0.71	0.74
Mercer Cr.	11%	15%[d]	0.80	0.80
Poplar Cr.	14%	19%[d]	1.19	1.17
San Francisquito Cr.	0%	10%	1.13	2.08
Morrison Cr.	11%	11%	0.82	1.12[d]
Northeast Branch Anacostia R.	15%	19%[d]	0.91	0.93
Peachtree Cr.	15%	21%[d]	0.90	0.98[d]
Salt Cr.	13%	12%	0.97	0.80[d]

[a] Period 1 is 1969 to 1978 for Hydrologic Benchmark Network streams and 1958 to 1967 for urban streams.
[b] Period 2 is 1991 to 2000.
[c] Probability < 0.10 that value was the same for both periods based on two-tailed Wilcoxon rank-sum test.
[d] Probability < 0.05 that value was the same for both periods or lower for period 2 based on a one-tailed Wilcoxon rank-sum test.

TABLE 7. Median annual values of low-flow statistics.

Stream	90th-percentile streamflow (m^3/s)		90th-percentile streamflow/ median daily streamflow	
	Period 1[a]	Period 2[b]	Period 1[a]	Period 2[b]
Hydrologic Benchmark Network				
Elder Cr.	0.03	0.03	0.20	0.17
Holiday Cr.	0.07	0.08	0.41	0.51
Andrews Cr.	0.07	0.10[c]	0.45	0.55
Mogollon Cr.	0.01	0.02	0.09	0.09
Popple R.	1.06	1.13	0.54	0.58
Urban				
Valley Stream	0.01	0.00	0.16	0.00
Mercer Cr.	0.15	0.20[d]	0.40	0.54[d]
Poplar Cr.	0.03	0.06[d]	0.20	0.14
San Francisquito Cr.	0.00	0.01[d]	0.50	0.23
Morrison Cr.	0.10	0.10	0.65	0.62
Northeast Branch Anacostia R.	0.35	0.48	0.34	0.39
Peachtree Cr.	0.69	0.72	0.41	0.46
Salt Cr.	0.48	1.58[d]	0.40	0.50[d]

[a] Period 1 is 1969 to 1978 for Hydrologic Benchmark Network streams and 1958 to 1967 for urban streams.
[b] Period 2 is 1991 to 2000.
[c] Probability < 0.10 that the value was the same for both periods based on a two-tailed Wilcoxon rank-sum test.
[d] Probability < 0.05 that value was the same for both periods based on a two-tailed Wilcoxon rank-sum test.

(1995) (Table 2). Moreover, increases in low flow would likely promote biological responses opposite from the other hydrologic effects of urban development (Table 2).

Framework for Assessing the Ecological Effects of Urban Streamflow Patterns

Urban development modifies the basic hydrologic processes generating runoff in many river basins. In our analysis of eight urban streams, changes in some streamflow patterns were comparable to hydrologic variation associated with demonstrated differences in the structure and composition of lotic communities in other streams. Hydrologic changes in urban streams are likely to affect three streamflow characteristics with ecological consequences: high-flow frequency, distribution of water between storm flow and base flow, and daily flow variability. These changes may contribute to dominance of fishes representing trophic and habitat generalists and also to lower periphyton biomass, diversity, and richness.

Biological responses to urban streamflow patterns also depend on the larger physiographic setting (climate, sediment supply, and channel/valley geomorphology) and other urban stressors (habitat alteration, changes in water chemistry). For example, even though minimum flows were not lower in the urban streams considered here, water depths during low flow may be less in an urban stream if its channel has widened in response to increased high flows.

Effects of urban development are neither uniform nor invariant—naturally "flashy" systems in arid regions may not become much more flashy after urbanization. It is likely that hydrologic changes are greatest in small to intermediate-sized streams with naturally low seasonal and storm flow variability. Large rivers may be buffered because urban development does not cover the whole basin, precipitation may not fall over the whole basin during small storms, and channel routing and overbank storage may attenuate high flows. Poff and Allan (1995) noted that large streams may function like headwater streams if they are seasonally variable. Likewise, the lotic communities in urban streams may function like those in smaller and more hydrologically variable streams because of hydrologic modification.

In streams and rivers with hydrologic changes from urban development, those changes are not necessarily outside the bounds of "natural" variation in streamflow patterns as defined by all streams spanning the range from arid to humid climates. Likewise, variation in weather patterns produces a wide range of natural variation in streamflow regardless of land use. Both types of natural hydrologic variability make it difficult to attribute specific streamflow patterns to land-use changes and to assign specific ecological consequences to streamflow patterns, because lotic species are adapted to natural hydrologic variation. Only through the systematic assessment of a long-term record is it possible to identify those changes that are persistent in hydrologic effects and ecological consequences.

Management Responses to Hydrologic Modification in Urban Streams

Two management questions arise if hydrologic modification is acknowledged as a significant factor in the biological degradation of urban streams. First, are the hydrologic influences on urban streams limited to a few, identifiable mechanisms that could become objectives (or priorities) for mitigation through stormwater management? Second, given the difficulties in accomplishing hydrologic restoration, what is the best possible condition for urban streams under an urban streamflow regime and what marginal effects might result from managing other aspects of the ecosystem (e.g., water quality treatment, stream habitat, riparian zone)?

The varied nature of urban-induced hydrologic changes and the many ways in which aquatic biota interact with flow regime suggest that there are not one or two aspects of an altered hydrologic regime that cause biological consequences. The belief in "single-issue" hydrologic changes is reminiscent of past approaches to stormwater management, namely to achieve flood control by limiting peak discharges. The single-issue focus failed to protect biological systems, because single-storm peak discharge is not a hydrologic characteristic of particular significance to biota. Ironically, that focus has even failed to provide flood protection, because most analyses have not recognized the full range of urban-induced hydrologic changes, including redistribution of streamflow, increased daily variation of streamflow, sequential peak flow events (Booth et al. 2002), and extended flow durations that result in enhanced sediment transport and deposition (Booth 1990). Inspection of comparative hydrographs (e.g., Figure 2) reveals changes in all seasons, across multiple temporal scales, and at every level of discharge. Even if hydrologic changes are limited to the types identified by the analysis of long-term streamflow patterns, mitigation would require greatly increased storage capacity of stormwater management systems. Moreover, the predevelopment patterns of many streams could not be replicated by large detention ponds with simple control structures to release the water because they cannot support gradual, sustained recessional flows after storms.

The variety of hydrologic changes is symptomatic of an underlying cause, namely the pervasive loss of water storage capacity of hillslopes that accompanies urban development where it reduces vegetation, topographic depressions, soil depths, and infiltration capacity of the land surface. Ultimately, any true solution must account for each of these hydrologic changes. The classic mitigation approach, detention ponds, fails because (1) it replaces only a scant fraction of the storage capacity of hillslopes that was lost (Booth and Jackson 1997); (2) it converts what was once spatially distributed subsurface runoff into a point discharge at a surface water outfall; and (3) it reduces the rate and changes the location of groundwater recharge and subsequent discharge because even the largest detention ponds cannot delay the production of runoff from large or long storms in the same fashion as large areas of natural landscape with intact vegetation, topography, and soils. For these reasons, objective assessments of detention-pond performance are rarely encouraging (e.g., Maxted and Shaver 1999).

Directly addressing the loss of long-term storage of storm flow is more challenging and much less widely implemented. Two other traditional engineering approaches, infiltration ponds and bypass pipelines, provide plausible opportunities for hydrologic mitigation but are each constrained by site requirements. Infiltration ponds (i.e., centralized facilities to store and reintroduce stormwater into the groundwater system) require deep infiltrative geologic strata at or near the ground surface to accommodate the large volumes of runoff collected. In many areas. such geologic conditions are ubiquitous; in others, they are widely scattered or absent altogether. Bypass systems provide partial hydrologic mitigation at best—they reduce total postdevelopment runoff volume in noninfiltrative soils and can provide nearly fail-safe reductions of peak flows and/or flow durations. However, they can leave the plethora of small and moderate discharges from paved surfaces nearly unaffected. Conversely they may eliminate all base flow once contributed from now-paved upland areas. In either case, bypass systems release all runoff as surface flow at a point discharge without any opportunity for groundwater recharge.

These considerations lead to the conclusion that hydrologic restoration requires the distributed retention of stormwater in reservoirs with a combined volume equivalent to the original soil layer and land surface. Where the land surface was steep, the soil was thin, and its infiltration capacity naturally low, the changes caused by urbanization may not be great, and achieving an equivalent degree of stormwater retention in the built environment may be relatively simple. Where predevelopment soil depths naturally approach a meter or more and porosities approach 50% and where wetlands retained much of the rainfall, simple strategies and normal land-development practices will not be successful at reestablishing streamflow patterns.

Recent efforts to achieve more comprehensive hydrologic mitigation have involved a suite of engineering and site-design approaches collectively known as “low impact development” (LID) (U.S. Environmental Protection Agency 2000). This strategy seeks to store, infiltrate, evaporate, or otherwise slowly release stormwater runoff in a close approximation of the rates and processes of the predevelop-ment hydrologic regime (Konrad and Burges 2001). To date, LID has been pursued primarily in humid, temperate regions where urban development can create the greatest alteration to the predevelopment hydrologic regime. Challenges for more widespread use of this approach include uncertainty in its application on relatively noninfiltrative soils, its effectiveness in mitigating high-intensity and (or) large-volume storms, and its construction in new or previously developed areas. Long-term performance, coupled with the uncertain level of long-term maintenance, require further study before the effectiveness of this stormwater-management approach can be advocated or its biological consequences evaluated. Until such time, we recognize no stormwater management strategy, or suite of approaches, that can achieve anything approaching full hydrologic mitigation. Some significant, measurable degree of biological decline is thus unavoidable in urban watersheds where streamflow has been altered for the foreseeable future.

Ecological Management Goals for Urban Streams

Streams nominally protected under land-use regulations have still experienced significant biological degradation. Widespread adoption of some types of development and stormwater management strategies may prevent further degradation of some streams. Yet, the full range of hydrological and ecological processes cannot be restored in urban streams, and we see no basis to expect that improvements in other aspects of stream ecosystems can mitigate the hydrologic consequences of urban development (Booth et al. 2004; Konrad 2003b). As a result, urban stream managers face the difficulty of acknowledging limits on stream ecosystem improvement imposed by the failure to restore streamflow patterns while maintaining prospects for future ecosystem recovery.

Stream restoration in the Pacific Northwest typically focuses on habitat elements in fish-bearing streams include large woody debris, pools, protective cover, gravel deposits, floodplains, and riparian vegetation. Although these elements can be imported to a site or otherwise constructed, neither the biological effectiveness (e.g., Larson et al. 2001) nor longevity (Frissell and Nawa 1992) of artificially placed elements is well documented. Over the long term, habitat elements of streams will be created and maintained only through functioning ecological processes. In recognition of the distinction between direct manipulation of stream habitat elements for an outcome and the self-regulation of habitat conditions in natural stream ecosystems, we discriminate between short-term and long-term actions for improving the biological conditions of streams. Short-term actions are generally feasible under many different management settings but are unlikely to produce long-lasting effects; long-term actions are necessary for true ecosystem enhancement but may be intractable under present-day regulatory, economic, or land-use conditions.

Short-term actions include riparian planting, water quality source control, fish-passage projects, selective instream structures, and social amenities (Bethel and Neal 2003). They address acute problems typical to stream channels in urban and urbanizing settings (e.g., denuded vegetation buffers, point source pollution, fish blockages, simplified channel structure, and dumping of solid wastes). They are not comprehensive, so their efficacy is limited by elements of the ecosystem that are missing or degraded in urban areas. Although some aspects of hydrologic modification in urban areas such as increased frequency of high flows may be addressed in part by short-term actions such as peak flow control, the range of changes in streamflow patterns resulting from urban development are unlikely to be addressed in the short term. As a consequence, hydrologic changes are likely to have persistent effects on the biological conditions of many urban streams.

Short-term actions also acknowledge the presence of people in the urban environment. Streams are affected, often irrevocably, by activities of streamside residents in pursuit of personal esthetics or low maintenance (Schauman 1998). Management actions that enhance interactions between the public and urban streams are likely to be supported, but the financial costs for such actions are commonly greater than the ecosystem benefits. Such actions are often considered desirable because they improve "quality of life" or the stream's value as a public amenity or educational resource.

Long-term, self-sustaining actions must ultimately address all of the five aspects of stream ecosystems (Figure 1). These actions might include various types of land-use planning (e.g., preserves, zoning), minimizing or redesigning road and utility crossings, upland hydrologic rehabilitation (e.g., stormwater infiltration), erosion control, riparian vegetation restoration, and reconnection of streams with floodplains. In some cases, there are fundamental conflicts between the processes that create and maintain aspects of stream ecosystems (e.g., flooding) and extensive infrastructure and human occupation around streams. Streamflow is a key habitat-forming process, and failure to reestablish streamflow patterns almost certainly precludes full restoration of the ecosystem. Over the long term, restoration of an urban stream to a predevelopment state is likely to require the exclusion of people from the immediate environment (channel and riparian areas) and, perhaps, even upland portions of the basin, even as it requires their support to ensure its implementation and success. Support and stewardship by the surrounding community would be essential to such an ambitious goal, but people's involvement would be very different from the short-term efforts that depend on human actions in and along stream channels.

Evidence to date suggests that short-term actions alone, and even some well-intentioned and well-reasoned long-term actions, do not achieve broad ecosystem protection in the urban environment. At best, biological communities in urban streams may be diverse and complex, but they will depart significantly from predevelopment conditions. These streams can be neighborhood amenities and provide nearby residents with a connection to a place not completely managed by human actions, and with a self-sustaining and self-regulating biological community. These outcomes for urban streams should be achievable even in the absence of reestablishing natural hydrologic processes, which is the likely scenario for many streams in urban and urbanizing watersheds.

Conclusions

"Urbanization" is not a single condition; it is a collection of actions that lead to recognizable landscape forms and, changes in stream conditions. Because of the broad and important roles of streamflow in fluvial ecosystems, hydrologic changes in urban streams pose significant ecological risks that warrant evaluation along with other changes (e.g., water chemistry, physical habitat, riparian conditions) long associated with urban development. Three types of hydrologic changes of ecological significance are likely to result from urban development: increased frequency of high flows; redistribution of water from periods of base flow to periods of storm flow, and increased daily variation in streamflow. These changes do not necessarily occur in all urban streams, but they are common and need to be addressed as part of any comprehensive effort to rehabilitate urban streams. Other hydrologic changes may also occur in urban streams, particularly where surface water or shallow groundwater are used for water supplies, which may also have ecological repercussions. Urban streams can provide habitat for biological communities even if the hydrologic consequences of urban development are not addressed, but the structure and composition of these communities are likely to depart from those of the predevelopment stream. Ultimately, true restoration of urban streams can only occur if hydrologic processes and the spatial distribution of the water-storage capacity is reestablished across the urban landscape.

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