# Streams in the Urban Landscape

**Michael J. Paul and Judy L. Meyer**

**Abstract** The world's population is concentrated in urban areas. This change in demography has brought landscape transformations that have a number of documented effects on stream ecosystems. The most consistent and pervasive effect is an increase in impervious surface cover within urban catchments, which alters the hydrology and geomorphology of streams. This results in predictable changes in stream habitat. In addition to imperviousness, runoff from urbanized surfaces as well as municipal and industrial discharges result in increased loading of nutrients, metals, pesticides, and other contaminants to streams. These changes result in consistent declines in the richness of algal, invertebrate, and fish communities in urban streams. Although understudied in urban streams, ecosystem processes are also affected by urbanization. Urban streams represent opportunities for ecologists interested in studying disturbance and contributing to more effective landscape management.

**Keywords:** impervious surface cover · hydrology · fluvial geomorphology · contaminants · biological assessment

## Introduction

Urbanization is a pervasive and rapidly growing form of land use change. More than 75% of the U. S. population lives in urban areas, and it is expected that more than 60% of the world's population will live in urban areas by the year 2030, much of this growth occurring in developing nations (UN Population Division 1997, US Census Bureau 2001). Whereas the overall land area covered by urban growth remains small (2% of earth's land surface), its ecological footprint can be large (Folke et al. 1997). For example, it is estimated that urban centers produce more than 78% of global greenhouse gases (Grimm et al. 2000) and that some cities in the Baltic region claim ecosystem support areas 500 to 1000 times their size (Boland & Hanhammer 1999).

This extensive and ever-increasing urbanization represents a threat to stream ecosystems. Over 130,000 km of streams and rivers in the United States are impaired by urbanization (USEPA 2000). This makes urbanization second only to agriculture as the major cause of stream impairment, even though the total area covered by urban land in the United States is minor in comparison to agricultural area. Urbanization has had similarly devastating effects on stream quality in Europe (House et al. 1993).

M.J. Paul
Institute of Ecology, University of Georgia, Athens, GA 30602 USA
e-mail: mike@sparc.ecology.uga.edu

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Despite the dramatic threat urbanization poses to stream ecosystems, there has not been a thorough synthesis of the ecological effects of urbanization on streams. There are reviews discussing the impacts of a few aspects of urbanization [biology of pollution (Hynes 1960), physical factors associated with drainage (Butler & Davies 2000), urban stream management (Baer & Pringle 2000)] and a few general reviews aimed at engineers and invertebrate biologists (House et al. 1993, Ellis & Marsalek 1996, Suren 2000), but the ecological effects of urban growth on stream ecosystems have received less attention (Duda et al. 1982, Porcella & Sorensen 1980).

An absolute definition of urban is elusive. *Webster's New Collegiate Dictionary* defines urban as "of, relating to, characteristic of, or constituting a city," where the definition of city is anything greater than a village or town. In human population terms, the U. S. Census Bureau defines urban as "comprising all territory, population, and housing units in urbanized areas and in places of 2,500 or more persons outside urbanized areas," where urbanized areas are defined as places with at least 50,000 people and a periurban or suburban fringe with at least 600 people per square mile. The field of urban studies, within sociology, has a variety of definitions, which all include elements of concentrated populations, living in large settlements and involving some specialization of labor, alteration of family structure, and change in political attitudes (Danielson & Keles 1985). In this review, we rely on the census-based definition, as it includes suburban areas surrounding cities, which are an integral part of many urban ecological studies and represent, in many cases, areas that will develop into more densely populated centers. However, many industrial/commercial/transportation areas that are integral parts of urban and urbanizing areas have low resident population densities, but are certainly contained within our view of urban areas.

Ecological studies of urban ecosystems are growing (McDonnell & Pickett 1990, USGS 1999, Grimm et al. 2000). A valuable distinction has been drawn between ecology in cities versus ecology of cities (Grimm et al. 2000). The former refers to the application of ecological techniques to study ecological systems within cities, whereas the latter explores the interaction of human and ecological systems as a single ecosystem. Although our review focuses on stream ecology in cities, it is our hope that it will provide information of value to the development of an ecology of cities. The goal of this review is to provide a synthesis of the diverse array of studies from many different fields related to the ecology of urban streams, to stimulate incorporation of urban streams in ecological studies, and to explore ecological findings relevant to future policy development. This review is a companion to the review of terrestrial urban ecosystems by Pickett et al. (2001). The review is structured in three parts that focus on the physical, chemical, and biological/ecological effects of urbanization on streams.

## Physical Effects of Urbanization

### *Hydrology*

A dominant feature of urbanization is a decrease in the perviousness of the catchment to precipitation, leading to a decrease in infiltration and an increase in surface runoff (Dunne & Leopold 1978). As the percent catchment impervious surface cover (ISC) increases to 10–20%, runoff increases twofold; 35–50% ISC increases runoff threefold; and 75–100% ISC increases surface runoff more than fivefold over forested catchments (Fig. 1) (Arnold & Gibbons 1996). Imperviousness has become an accurate predictor of urbanization and urban impacts on streams (McMahon & Cuffney 2000), and many thresholds of degradation in streams are associated with an ISC of 10–20% (Table 1) [hydrologic and geomorphic (Booth & Jackson 1997), biological (Klein 1979, Yoder et al. 1999)].

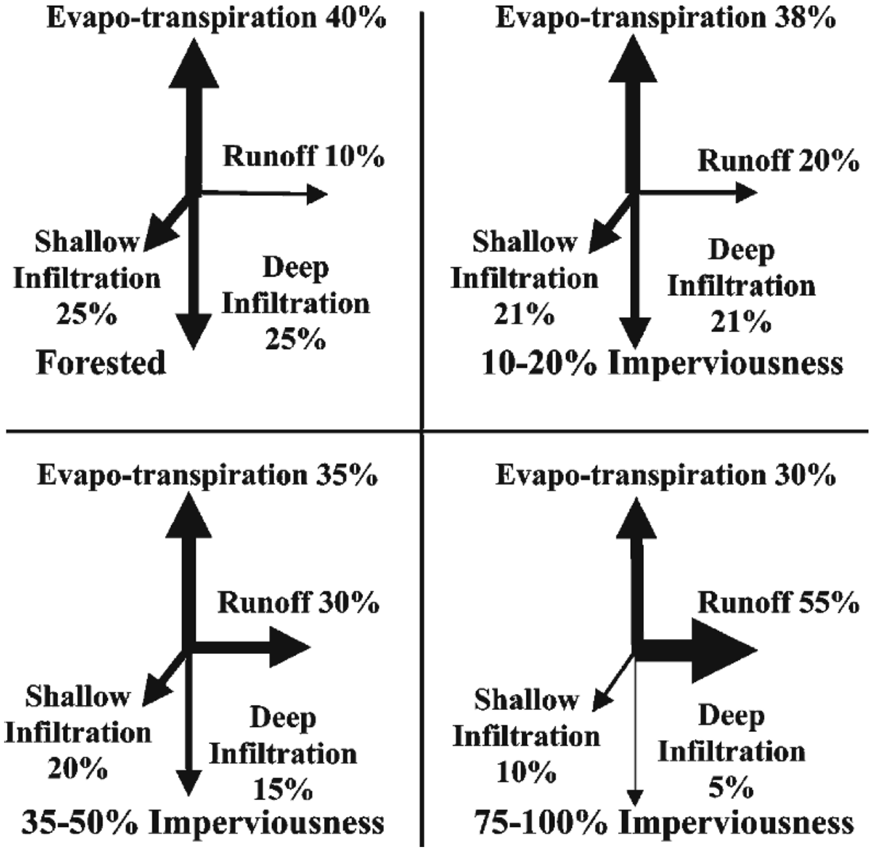


**Fig. 1** Changes in hydrologic flows with increasing impervious surface cover in urbanizing catchments (after Arnold & Gibbons 1996)

Various characteristics of stream hydrography are altered by a change in ISC. Lag time, the time difference between the center of precipitation volume to the center of runoff volume, is shortened in urban catchments, resulting in floods that peak more rapidly (Espey et al. 1965, Hirsch et al. 1990). Decreases in flood peak widths from 28–38% over forested catchments are also observed, meaning floods are of shorter duration (Seaburn 1969). However, peak discharges are higher in urban catchments (Leopold 1968). Flood discharges increase in proportion to ISC and were at least 250% higher in urban catchments than forested catchments in Texas and New York after similar storms (Espey et al. 1965, Seaburn 1969). Flood discharges with long-term recurrence intervals are less affected by urbanization than more frequent floods, primarily because elevated soil moisture associated with large storms results in greater surface runoff in forested catchments (Espey et al. 1965, Hirsch et al. 1990). Some exceptions to these observations have been noticed, largely depending on the location of urbanization within a catchment. If the ISC occurs lower in a catchment, flooding from that portion can drain faster than stormflow from forested areas higher in the catchment, leading to lower overall peak flood discharge and increased flood duration (Hirsch et al. 1990). In addition, blocked culverts and drains, swales, etc. may also detain water and lower peak flood discharges (Hirsch et al. 1990).

A further result of increased runoff is a reduction in the unit water yield: a greater proportion of precipitation leaves urban catchments as surface runoff (Fig. 1) (Espey et al. 1965, Seaburn 1969). This reduces groundwater recharge and results in a reduction of baseflow discharge in urban streams

**Table 1** Effects of impervious surface cover (ISC) resulting from urbanization on various physical and biological stream variables[a]

| Study subject | Findings | Reference |
|---|---|---|
| **Physical responses: hydrology** | | |
| Streams in Texas | Peak discharge increases and lag time decreases with ISC. | Espey et al. 1965 |
| Streams in Pennsylvania | Bankfull discharge increases and lag time decreases with catchment ISC. | Leopold 1968 |
| Review | Surface runoff increases and lag time decreases with increasing ISC (see Fig. 1). | Arnold & Gibbons 1996 |
| Streams in Washington | Increase in bankfull discharge with increasing ISC. At 10%, 2 y urban flood equals a 10 y forested flood. | Booth & Jackson 1997 |
| **Physical responses: geomorphology** | | |
| Streams in Pennsylvania | Channel enlargement increases with increasing ISC. | Hammer 1972 |
| Streams in New York | Channel enlargement begins at 2% ISC. | Morisawa & LaFlure 1979 |
| Streams in New Mexico | Dramatic changes in channel dimensions at 4% ISC | Dunne & Leopold 1978 |
| Streams in Washington | Channels begin widening at 6% ISC; channels universally unstable above 10% ISC | Booth & Jackson 1997 |
| **Physical responses: temperature** | | |
| Streams in Washington, DC | Stream temperatures increase with increasing ISC. | Galli 1991 |
| **Biological responses: fish** | | |
| Streams in Maryland | Fish diversity decreased dramatically above 12–15% ISC and fish were absent above 30–50% ISC. | Klein 1979 |
| Streams in Ontario, Canada | Fish IBI decreased sharply above 10% ISC, but streams with high riparian forest cover were less affected. | Steedman 1988 |
| Streams in New York | Resident and anadromous fish eggs and larvae densities decreased to 10% urban land use and then were essentially absent. | Limburg & Schmidt 1990 |
| Streams in Maryland | Fish diversity decreased dramatically above 10–12% ISC. | Schueler & Galli 1992 |
| Streams in Wisconsin | Fish IBI decreased rapidly at 10% ISC. | Wang et al. 1997 |
| Streams in Ohio | Fish IBI decreased rapidly between 8% and 33% urban land use. | Yoder et al. 1999 |
| **Biological responses: invertebrates** | | |
| Streams in Maryland | Invertebrate diversity decreased sharply from 1% to 17% ISC. | Klein 1979 |
| Streams in Northern Virginia | Insect diversity decreased between 15% and 25% ISC. | Jones & Clark 1987 |
| Streams in Maryland | Insect diversity metrics moved from good to poor at 15% ISC. | Schueler & Galli 1992 |
| Streams in Washington | Insect IBI decreased sharply between 1% and 6% ISC, except where streams had intact riparian zones. | Horner et al. 1997 |
| Streams in Ohio | Insect diversity, biotic integrity decreased between 8% and 33% ISC. | Yoder et al. 1999 |

[a]IBI, index of biotic integrity.

(Klein 1979, Barringer et al. 1994). However, this phenomenon has been less intensively studied than flooding, and the effects of irrigation, septic drainage, and interbasin transfers may mitigate the effects of reduced groundwater recharge on baseflow (Hirsch et al. 1990). Baseflow may also be augmented by wastewater treatment plant (WWTP) effluent. The Acheres (Seine Aval) treatment plant, which serves 8.1 million people, discharges 75 km west of Paris and releases 25,000 liters/s during low flow periods (Horowitz et al. 1999), increasing baseflow discharge in the Seine by up to 40% during low flow periods. More strikingly, wastewater effluent constitutes 69% annually and at times 100% of discharge in the South Platte River below Denver, Colorado (Dennehy et al. 1998). In our experience, high percentage contributions of wastewater discharge to urban rivers are not uncommon.

## *Geomorphology*

The major impact of urbanization on basin morphometry is an alteration of drainage density, which is a measure of stream length per catchment area ($km/km^2$). Natural channel densities decrease dramatically in urban catchments as small streams are filled in, paved over, or placed in culverts (Dunne & Leopold 1978, Hirsch et al. 1990, Meyer & Wallace 2001). However, artificial channels (including road culverts) may actually increase overall drainage densities, leading to greater internal links or nodes that contribute to increased flood velocity (Graf 1977, Meyer & Wallace 2001).

A dominant paradigm in fluvial geomorphology holds that streams adjust their channel dimensions (width and depth) in response to long-term changes in sediment supply and bankfull discharge (recurrence interval average = 1.5 years) (Dunne & Leopold 1978, Roberts 1989). Urbanization affects both sediment supply and bankfull discharge. During the construction phase erosion of exposed soils increases catchment sediment yields by $10^2$–$10^4$ over forested catchments and can be more exaggerated in steeply sloped catchments (Wolman 1967, Leopold 1968, Fusillo et al. 1977). Most of this export occurs during a few large, episodic floods (Wolman 1967). This increased sediment supply leads to an aggradation phase as sediments fill urban channels (Fig. 2). During this phase stream depths may decrease as sediment fills the channel, and the decreased channel capacity leads to greater flooding and overbank sediment deposition, raising bank heights (Wolman 1967). Therefore, overall channel cross-sections stay the same or even decrease slightly (Robinson 1976). Ironically, the flooding associated with aggradation may help attenuate increased flows resulting from increased imperviousness by storing water in the floodplain, temporarily mitigating urban effects on hydrography (Hirsch et al. 1990).

After the aggradation phase sediment supply is reduced and geomorphic readjustment initiates a second, erosional phase (Fig. 2). High ISC associated with urbanization increases the frequency of bankfull floods, frequently by an order of magnitude or, conversely, increases the volume of the bankfull flood (Leopold 1973, Dunne & Leopold 1978, Arnold et al. 1982, Booth & Jackson 1997). As a result, increased flows begin eroding the channel and a general deepening and widening of the channel (channel incision) occurs to accommodate the increased bankfull discharge (Hammer 1972, Douglas 1974, Roberts 1989, Booth 1990). Increased channel water velocities exceed minimum entrainment velocities for transporting bed materials, and readily moveable sediment is lost first as channels generally deepen (Leopold 1973, Morisawa & LaFlure 1979). Channels may actually narrow during this phase as entrained sediment from incision is deposited laterally in the channel (Dunne & Leopold 1978). After incision channels begin to migrate laterally, bank erosion begins, which leads to general channel widening (Booth 1990, Booth & Jackson 1997, Trimble 1997).

During the erosional phase channel enlargement can occur gradually if increases in width and depth keep pace with increases in discharge associated with increasing ISC. In this case the channel enlargement may be barely noticeable (Booth 1990). However, erosion more commonly occurs disproportionately to discharge changes, often leading to bank failure and catastrophic erosion in urban streams (Neller 1988, Booth 1990). In developed urban catchments, as a result of this erosional readjustment phase, the majority of sediment leaving the catchment comes from within-channel erosion as opposed to hillslope erosion (Trimble 1997). The magnitude of this generalized geomor-

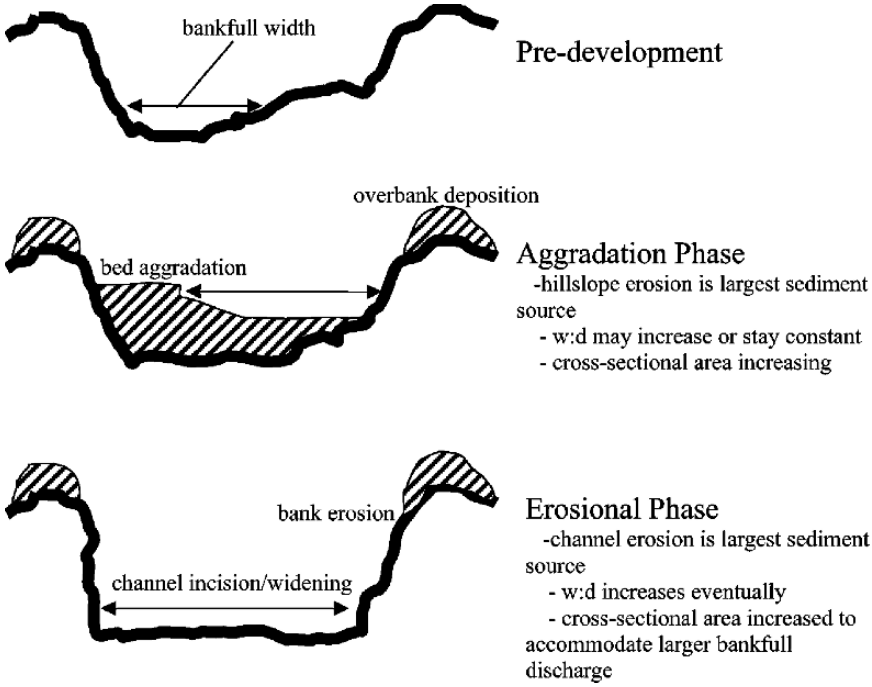


**Fig. 2** Channel changes associated with urbanization. During the construction phase of urbanization, hillslope erosion increases sediment supply leading to bed aggradation and overbank deposition. After construction ceases hillslope sediment supply is reduced, but bankfull flows are increased owing to increases in imperviousness. This leads to increased channel erosion as channel incision and widening occur to accommodate increased bankfull discharge

phic response will vary longitudinally along a stream network as well as with the age of development, catchment slope, geology, sediment characteristics, type of urbanization, and land use history (Gregory et al. 1992).

Urban streams differ in other geomorphic characteristics from forested catchments as well. The spacing between pool-riffle sequences (distance between riffles) is generally constant at 5–7 times channel width in forested catchments (Gregory et al. 1994). Generally, this ratio stays constant in urban channels as they widen, which means the absolute distance between pool-riffle units increases, although there is some evidence that this spacing may decrease to 3–5 times channel width (Gregory et al. 1994).

Changes in sediment supply may also alter channel pattern. Increased sediment supply during construction has converted some meandering streams to braided patterns or to straighter, more channelized patterns (Arnold et al. 1982). In the latter case, channelizing leads to increased slope and therefore higher in-stream velocities, especially where artificial channel alteration is carried out to increase the efficiency of the channel in transporting flows (Pizzuto et al. 2000).

Urbanization can also alter sediment texture. Less fine sediment, increased coarse sand fractions, and decreased gravel classes have been observed in urban channels as a result of alteration of sediment supply and altered velocities (Finkenbine et al. 2000, Pizzuto et al. 2000). In addition to sediment changes, large woody debris is also reduced in urban channels. Catchments in Vancouver, British Columbia with greater than 20% ISC generally have very little large woody debris, a structural element important in both the geomorphology and ecology of Pacific Northwest stream ecosystems (Finkenbine et al. 2000).

Other geomorphic changes of note in urban channels include erosion around bridges, which are generally more abundant as a result of increased road densities in urban channels (Douglas 1974).

Bridges have both upstream and downstream effects, including plunge pools created below bridge culverts that may serve as barriers to fish movement. Knickpoints are another common feature of urban channels. These readily erodeable points of sudden change in depth are created by channel erosion, dredging, or bridge construction and are transmitted throughout the catchment, causing channel destabilization (Neller 1988). Other features include increased tree collapse, hanging tributary junctions as a result of variable incision rates, and erosion around artificial structures (e.g., utility support pilings) (Roberts 1989).

Changes in the hydrology and geomorphology of streams likely affect the hydraulic environment of streams, altering, among other things, the velocity profiles and hyporheic/parafluvial dynamics of channels. Such changes would affect many ecological processes, from filter-feeding organisms (Hart & Finelli 1999) to carbon processing and nutrient cycling (Jones & Mulholland 2000).

### *Temperature*

Stream temperature is an important variable affecting many stream processes such as leaf decomposition (Webster & Benfield 1986) and invertebrate life history (Sweeney 1984). Urbanization affects many elements of importance to stream heat budgets. Removal of riparian vegetation, decreased groundwater recharge, and the "heat island" effect associated with urbanization, covered more fully in a companion review (Pickett et al. 2001), all affect stream temperature (Pluhowski 1970), yet very little published data exists on temperature responses of streams to urbanization. In one study on Long Island urban streams had mean summer temperatures 5–8°C warmer and winter temperatures 1.5–3°C cooler than forested streams. Seasonal diurnal fluctuations were also greater in urban streams, and summertime storms resulted in increased temperature pulses 10–15°C warmer than forested streams, a result of runoff from heated impervious surface (Pluhowski 1970). Similar effects on summer temperatures and daily fluctuations have also been observed elsewhere (Table 1) (Galli 1991, Leblanc et al. 1997).

## Chemical Effects of Urbanization

Chemical effects of urbanization are far more variable than hydrologic or geomorphic effects and depend on the extent and type of urbanization (residential versus commercial/industrial), presence of wastewater treatment plant (WWTP) effluent and/or combined sewer overflows (CSOs), and the extent of stormwater drainage. Overall, there are more data on water and sediment chemistry in urban streams than any other aspect of their ecology. This is aided by several very large national datasets of stream chemistry that focus in whole or in part on urbanization [e.g., National Urban Runoff Program (United States), National Water Quality Assessment Program (USGS 2001), Land-Ocean Interaction Study (UK) (Neal & Robson 2000)].

In general, there is an increase in almost all constituents, but consistently in oxygen demand, conductivity, suspended solids, ammonium, hydrocarbons, and metals, in urban streams (Porcella & Sorensen 1980, Lenat & Crawford 1994, Latimer & Quinn 1998, USGS 1999). These increases can be attributed to both WWTP effluent and non-point source (NPS) runoff. Many countries have accomplished significant reductions in chemical constituents as a result of adopting better WWTP technologies (e.g., Krug 1993, Litke 1999). However, treatment cannot remove all constituents from wastewater, treatment systems fail, and permitted discharge limits are exceeded. There are more than 200,000 discharges subject to permitting in the United States (USEPA 2001), and of 248 urban centers studied, 84% discharge into rivers (40% of those into rivers with mean annual discharges less than 28 $m^3/s$) (Heaney & Huber 1984). In addition, CSO systems are still common, in which stormwater and untreated sewage are combined and diverted to streams and rivers during storms. At least 28% of the urban centers mentioned above contained CSOs, and in the United Kingdom 35% of the annual pollutant discharge comes from CSOs and storm drains during less than 3% of the time (Heaney & Huber 1984, Faulkner et al. 2000). In addition, illicit discharge connections, leaking sewer systems, and failing septic systems are a large and persistent contributor of pollutants to urban streams (Faulkner et al. 2000). In the Rouge River catchment in Detroit, Michigan, the focus of an intense federal NPS management program, septic failure rates between 17% and 55% were reported from different subcatchments, and it was estimated that illicit untreated sewage discharge volume at more than 193,000 $m^3$/yr (Johnson et al. 1999). The ubiquitous nature of small, NPS problems in urban catchments has led some to suggest that the cumulative effect of these small problems may be the dominant source of biological degradation in urban catchments (Duda et al. 1982).

## *Nutrients and Other Ions*

Urbanization generally leads to higher phosphorus concentrations in urban catchments (Omernik 1976, Meybeck 1998, USGS 1999, Winter & Duthie 2000). An urban effect is most often seen in total phosphorus as a result of increased particle-associated phosphorus, but dissolved phosphorus levels are also increased (Smart et al. 1985). In some cases increases in phosphorus can even rival those seen in agricultural catchments both in terms of concentration and yield (Omernik 1976). Even an attempt to understand the agricultural contribution to catchment phosphorus dynamics in a midwestern catchment discovered that urbanization was a dominant factor (Osborne & Wiley 1988). Even though urban areas constituted only 5% of the catchment area and contributed only a small part to the total annual yield of dissolved phosphorus, urban land use controlled dissolved phosphorus concentration throughout the year.

Sources of phosphorus in urban catchments include wastewater and fertilizers (LaValle 1975). Lawns and streets were the primary source of phosphorus to urban streams in Madison, Wisconsin as a result of fertilizer application (Waschbusch et al. 1999). Soils are important in phosphorus dynamics, and the retention of groundwater phosphorus from septic fields affects stream phosphorus concentrations (Hoare 1984, Gerritse et al. 1995). Phosphorus stored in soils as a result of fertilization, however, can be mobilized by soil erosion and contribute to eutrophication of receiving waters. This effect has been called the "chemical time bomb" and is of particular concern when previously agricultural land is cleared for urban growth (Bennett et al. 1999).

Although phosphorus concentrations are elevated in urban streams, the effective increase is not as great as that observed for nitrogen. Urban centers have been shown to increase the nitrogen concentration in rivers for hundreds of kilometers (Meybeck 1998, USGS 1999). Increases have been observed for ammonium as well as nitrate (McConnell 1980, Hoare 1984, Zampella 1994, Wernick et al. 1998). The extent of the increase depends on wastewater treatment technology, degree of illicit discharge and leaky sewer lines, and fertilizer use. As with phosphorus, nitrogen concentrations in streams draining agricultural catchments are usually much higher (USGS 1999), but some have noticed similar or even greater levels of nitrogen loading from urbanization (Omernik 1976, Nagumo & Hatano 2000). Soil characteristics also affect the degree of nitrogen retention, of importance when on-site septic systems are prevalent (Hoare 1984, Gerritse et al. 1995).

Other ions are also generally elevated in urban streams, including calcium, sodium, potassium, and magnesium (McConnell 1980, Smart et al. 1985, Zampella 1994, Ometo et al. 2000). Chloride ions are elevated in urban streams, especially where sodium chloride is still used as the principal road deicing salt. A significant portion of the more than 100,000 tons of sodium chloride applied in metropolitan Toronto annually for deicing enters long-turnover groundwater pools and is released slowly, raising stream chloride concentrations throughout the year (Howard & Haynes 1993). The combined effect of heightened ion concentrations in streams is the elevated conductivity observed in most urban streams. The effect is so common that some have suggested using chloride concentration or conductivity as general urban impact indicators (Wang & Yin 1997, Herlihy et al. 1998).

## Metals

Another common feature of urban streams is elevated water column and sediment metal concentrations (Bryan 1974, Wilber & Hunter 1977, Neal et al. 1997, Horowitz et al. 1999, Neal & Robson 2000). The most common metals found include lead, zinc, chromium, copper, manganese, nickel, and cadmium (Wilber & Hunter 1979), although lead has declined in some urban river systems since its elimination as a gas additive (Frick et al. 1998). Mercury is also elevated in some urban streams, and particle-bound methyl-mercury can be high during stormflow (Mason & Sullivan 1998, Horowitz et al. 1999). In addition to industrial discharges, there are many NPSs of these metals in urban catchments: brake linings contain nickel, chromium, lead, and copper; tires contain zinc, lead, chromium, copper, and nickel; and metal alloys used for engine parts contain nickel, chromium, copper, and manganese among others (Muschak 1990, Mielke et al. 2000). All of these metals accumulate on roads and parking lots (Sartor et al. 1974, Forman & Alexander 1998). Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Khamer et al. 2000, Neal & Robson 2000). Not surprisingly, it appears that NPSs of metals are more important than point sources in urban streams (Wilber & Hunter 1977, Mason & Sullivan 1998).

The concentration, storage, and transport of metals in urban streams is connected to particulate organic matter content and sediment characteristics (Tada & Suzuki 1982, Rhoads & Cahill 1999). Organic matter has a high binding capacity for metals, and both bed and suspended sediments with high organic matter content frequently exhibit 50–7500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content (Warren & Zimmerman 1994, Mason & Sullivan 1998, Gonzales et al. 2000). Sediment texture is also important, and metal concentration in sediments was inversely correlated to sediment particle size in several urban New Jersey streams (Wilber & Hunter 1979). In addition, geomorphic features have been shown to influence metal accumulations. Higher sediment metal concentrations were found in areas of low velocity (stagnant zones, bars, etc.) where fine sediments and organic particles accumulate, whereas areas of intermediate velocities promoted the accumulation of sand-sized metal particles, which can also be common in urban streams (Rhoads & Cahill 1999).

Several organisms (including algae, mollusks, arthropods, and annelids) have exhibited elevated metal concentrations in urban streams (Davis & George 1987, Rauch & Morrison 1999, Gundacker 2000), and ecological responses to metals include reduced abundances and altered community structure (Rauch & Morrison 1999). It is important to note that the route of entry appears to be both direct exposure to dissolved metals and ingestion of metals associated with fine sediments and organic matter. This has led a few researchers to suggest that metal toxicity is most strongly exerted through the riverbed rather than the overlying water (Medeiros et al. 1983, House et al. 1993), although only dissolved metal concentrations in the water column are regulated in the United States.

## Pesticides

Pesticide detection frequency is high in urban streams and at concentrations frequently exceeding guidelines for the protection of aquatic biota (USGS 1999, Hoffman et al. 2000). These pesticides include insecticides, herbicides, and fungicides (Daniels et al. 2000). In addition, the frequent detection of banned substances such as DDT and other organochlorine pesticides (chlordane and dieldrin) in urban streams remains a concern (USGS 1999). Most surprising is that many organochlorine pesticide concentrations in urban sediments and biota frequently exceed those observed in intensive agricultural areas in the United States (USGS 1999), a phenomenon observed in France as well (Chevreuiel et al. 1999). Additionally, it is estimated that the mass of insecticides contributed by urban areas is similar to that from agricultural areas in the United States (Hoffman et al. 2000).

There are many sources of pesticides in urban catchments. Urban use accounts for more than 136,000 kg, which is a third of U.S. pesticide use (LeVeen & Willey 1983). They are frequently applied around homes (70–97% of U.S. homes use pesticides) and commercial/industrial buildings and are intensively used in lawn and golf course management (LeVeen & Willey 1983, USGS 1999). Areal application rates in urban environments frequently exceed those in agricultural applications by nearly an order of magnitude (Schueler 1994b). For example, pesticide application rates on golf courses (including herbicides, insecticides, and fungicides) exceed 35 pounds/acre/year, whereas corn/soybean rotations receive less than 6 pounds/acre/year (Schueler 1994b). However, unlike agricultural use, urban pesticide application rates are generally not well documented (LeVeen & Willey 1983, Coupe et al. 2000).

As with metals, the main vector of transport of pesticides into urban streams appears to be through NPS runoff rather than WWTP effluent (Foster et al. 2000). A strong correlation between particle concentration and pesticide concentration was found in the Anacostia River basin in Maryland and the San Joaquin River in California, suggesting NPS inputs are most important (Pereira et al. 1996, Foster et al. 2000). Volatilization and aerosol formation contributed to higher pesticide concentrations, including atrazine, diazinon, chlorpyrifos, p,p′-DDE (a DDT metabolite), and other organochlorines, in precipitation in urban areas and may contribute directly to greater pesticide concentrations and yields in urban areas (Weibel et al. 1966, Coupe et al. 2000).

## *Other Organic Contaminants*

A whole suite of other organic contaminants are frequently detected in urban streams, including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and petroleum-based aliphatic hydrocarbons (Whipple & Hunter 1979, Moring & Rose 1997, Frick et al. 1998). PCBs are still frequently detected in urban areas of the United States, even though their use in manufacturing was outlawed because of their carcinogenic effects. These compounds are very stable and are still found in fish at concentrations exceeding consumption-level guidelines in urban rivers such as the Chattahoochee River below Atlanta, Georgia (Frick et al. 1998). PCB concentrations were highly correlated with urban land use in the Willamette Basin in Oregon as well (Black et al. 2000). As with metals and pesticides, PCBs are primarily particle associated, and in the absence of industrial point sources, it is assumed that stormwater runoff is the major route of entry (Foster et al. 2000).

PAHs are a large class of organic compounds that include natural aromatic hydrocarbons but also many synthetic hydrocarbons including organic solvents with different industrial uses (Yamamoto et al. 1997). For this reason, the unnatural PAHs are probably derived from industrial effluent or episodic spills. Very little is known about these compounds in urban streams. In Dallas–Fort Worth, Texas streams, 24 different industrial PAHs were detected, including 4 of the top 10 U. S. Environmental Protection Agency (EPA) most hazardous substances, and at concentrations exceeding human health criteria (Moring & Rose 1997). In Osaka, Japan streams, 55 PAHs were detected, including 40 EPA target compounds. Organic solvents (e.g., toluene, trichloroethane, and dichlorethane) were most common (Yamamoto et al. 1997).

It is difficult to find automobile parking spaces without oil stains in any city. The result of these leaky crankcases is a cornucopia of different petroleum-based aliphatic hydrocarbons in storm runoff associated primarily with particles (Whipple & Hunter 1979). Although there are natural aliphatic hydrocarbons in streams, these are generally overwhelmed by petroleum-based compounds in urban stream bed and water-column sediments (Hunter et al. 1979, Mackenzie & Hunter 1979, Eganhouse et al. 1981). Evidence suggests that these are frequently at concentrations that are stressful to sensitive stream organisms (Latimer & Quinn 1998). Most striking is the yield of these compounds from urban catchments. An estimated 485,000 liters of oil enters the Narragansett Bay each year, a volume equal to nearly 50% of the disastrous 1989 *World Prodigy* oil spill in that same bay (Hoffman et al. 1982, Latimer & Quinn 1998). Similarly, it is estimated that the Los Angeles River alone contributes about 1% of the annual world petroleum hydrocarbon input to the ocean (Eganhouse et al. 1981).

Lastly, recent data suggest pharmaceutical substances from hospital effluent may contribute an array of different chemical compounds into streams. Detectable levels of antiobiotics, genotoxic chemotherapeutic drugs, analgesics, narcotics, and psychotherapeutic drugs have been reported from effluent and/or surface waters (Halling-Sorensen et al. 1998). Although there is some information on the toxicity of these different compounds from laboratory studies, there are insufficient data on the nature or extent of the threat they pose to urban stream biota.

## Biological and Ecological Effects of Urbanization

The ecological implications of urbanization are far less studied than the chemical effects, an absence noted in several studies (Porcella & Sorensen 1980, Duda et al. 1982, Medeiros et al. 1983). Nevertheless, much is known about the response of stream organisms, especially invertebrates, to urbanization; far less is known about urban effects on fish (Mulholland & Lenat 1992). Of even greater concern is the lack of mechanistic studies; few studies analyze whether physical habitat, water quality, or food web disturbances (either resource effects or altered community interactions) are the cause of biological degradation in urban streams (Suren 2000). Grossly underrepresented are studies of population dynamics, community interactions, and ecosystem ecology of urban streams, which is surprising given the level of knowledge within the field (Allan 1995). Lastly, very little information has been gathered on biological monitoring of restoration or best management practice implementation in urban catchments (Riley 1998). Most studies assess performance based on stream channel condition or pollutant reduction; few, if any, monitor biological response (Benke et al. 1981, Center for Watershed Protection 2000). In this section, we discuss the effects of urbanization on microbes, algae, macrophytes, invertebrates, and fish.

### *Microbes*

Bacterial densities are usually higher in urban streams, especially after storms (Porcella & Sorensen 1980, Duda et al. 1982). Much of this is attributable to increased coliform bacteria, especially in catchments with wastewater treatment plant (WWTP) and combined sewer overflow (CSO) effluent (Gibson et al. 1998, Young & Thackston 1999). In Saw Mill Run, an urban stream near Pittsburgh, Pennsylvania, fecal coliform colony–forming units (CFU) increased from 170–13,300 CFU/100 ml during dry weather to 6,100–127,000 CFU/100 ml during wet weather (Gibson et al. 1998). CSOs contributed 3,000–85,000 CFU/100 ml during wet weather. These data indicate that non–point sources (NPSs) as well as point sources contribute to fecal coliform loads in urban streams. High values during dry weather are not uncommon in urban streams and may indicate chronic sewer leak-age or illicit discharges. Storm sewers were also a significant source of coliform bacteria in Vancouver, British Columbia; stormwater there contained both human and nonhuman fecal coliform bacteria (Nix et al. 1994). Other pathogens, including *Cryptosporidum* and *Giardia*, have also been associated with CSOs (Gibson et al. 1998).

Increased antibiotic resistance has been seen in some urban bacterial populations (Goni-Urriza et al. 2000). Increased resistance to several antiobiotics, including nalidixic acid, tetracycline, beta-lactam, and co-trimoxazole, has been observed from several enteric as well as native stream species isolated from a river downstream of a WWTP discharge in Spain. It may be that resistant bacteria are passing through the treatment process and conferring resistance to native bacteria. Recent evidence suggests that metal toxicity may also be indirectly involved in increasing antibiotic resistance in stream bacteria. Bacterial resistance to streptomycin and kanamycin were positively correlated with sediment mercury concentration in streams below nuclear reactors and industrial facilities, a result of indirect selection for metal tolerance (McArthur & Tuckfield 2000). Metals may also affect bacterial enzyme activity in urban streams. Enzyme levels were inversely correlated to sediment metal concentration in an urban stream, and this was especially pronounced below an industrial effluent (Wei & Morrison 1992).

Nitrifying bacteria, responsible for the oxidation of reduced nitrogen, are also influenced by urbanization. WWTP effluent can represent a significant source of nitrifying bacteria to urban streams (Brion & Billen 2000). These bacteria are used to oxidize ammonium during the treatment process, but escape into streams in effluent and contribute to the high nitrifier activity observed below some WWTP discharges (Jancarkova et al. 1997). Nitrification rates were as much as six times higher in treated effluent entering the Seine than in receiving river water upstream (Brion & Billen 2000). Ironically, because so many nitrifiers entered the Seine River in France via untreated sewage historically, the reduction in untreated sewage via improved sewage design contributed to a reduction in ammonium oxidation rates in the river from 1.5 $\mu$mol/liter/h in 1976 to 1.0 $\mu$mol/liter/h in 1993 (Brion & Billen 2000). In addition to nitrifiers, iron-oxidizing bacteria are often abundant in urban streams, especially where reduced metals emerge from anoxic urban groundwater or storm sewers (Dickman & Rygiel 1998).

## *Algae*

The use of algae to indicate water quality in Europe and the United States has a long history (Kolkwitz & Marsson 1908, Patrick 1973). As a result, information exists on algal species and community responses to organic pollution; however, the response of algae to all aspects of urbanization is far less studied. The increasing proportion of urban land use in a catchment generally decreases algal species diversity, and this change has been attributed to many factors including water chemistry (Chessman et al. 1999). Elevated nutrients and light levels typically favor greater algal biomass, which has been observed in many urban streams, where algae do not appear to be nutrient limited (Chessman et al. 1992, Richards & Host 1994). However, the shifting nature of bed sediment in urban streams, frequent bed disturbance, and high turbidity may limit algal accumulation (Burkholder 1996, Dodds & Welch 2000). In addition, several algal species are sensitive to metals, and stream sediment metal accumulation can result in reduced algal biomass (Olguin et al. 2000). Lastly, the frequent detection of herbicides in streams, some with known effects on algae (Davies et al. 1994), will undoubtedly affect stream algal communities

## *Macrophytes*

Little has been written on macrophyte response to urbanization. Most of the work has been done in New Zealand and Australia, where bed sediment changes, nutrient enrichment, and turbidity all contribute to reduced diversity of stream macrophytes (Suren 2000). Exotic species introductions in urban streams have also resulted in highly reduced native macrophyte diversity (Arthington 1985, Suren 2000). Excessive macrophyte growth as a result of urbanization has not been observed in New Zealand, even though nutrient and light levels are higher (Suren 2000).

## *Invertebrates*

Literature searches revealed more studies of urban effects on aquatic invertebrates than on any other group, and the available data are being expanded by groups biomonitoring urban systems (e.g., USGS National Water Quality Assessment, U.S. EPA, state agencies, and others). All aspects of aquatic invertebrate habitat are altered by urbanization. One of the historically well-studied aspects has been the effects of organic pollutants (especially WWTP effluent) on invertebrates. Organic pollution generally reduces invertebrate diversity dramatically, resulting in a community dominated by Chironomidae (Diptera) and oligochaetes (Campbell 1978, Seager & Abrahams 1990, Wright et al. 1995). However, general effects of urbanization on stream invertebrates have also been studied and general invertebrate responses can be summarized as follows: decreased diversity in response to toxins, temperature change, siltation, and organic nutrients; decreased abundances in response to toxins and siltation; and increased abundances in response to inorganic and organic nutrients (Resh & Grodhaus 1983, Wiederholm 1984).

Studies of the effects of urban land use on invertebrates can be divided into three types: those looking along a gradient of increasing urbanization in one catchment, those looking at an urbanized versus a reference catchment, and large studies considering urban gradients and invertebrate response in several catchments. All single catchment gradient studies find a decrease in invertebrate diversity as urban land use increases, regardless of the size of the catchment (Pratt et al. 1981, Whiting & Clifford 1983, Shutes 1984, Hachmoller et al. 1991, Thorne et al. 2000). Decreases were especially evident in the sensitive orders—Ephemeroptera, Plecoptera, and Trichoptera (Pratt et al. 1981, Hachmoller et al. 1991). Most of these studies observed decreases in overall invertebrate abundance, whereas the relative abundance of Chironomidae, oligochaetes, and even tolerant gastropods increased (Pratt et al. 1981, Thorne et al. 2000). Comparative catchment studies show the same trends with increasing urbanization as those observed in single catchment studies: decreased diversity and overall abundance and increased relative abundance of tolerant Chironomidae and oligochaetes (Medeiros et al. 1983, Garie & McIntosh 1986, Pederson & Perkins 1986, Lenat & Crawford 1994).

The multi-catchment studies attempt to relate differing amounts of urbanization in many catchments to particular invertebrate community responses, often using a gradient analysis approach. As discussed above, all find decreases in diversity and overall invertebrate abundance with increased urbanization. This response is correlated with impervious surface cover, housing density, human population density, and total effluent discharge (Klein 1979, Benke et al. 1981, Jones & Clark 1987, Tate & Heiny 1995, Kennen 1999). Klein (1979) studied 27 small catchments on the Maryland Piedmont and was among the first to identify impervious surface cover (ISC) as an important indicator of degradation. Invertebrate measures declined significantly with increasing ISC until they indicated maximum degradation at 17% ISC (Table 1). Degradation thresholds at ISC between 10 and 20% have been supported by numerous other studies for many different response variables (see Schueler 1994a). Residential urbanization in Atlanta, Georgia had dramatic effects on invertebrate diversity, but there were very few clues as to the mechanisms responsible, although leaky sewers were implicated in these and other urban residential catchments (Benke et al. 1981, Johnson et al. 1999).

Few studies have considered specific mechanisms leading to the observed effects of urbanization. This is a difficult task because of the multivariate nature of urban disturbance. Increased turbidity has been associated with higher drift densities of insects (Doeg & Milledge 1991), but more work has focused on the instability of smaller and more mobile bed sediments associated with urban sedimentation. In general, the change in bed sediments favors species adapted to unstable habitats, such as the chironomid dipterans and oligochaete annelids (Pedersen & Perkins 1986, Collier 1995). Where slopes are steeper, and smaller sediments are removed by increased water velocities, localized areas of higher invertebrate diversity are observed within the coarser sediments (Collier 1995). Pools are particularly affected by sediment accumulation in urban streams, and invertebrate communities within these habitats are degraded (Hogg & Norris 1991). Lastly, sedimentation associated with urban streams reduces available refugial space, and invertebrates are more susceptible to drift when refugial space is limited during the frequent floods characteristic of urban environments (Borchardt & Statzner 1990). Storm-flows in urban streams introduce the majority of pollutants and also move the bed sediment frequently. The mortality of *Pteronarcys dorsata* (Plecoptera) in cages in urban streams was attributed to sedimentation associated with storms (Pesacreta 1997).

Sediment toxicity has also been explored. As mentioned above, benthic organic matter binds many toxins and is also a major food resource for many stream invertebrates (Benke & Wallace 1997). Mortality of aquatic invertebrates remains high in many urban streams even during low flow periods, suggesting that toxicity associated with either exposure in the bed or ingestion of toxins associated with organic matter contributes to invertebrate loss (Pratt et al. 1981, Medeiros et al. 1983).

Riparian deforestation associated with urbanization reduces food availability, affects stream temperature, and disrupts sediment, nutrient, and toxin uptake from surface runoff. Invertebrate bioassessment metrics decreased sharply in Puget Sound, Washington tributaries with increasing ISC (Horner et al. 1997). However, streams that had higher benthic index of biotic integrity scores for a given level of ISC were always associated with greater riparian forest cover in their catchment, suggesting that riparian zones in some urban catchments may buffer streams from urban impacts. Above 45% ISC, all streams were degraded, regardless of riparian status. The value of riparian forests is also reduced if the stormwater system is designed to bypass them and discharge directly into the stream.

Road construction associated with urbanization impacts stream invertebrates. Long-term reductions (>6 y) in invertebrate diversity and abundances were observed in association with a road construction project in Ontario (Taylor & Roff 1986). General effects of roads on streams has been reviewed recently (Forman & Alexander 1998).

Very little ecological data beyond presence/absence or abundance data have been reported for urban stream invertebrates. Aquatic insect colonization potential was reported to be high in some urban streams, suggesting restoration efforts would not be limited in this regard (Pedersen & Perkins 1986), but little is known about colonization or adult aquatic insect ecology in urban streams. Urban stream restoration work focuses largely on channel geomorphological stability, with relatively little attention given to biological restoration (Riley 1998), although restoration of Strawberry Creek on the campus of the University of California at Berkeley has resulted in detectable increases in invertebrate diversity and abundance (Charbonneau & Resh 1992). Drift of aquatic invertebrates is a well studied phenomenon in streams, but with one exception (Borchardt & Statzner 1990), little has been published on insect drift in urban streams. We found no published work regarding life cycle ecology (e.g., voltinism or emergence timing), population dynamics, behavioral ecology, community interactions, or production of aquatic invertebrates in urban streams.

## *Fish*

Less is known about fish responses to urbanization than about invertebrates, and a general response model does not exist. However, the Ohio Environmental Protection Agency has a very large database of land use and fish abundance from around their state and has suggested three levels of general fish response to increasing urbanization: from 0 to 5% urban land use, sensitive species are lost; from 5 to 15%, habitat degradation occurs and functional feeding groups (e.g., benthic invertivores) are lost; and above 15% urban land use, toxicity and organic enrichment result in severe degradation of the fish fauna (Table 1) (Yoder et al. 1999). This model has not been verified for other regions of the country, where studies have focused on various aspects of urbanization. Here we consider three types of urban land use studies with regards to fish: gradients of increasing urbanization within a single catchment, comparing an urban and reference catchment, and large, multi-catchment urban gradient studies.

Along urban gradients within single catchments, fish diversity and abundances decline, and the relative abundance of tolerant taxa increases with increasing urbanization (Table 1) (Onorato et al. 2000, Boet et al. 1999, Gafny et al. 2000). Invasive species were also observed to increase in more urbanized reaches of the Seine River, France, and this effect extended more than 100 km below Paris (Boet et al. 1999). Summer storms in that river were associated with large fish kills as a result of dissolved oxygen deficits, an effect also observed for winter floods in Yargon Stream, the largest urban stream in Israel (Gafny et al. 2000). Comparisons with historical collections, an approach used commonly with fish studies, revealed that several sensitive species were extirpated from the Upper Cahaba River system in Alabama between 1954 and 1995, a period coinciding with the rapid growth of Birmingham, Alabama (Onorato et al. 2000). Extirpation of fish species is not uncommon in urban river systems (Ragan & Dietmann 1976, Weaver & Garman 1994, Wolter et al. 2000).

Comparative catchment studies also find dramatic declines in fish diversity and abundances in urban catchments compared with forested references (Scott et al. 1986, Weaver & Garman 1994, Lenat & Crawford 1994). Kelsey Creek, a well-studied urban stream in Washington, is unusual in that it has sustained salmonid populations, especially cutthroat trout (*Oncorhynchus clarki*), even though coho salmon (*Oncorhyncus kisutch*) and many nonsalmonid species have disappeared (Scott et al. 1986). Salmonids in the urban stream actually grow more rapidly and to larger sizes, increasing fish production up to three times that in the forested reference site, presumably a result of warmer temperatures and greater invertebrate biomass in the urban stream. However, the population size structure is different in the two streams, with year 0 and 1 cutthroat underrepresented in the urban stream (Scott et al. 1986).

Large multi-site studies of fish responses to urban gradients also find dramatic decreases in diversity or fish multimetric indices [index of biotic integrity (IBI)] with increasing ISC or other urban land use indicators (Table 1) (Klein 1979, Steedman 1988, Wang et al. 1997, Frick et al. 1998, Yoder et al. 1999). Similar to effects observed for invertebrates, these studies also find precipitous declines in fish metrics between 0 and 15% ISC or urban land use, beyond which fish communities remain degraded (Klein 1979, Yoder et al. 1999). The effect of urbanization on fish appears at lower percent land area disturbed than effects associated with agriculture. In Wisconsin and Michigan few fish community effects were observed in agricultural catchments up to 50% agricultural land use in the catchment (Roth et al. 1996, Wang et al. 1997), and mixed agriculture and urban catchments had significantly lower IBI scores than strictly agricultural catchments (Wang et al. 2000). This suggests that although total urban land use occupies a smaller area globally, it is having disproportionately large effects on biota when compared with agriculture. However, it is crucial to recognize that all urban growth does not have the same effects. Extensive fish surveys in Ohio suggest that residential development, especially large-lot residential development, has less of an effect on stream fishes than high-density residential or commercial/industrial development (Yoder et al. 1999). They hypothesize that riparian protection and less channel habitat degradation are responsible for protecting the fauna in these streams, even up to 15% urban land use. Similar benefits of riparian forests to fish in urban streams were observed in the Pacific Northwest (Horner et al. 1997).

Few studies have explored specific mechanisms causing changes in fish assemblages with urbanization. Sediment is presumably having effects on fish in urban streams similar to those observed in other systems although toxin-mediated impacts may be greater (Wood & Armitage 1997). Road construction results in an increase in the relative abundance of water-column feeders as opposed to benthic feeders, likely a response to a decrease in benthic invertebrate densities (Taylor & Roff 1986). Benthic feeders quickly reappeared as sedimentation rates declined after construction. Flow modification associated with urbanization also affects stream fish. In the Seine, modification of flow for flood protection and water availability has affected pike (*Esox lucius*) by reducing the number of flows providing suitable spawning habitat. With urbanization, the river contains enough suitable spawning habitat in only 1 out of 5 years as opposed to 1 out of every 2 years historically (Boet et al. 1999). Last, WWTP effluent clearly affects fishes. Reductions in WWTP effluent have been associated with the recovery of the fish community in a River Trent tributary near Birmingham, England (Harkness 1982). After nearly 250 years of degradation, effluent reductions, improved treatment, and construction of run-of-the river purification have resulted in an increase in fish diversity and abundances.

A few studies have actually examined ecological factors regulating stream fish populations and communities in urban streams. Recruitment of anadromous fish in the Hudson River Basin in New York was limited by suitable spawning habitat as a result of urbanization (Limburg & Schmidt 1990). Numbers of alewife (*Alosa pseudoharengus*) eggs and larvae in tributary streams decreased sharply between 0 and 15% urban land use. Beyond 15%, no eggs or larvae were found. The Kelsey Creek study discussed above showed impacts on salmonid population structure associated with urbanization, suggesting that urban streams may serve as population sinks for cutthroat, and that fish populations in those streams are dependent on recruitment from source populations with normal population age structures (Scott et al. 1986). Few data on the diet of fish in urban streams have been published, although a shift in diet was observed for fish along an urban gradient in Virginia (Weaver & Garman 1994).

Introduced fish species are also a common feature of urban streams. As a result of channelization, other river transportation modifications, and voluntary fisheries efforts in the Seine around Paris, 19 exotic species have been introduced, while 7 of 27 native species have been extirpated (Boet et al. 1999). The red shiner (*Cyprinella lutrensis*), a Mississippi drainage species commonly used as a bait fish, has invaded urban tributaries of the Chattahoochee River in Atlanta, Georgia where it has displaced native species and now comprises up to 90% of the fish community (DeVivo 1995).

As observed above for invertebrates, real gaps exist in our understanding of fish ecology in urban streams. The effects of urbanization on fishes have focused primarily on patterns of species presence, absence, or relative abundance. We found no published information on behavioral ecology, community interactions, or the biomass and production of nonsalmonid fishes in urban streams.

## *Ecosystem Processes*

Ecosystem processes such as primary productivity, leaf decomposition, or nutrient cycling have been overlooked in urban streams, although they have been extensively studied in other types of stream ecosystems (Allan 1995). A few studies have considered organic matter in streams. WWTP effluent and CSO discharges can dramatically increase dissolved and particulate organic carbon concentrations, especially during storms (McConnell 1980). However, much less is known about baseflow concentrations of particulate and dissolved carbon in urban streams—natural or anthropogenic. The carbon inputs associated with sewage are generally more labile than natural transported organic matter and they affect dissolved oxygen in streams. Oxygen deficits associated with high biological oxygen demand during and after storms are common (McConnell 1980, Faulkner et al. 2000, Ometo et al. 2000). In addition, nonrespiratory oxygen demands associated with chemical oxidation reactions are also elevated in urban streams and can be much higher than biological oxygen demand in stormwater runoff (Bryan 1972). These inputs explain in part why more than 40% of 104 urban streams studied in the United States showed a high probability of greater than average oxygen deficits, with dissolved oxygen concentrations below 2 mg/liter and daily fluctuations up to 7 mg/liter not uncommon (Keefer et al. 1979). In a comparison of 2 forested and 4 urban catchments, average organic matter standing stocks were significantly lower in urban streams near Atlanta, Georgia (Paul 1999). This was attributed to greater scouring of the highly mobile sandy substrates in urban channels as a result of more severe flows.

Organic matter quality has been characterized in a few urban streams. In Kelsey Creek, particulate organic matter (POM) carbohydrate concentrations were higher than in POM in a nearby forested reference stream, suggesting that urbanization affects the nature of transported organic matter as well (Sloane-Richey et al. 1981). In addition to differences in organic matter quantity and quality, urban streams also differ in organic matter retention. Coarse and fine particles released to measure organic matter transport in Atlanta, Georgia streams traveled much farther before leaving the water column in urban streams than in forested streams (Paul 1999). Combined with the data from benthic organic matter (BOM) storage, these data indicate that these urban streams retain less organic matter, a fact that could limit secondary production in these urban streams (Paul 1999).

Ecosystem metabolism has also been measured in a few urban streams. In a comparison of three rivers in Michigan the urban river had higher gross primary production and community respiration than the forested river (Ball et al. 1973). In addition, the gross primary productivity to community respiration (P/R) ratio in the urban river without municipal effluent was greater than the forested stream and greater than 1.0, indicating that autotrophy dominated organic matter metabolism. However, in a downstream reach of the urban river receiving effluent, respiration was higher and the P/R ratio less than the forested river and far less than 1.0, indicating that heterotrophic metabolism predominated. Similar results were observed for urban streams in Atlanta, where gross primary production and community respiration were higher in urban streams than forested streams, and urban streams had more negative net ecosystem metabolism (gross primary production–community respiration), indicating greater heterotrophy (Paul 1999). However, because carbon storage was far less in the urban streams, carbon turnover was faster, supporting the hypothesis that respiration in urban streams was driven by more labile sources of carbon, such as sewage effluent.

Decomposition of organic matter has been measured in a few urban streams. Willow leaves decayed much faster in two suburban New Zealand streams than ever reported for any other stream; this occurred regardless of whether shredding insects were present or absent (Collier & Winterbourn 1986). The same results were observed for chalk maple (*Acer barbatum*) decay in urban streams in Atlanta, where rates were far faster in urban streams than rates observed for any woody leaf species in any stream (Paul 1999). Fungal colonization of leaves was only slightly lower in the urban streams, but there were no shredding insects associated with packs. These results suggested that higher stormflow was responsible for greater fragmentation of leaves in the urban streams, resulting in faster decay rates (Paul 1999).

Removal of added nutrients and contaminants is an ecological service provided by streams and relied upon by society. Although nutrient uptake in flowing waters has been extensively studied in forested ecosystems (Meyer et al. 1988, Stream Solute Workshop 1990, Marti & Sabater 1996), urban settings have been largely ignored. Studies in enriched reaches of river below the effluent from wastewater treatment plants have provided opportunities to examine patterns of denitrification in rivers (e.g., Hill 1979) and seasonal patterns of phosphorus removal and retention in a eutrophic river (e.g., Meals et al. 1999). Recently, ecologists have used the nutrients added by a wastewater treatment plant to measure nutrient uptake length, which is the average distance downstream traveled by a nutrient molecule before it is removed from the water column (Marti et al. 2001, Pollock

& Meyer 2001). Uptake lengths in these rivers are much longer than in nonurban rivers of similar size, suggesting that not only is nutrient loading elevated in urban streams, but also nutrient removal efficiency is greatly reduced. The net result of these alterations in urban streams is increased nutrient loading to downstream lakes, reservoirs, and estuaries.

## Opportunities and Imperatives for an Ecology of Urban Streams

Urban streams are common features of the modern landscape that have received inadequate ecological attention. That is unfortunate because they offer a fertile testing ground for ecological concepts. For example, hydrologic regime is a master variable in streams (Minshall 1988), influencing channel form, biological assemblages, and ecosystem processes. As discussed in this review, impervious surfaces result in characteristically altered and often extreme hydrologic conditions that provide an endpoint on a disturbance gradient and that offer opportunities to quantify the relationships between channel form, biological communities, and ecosystem processes (Meyer et al. 1988). Does a continuous gradient of impervious surface cover result in a similar gradient of ecological pattern and process or are there thresholds? Answering that question is of both theoretical and practical interest. Developing a mechanistic understanding of the linkages between urbanization and stream ecosystem degradation is elusive but essential if ecologists hope to understand the nature of ecological response to disturbance and if they want to contribute to the development of scenarios that can guide planning decisions.

Many urban centers developed around rivers, which were the lifeblood of commerce. These commercial uses of rivers ignored and degraded the ecological services rivers provide, a phenomenon continuing today as urban sprawl accelerates. Despite widespread degradation, urban rivers and streams offer local communities an easily accessible piece of nature. Most people live in urban areas, and many children first encounter nature playing in urban streams. Hence, urban streams offer opportunities for ecological outreach and education that ecologists are only beginning to explore. The meteoric rise in numbers of local catchment associations and adopt-a-stream monitoring groups is testimony to an audience eager for ecological insights.

Urban streams also offer ecologists an opportunity to test concepts of system organization through restoration projects. The field of urban stream restoration is dominated by physical scientists and engineers and rarely extends beyond stormwater management and bank stabilization with a goal of reestablishing a channel geomorphology in dynamic equilibrium with the landscape (e.g., Riley 1998). Little attention is given to restoration of a native stream biota or the ecological services streams provide. Urban stream restoration offers challenges not only in integrating physical, chemical, and biological processes to rehabilitate impaired ecosystems, but also requires an attention to esthetics and human attitudes toward the landscape. This offers an opportunity for the integration of ecological and social sciences with landscape design, which if successful will provide an avenue for ecologists to participate in the creation of the sustainable metropolitan centers of the future.

Cities have been a part of human history for millenia, and projections suggest most humans will live in cities in the future. Hence, urban areas lie at the intersection of human and ecological systems. If we are to succeed in that often-stated goal of incorporating humans as components of ecosystems, cities and their streams can no longer be ignored.

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