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Water Quantity and Quality at the Urban–Rural Interface

Ge Sun and B. Graeme Lockaby

Abstract

Population growth and urban development dramatically alter natural watershed ecosystem structure and functions and stress water resources. We review studies on the impacts of urbanization on hydrologic and biogeochemical processes underlying stream water quantity and water quality issues, as well as water supply challenges in an urban environment. We conclude that converting forest lands to urban uses increases stormflow rates and volumes, alters baseflow dynamics, and degrades water quality by increasing impervious surface areas. Alterations of watershed water cycles are the root causes of many chain reactions of stream ecosystem degradation present in today's urban areas. Knowledge gaps exist regarding interactions among processes of urbanization (land conversion, increasing impervious areas, new pollutants), hydrological functions (water budget change, infiltration and evapotranspiration processes), and ecological (biota change) functions at different temporal and spatial scales. Innovative implementation of watershed services is the key to mitigating impacts of urbanization on water and sustaining urban–rural ecosystems.

Although human populations have lived a rural lifestyle throughout most of our history, the world's urban population is rising rapidly and this change has caused serious problems that have impacted human welfare. For the United States, only 5% of the population could be classified as urban in 1790, but today 80% of the population lives in urban areas. Worldwide, about one-half the total population lives in urban areas, and this number is expected to grow to 60% by the year 2025.

People are attracted to water, and in turn human activities have affected the quantity, quality, distribution of waters on Earth. Anthropogenic structures such as irrigation canals, wells, reservoirs for drinking water withdrawal, dams for power generation, and paved roads for transportation are just a few examples of how humans have shaped the natural landscape. In the 21st century, as human population rises, it becomes increasingly important to understand the impacts of "urban sprawl" at the urban–rural interface on ecosystem structure and functions, society, and culture (Foley et al., 2005).

The most direct impact of urbanization on ecosystems is altering the hydrologic cycle that controls the ecosystem energy and matter flows (DeFries and Eshleman, 2004). Indeed, water resources in urban environments around the world are increasingly stressed due to population rise, rapid land use change (Foley et al., 2005; Piao et al., 2007), and climate

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change and variability (McCray and Boving, 2007; Sun et al., 2008). In many parts of the world, water availability has severely limited environmental, social, and economic development (Vörösmarty et al., 2000; Falkenmark et al., 2007; McDonald et al., 2011). Water stress is especially problematic in population centers, and fast-growing population centers (Oki and Kanae, 2006) in particular, where water demands are high and water quality is generally low. Water has been listed as one the top issues facing land conservation and management in the United States (Fleishman et al., 2011).

In this chapter, we address water issues in the urban–rural interface by examining the interactions among water, climate, vegetation, and urbanization at the landscape and watershed scales. We view urbanization as the transformation from rural land uses for agriculture or forestry to urban land uses that is characterized by high population density and a large extent of impervious surface. We approach water issues from a watershed ecosystem point of view with an understanding that physical, biological, chemical, and socioeconomic processes are linked at the watershed scale (Paul and Meyer, 2001; Pickett et al., 2001; McCray and Boving, 2007). We first define water quantity (i.e., stormflow, peakflow, baseflow, total water yield), and water quality (i.e., nonpoint source pollutant concentrations and loads) in the context of watershed ecosystem services. Watershed services considered include regulating, provisioning, supporting, and cultural services. Using data from long-term hydrologic research in the eastern United States and elsewhere, we review principles of water and biogeochemical cycles in natural and altered ecosystems and examine the role of forest cover in maintaining water flows and biogeochemical cycling and improving watershed ecosystem services. Finally, we review existing approaches to minimizing the urbanization footprint on water quality and quantity, such as the maintenance of watershed services across developing landscapes (Postel and Thompson, 2005).

Principle of Water and other Biogeochemical Cycles

Urbanization affects many aspects of a watershed, including surface water dynamics, groundwater recharge, stream geomorphology, climate, biogeochemistry, and stream ecology (O'Driscoll et al., 2010). The key biogeochemical cycles that control watershed ecosystem functions and services (e.g., clean water supply, habitat) include water, nutrient, and carbon cycles. The movement of nutrients and carbon depends largely on water availability in both terrestrial and aquatic components of the landscape. Urbanization affects water quantity, quality (i.e., sediment, nutrient dynamics), and ecosystem primary productivity and carbon sequestration by altering physical, chemical, and biological processes. Key hydrologic processes in a natural forested watershed are illustrated in Fig. 3–1 and are described in detail in the following paragraph. As a comparison, characteristics of hydrologic and biogeochemical processes in an urbanized watershed are presented in Fig. 3–2 and 3–3. Urbanization at the urban–rural interface affects all aspects of the biogeochemical cycles, resulting in water quantity and quality issues observed on site or downstream.

Watershed Hydrologic Cycles and Water Balances

One of the major impacts of urbanization is the effect on watershed hydrology. Understanding the movement or flow of water and water balance is essential to understanding the impact of development on water supply, water quality, and ecological processes.

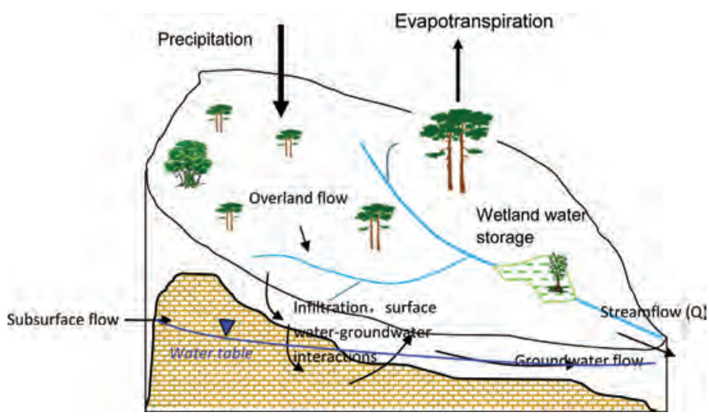

Fig. 3–1. Schematic sketch of water cycle in an undisturbed forested watershed.

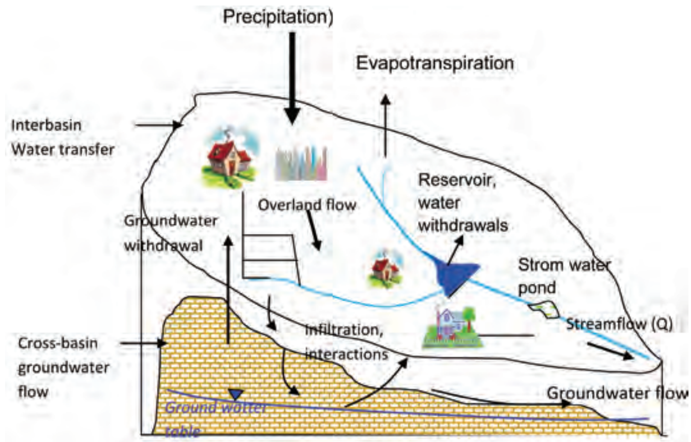

Fig. 3–2. Water cycle of human-impacted watershed.

The key hydrologic fluxes or components of the watershed hydrologic cycle include precipitation (P), evapotranspiration (ET), streamflow (Q), and change in water Storage (ΔS). According to the principles of mass balance, at any time scale (minutes, hours, days, months, years), these four dynamic components are balanced with following equation:

$$\Delta S = P - \mathrm{ET} - Q \quad [1]$$

Precipitation (P) is the largest water input to most watersheds and varies dramatically both in space and time. The distance from ocean and topography are two major controls on precipitation patterns. Global climate change has generally accelerated the hydrologic cycle and thus resulted in more precipitation on land, high interannual variability (more frequent wet or dry years), and high intensity storms (Karl and Knight, 1998).

Evapotranspiration (ET) is the second largest flux of the hydrologic cycle. For vegetated surfaces ET consists of plant transpiration (T) (a physical and physiological process) and evaporation from soil surface and vegetation surfaces (i.e., canopy interception). Evapotranspiration is not only important in understanding the water cycle since it is the largest "water loss" of precipitation to the atmosphere in many cases, it also has important significance in linking the energy balance, ecosystem carbon fluxes (Sun et al., 2011a,b), and predicting regional biodiversity (Currie, 1991). Key factors influencing ET rates include climate (precipitation patterns that affect soil water availability and canopy interception, air temperature, radiation, humidity, wind speed), vegetation types (forests vs. grass), and species composition (Ford et al., 2011). Urbanization and climate change affect watershed hydrology largely through altering the ET processes.

Streamflow (Q) represents the total observed water flow at the outlet of a watershed. The original source of streamflow is of course precipitation, but precipitation has to travel a series of pathways to reach the watershed outlet, and ultimately, the ocean, if not lost to evapotranspiration. The flow paths that redistribute precipitation volumes and chemical compositions are complex in undisturbed systems. Streamflow is a mix of overland flow, subsurface flow, and groundwater flow. Overland flow is water that runs on land surfaces, saturated or unsaturated, depending on rainfall intensity and soil conditions. Subsurface flow is water moved from subsurface soils to nearby streams. Groundwater flow is the water that originates from a saturated portion of the soil or ruptured bedrock. Groundwater can be found rather shallow (less than a meter) or deep (thousands of meters). Groundwater interacts with surface water in rivers or lakes to form one water resource (Winter et al., 1998).

Several terms are commonly used to describe streamflow and hydrographs, including peak flow, stormflow, and baseflow. Peakflow occurs when the streamflow rate reaches its maximum and most destructive force, often causing overbank flooding and channel erosion during a stormflow event. Baseflow refers to streamflow at its low-flow stage (i.e., minimum) during extended periods without precipitation. Baseflow is the water level that is critical to stream aquatic ecosystems and water supply provided by reservoirs (Smakhtin, 2001).

Carbon Cycles in an Urbanized Watershed

Studies on ecosystem carbon cycling have received increasing attention due to concerns about CO_2–induced global warming (Ryan et al., 2010; Shih et al., 2011). In particular, forested watershed ecosystems are believed to be carbon sinks (i.e., they store carbon, preventing it from entering the atmosphere). For example, U.S. forests can sequester about 20% of carbon emissions from the United States (Xiao et al., 2010), and the forest productivity and carbon sink strength is closely coupled with water balances (Xiao et al., 2011; Sun et al., 2011b). Urbanization is likely to increase carbon emissions from the burning of fossil fuels, while reducing carbon sequestration as land is converted from forests.

Terrestrial and aquatic organisms depend on carbohydrate captured by green plants through photosynthesis; this is known as ecosystem primary productivity. Terrestrial and aquatic ecosystem productivity is closely coupled with the hydrologic cycle through plant water use and water availability (Ju et al., 2006). Carbon exchange between a watershed ecosystem and the atmosphere is mostly vertical, but lateral movement of carbon through sediment and soluble forms are important to stream biota that depend on carbon as their ultimate food sources (Shih et al., 2011) (Fig. 3–3). Integrated ecosystem modeling studies suggest that lateral water movement must be considered to correctly estimate landscape carbon fluxes (Chen et al., 2005; Govind et al., 2010).

Similar to the water balance scenario, carbon balance in a watershed can be described as:

$$\Delta C = \mathrm{GPP} - R_t - Q - H_c \quad [2]$$

or

$$\Delta C = \mathrm{NPP} - R_h - Q - H_c \quad [3]$$

where GPP is the gross primary production (g m^{-2} $time^{-1}$) and represents carbon absorption from the atmosphere by green plants, NPP is net primary productivity (= GPP − R_a, where R_a is autorespiration and represents carbon loss for maintaining plant growth), R_t is ecosystem respiration (= $R_a + R_h$, where R_h is heterorespiration, g m^{-2} $time^{-1}$), R_t represents carbon loss to the atmosphere, Q is carbon export in streamflow (g m^{-2} $time^{-1}$), H_c is carbon emission due to human activity (g m^{-2} $time^{-1}$), and ΔC is the change in carbon storage (g m^{-2} $time^{-1}$).

Field carbon flux measurements and modeling studies suggest both GPP and R_t in the above equation are tightly coupled with ET, a key component of the water balance (Sun et al., 2011b). The carbon balance equation also shows the influences of streamflow flux on the changes in watershed carbon storage, i.e., carbon sequestration.

Mass Balances of Physical, Chemical, and Biological Materials in Watersheds

The net effect of land use and land cover on water quality is a function of the magnitude of inputs of a potential pollutant to a watershed and the capacity of processes within the watershed to produce and/or retain potential pollutants. As an example, in most cases, annual nitrogen inputs to a watershed with full forest cover are much less than the assimilation capacity of the forest system (e.g., vegetation, microbial community) for N. Consequently, strong sink activity (or retention) for N is exhibited. Similarly, little N is lost from the system as a result of N dynamics within the watershed because the quantities of N involved are much less than the retention capacity. In contrast, a highly disturbed system may have lost the capacity to act as an N sink because of the absence of vegetation uptake, and, consequently, N inputs are not retained and some of the N involved in intrasystem processes may leave the system as well. A similar analogy can be applied to other land uses and other potential pollutants, such as sediment, pathogens, or metals.

Understanding mass balance and nutrient cycling is the basis to evaluate how urbanization affects water quality. Since water is the carrier of nutrient movement, nutrient cycling is similar to water cycling in its pathways (Fig. 3–4). Any changes in water quantity may result changes in water quality as well.

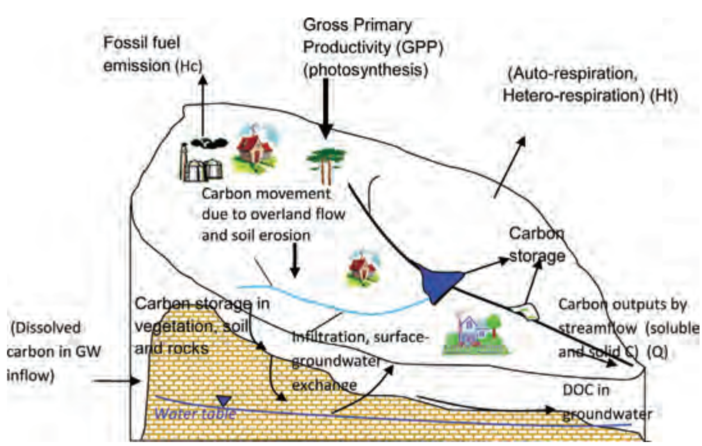

Fig. 3–3. The carbon cycle in a disturbed watershed at the urban–rural interface.

Nutrient balance can be expressed as:

$$\Delta SC_s = PC_p + I - V - QC_r \quad [4]$$

where C_p is the nutrient concentration in precipitation (mg L^{-1}); PC_p is total nutrient deposition (kg ha^{-2} $time^{-1}$), representing total nutrient input from the atmosphere during precipitation events; I is net nutrient flux from human activities (fertilization, harvesting) kg ha^{-2} $time^{-1}$); V is volatilization loss (kg ha^{-2} $time^{-1}$), repenting loss in gaseous forms; C_r is nutrient concentration in streamflow (mg L^{-1}); RC_p is export by streamflow (kg ha^{-2} $time^{-1}$); and C_s is concentration in water storage (mg L^{-1}); ΔSC_s is the change in nutrient quantity (kg ha^{-2} $time^{-1}$).

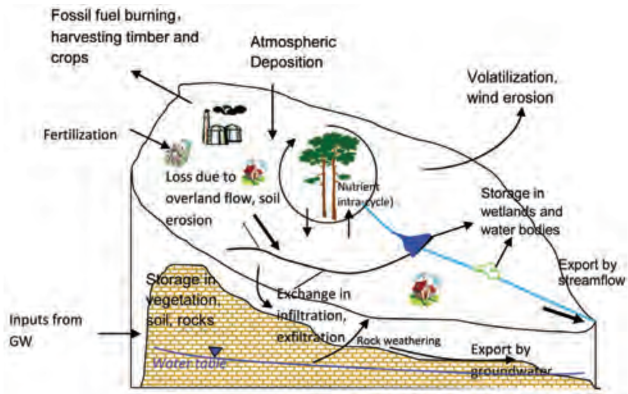

Fig. 3–4. The nutrient cycle in a disturbed watershed at the urban–rural interface.

Impacts of Urbanization on Water Quantity and Quality

Converting rural lands to urban uses alters the landscape structure, inflicts stresses to ecosystems, and has profound impacts on watershed ecosystem structure and functions, and thus water quantity and quality. We will summarize the likely stressors and potential impacts on water parameters and discuss mechanisms behind the impacts (Fig. 3–5).

Water Quantity

The most direct influences of urbanization on watershed ecosystems are alterations of the watershed hydrologic cycle (Fig. 3–2) through altering energy balance and local climate (DeFries and Eshleman, 2004). Using data generated from paired watersheds and other studies, we examine the relationships between the key disturbances and stresses (Fig. 3–5) to each of hydrologic components described in Eq. [1].

Climate and Evapotranspiration

Radiation absorbed by the land surface can be partitioned mainly into two fluxes: latent heat and sensible heat. Latent heat is consumed by evapotranspiration (evaporation and transpiration) by soil–vegetation systems. Evapotranspiration is an effective moderator of near-surface climates, particularly in the warm and dry mid- and low latitudes. Sensible heat is the energy exchanged

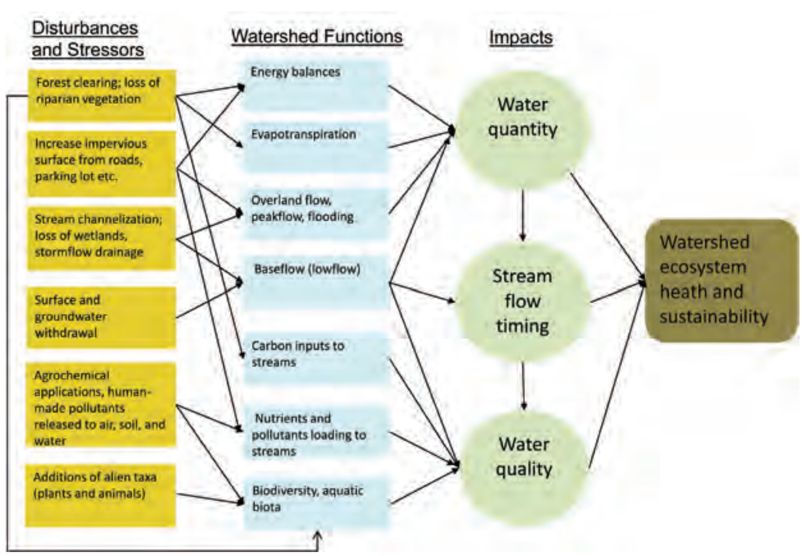

Fig. 3–5. Urbanization induced disturbances and stressors and their impacts on watershed ecosystem functions.

between an ecosystem and the atmosphere by thermodynamics that affect air temperature. Forest clearing or converting forest lands to urban use increases surface albedo (reflection), decreases net radiation, reduces latent heat (Sun et al., 2010), and increases sensible heat that warms the air (Taha, 1997). For example, the albedo of mid-rotation loblolly pine (*Pinus taeda* L.) stand is about 0.1 to 0.15, and a clear-cut site is about 0.16 to 0.18 (Sun et al., 2010), values which are similar to urban areas reported in the literature (Taha, 1997). Studies show that northern hemisphere urban areas have an average of 12% less solar radiation, 8% more clouds, 14% more rainfall, 10% more snowfall, and 15% more thunderstorms than their rural counterparts (Taha, 1997). Studies also suggest the urban heat island may enhance summer storm intensity and change the frequency of freezing rains (Shepherd, 2005). A complete review of the weather effects of urban areas for the southern United States (Atlanta, GA and Houston, TX regions) is found in (O'Driscoll et al., 2010). The ultimate consequences of these changes are reductions in the cooling effects of green vegetation and observed urban heat island phenomena. The magnitude of the changes from a rural environment to an urban environment depends on weather conditions, urban thermophysical and geometrical characteristics, and anthropogenic moisture and heat sources (Taha, 1997). In addition, urban pollutant concentrations can be 10 times higher and air temperatures can be up to 2°C higher than rural areas. Air pollution, such as ozone, can negatively affect ecophysiological processes of forest ecosystems (McLaughlin et al., 2007) and thereby affect the evapotranspiration.

Urban development that generally starts with removing forest vegetation cover reduces plant transpiration and canopy interception (evaporation from vegetation surface), resulting in a dramatic decrease of total ET and infiltration and a large increase in overland flow (Arnold and Gibbons, 1996). At least 20% of annual rainfall can be intercepted by forest canopies and be returned back to the atmosphere as part of ET in the southern United States (Helvey, 1967; Helvey and Patric, 1965; Swank et al., 1972). In central Massachusetts, hardwoods and conifers intercept 11 and 20% of annual rainfall (1140 mm), respectively. Trees consume a large amount of water to maintain productivity and growth. For example, in the southern Appalachians, an eastern white pine (*Pinus strobus* L.) tree with a 40-cm (16-inch) diameter at breast height (DBH) uses about 10 kg of water per day, while a tulip tree (*Liriodendron tulipifera* L.) with the same size consumes eight times as much (Vose et al., 2011). Mean total forest ET varies from 50 to 85% of annual precipitation depending on climate and forest structure, such as leaf area index (Sun et al., 2011a,b). Long-term ecohydrological studies at the Coweeta Hydrologic Laboratory in the southern Appalachians and elsewhere around the world have demonstrated that forest removal or conversion to farmlands could greatly reduce ET, degrade soil infiltration capacity, and increase total streamflow at the watershed scale. Paired watershed studies at Coweeta (annual precipitation = 1750 mm) suggest that clearing a watershed with deciduous trees can reduce ET by 254 mm yr^{-1} (10 inches), or about a 30% reduction of total forest ET.

Peakflow, Stormflow, and Streamflow

Several hydrological variables or indexes can be used to evaluate the impacts of urbanization on streamflow and associated aquatics biota. For example, total streamflow (or water yield) at monthly or longer time frames is most useful for assessing the accumulative impacts on water supply. Peakflow rate and the frequency of certain rare flow regimes or "flushness" of a watershed is most relevant to assess the impacts of land use change on flooding and sediment transport. Low flow is another important indicator of threshold changes of watershed characteristics and plays a major role in water quality, ability to receive waste), and water supply and maintaining aquatic life. Low flow is mainly controlled by geology and climate, but vegetation is also important.

The effects of forest conversion to other land uses on ET and streamflow have been well studied in the United States (Ice and Stednick, 2004) and around the world (Zhang et al., 2001; Andreassian, 2004), with hydrologists commonly use the small "paired watershed" approach as the basis to quantify hydrologic response. In general, watershed manipulation experiments worldwide show that deforestation elevates water yield, and reforestation decreases it (Fig. 3–6) (Andreassian, 2004). Regional reviews are available for the southern United States to examine the impacts of forest management (Sun et al., 2004) and urbanization (O'Driscoll et al., 2010) on watershed hydrology and water quality (Table 3–1). These studies suggest that the magnitude of invoked streamflow change due to land use or land cover change depends on the severity of disturbances (e.g., percentage of forest removal, soil compaction, extent of impervious area, road density), local climate (radiation inputs, rainfall patterns), soil and geology, and other watershed characteristic factors (Fig. 3–6) (Andreassian, 2004). Increases in stream peakflow rates, stormflow volume, and total flow after timber harvest have been attributed to reductions in ET and soil disturbances (Dietterick and Lynch, 1989). In general, forest clearing will cause higher impacts on ET and streamflow in watersheds receiving higher precipitation, such as in the Pacific Northwest. Watersheds with thicker soils may have higher impacts than those with shallow soils. For example, forest clearing in the Appalachians can increase water yield by 260 to 410 mm, or 28 to 65% during the first year after harvest (Swank et al., 2001). Regional simulation studies suggest that coastal areas in the southern United States are more sensitive to vegetation change because of local climatic characteristics (i.e., moderate precipitation and high ET) (Sun et al., 2005).

However, most watershed-scale vegetation manipulation experiments have limits in their potential to answer questions about the effects of urbanization on the hydrologic cycle because the soil and vegetation disturbances employed are generally moderate and temporary. Nevertheless, forest hydrologists have recognized the potential impacts of urbanization on hydrological processes and a forested watershed's ability to provide clean water since the 1960s (Lull and Sopper, 1969). The Baltimore Ecosystem Study (BES), a unique urban component of the Long-Term Ecological Research network (LTER) (Band et al., 2001; Pickett and Cadenasso, 2006) was set up to understand urbanization impacts on ecosystem processes, including water quantity and quality (Band et al., 2001; Meierdiercks et al., 2010).

Lull and Sopper (1969) simulated the likely impacts of four hypothetical urbanization scenarios for a 619-ha (1529-acre) watershed in Pennsylvania. They reported that annual potential ET decreased 19, 38, and 59% if the watershed were covered with 25, 50, and 75% impervious area, respectively, due to vegetation removal. Consequently, runoff increased 15, 29, and 41% with increases in impervious surface area. The estimated increases of runoff occurred mostly during summer months. Lull and Sopper (1969) further examined long-term (24–35 yr) streamflow data in three increasingly urbanized watersheds (25–98% urban area), four partially urbanized watersheds, and nine forested watersheds. They concluded that urbanization in the first three watersheds caused significant increases in annual runoff, stormflow, and annual maximum peak flow with time, and stormflow response (annual volume/precipitation) was most sensitive to urbanization. Comparisons between partially urbanized watersheds and forested watersheds suggested that both designated high and low flows of the urban watersheds were higher than of the forested watershed. During the summer, on average, 12 and 8% of precipitation appeared as streamflow in the urban and forested watersheds, respectively. Lull and Sopper (1969) attributed the hydrologic differences to greater ET and higher infiltration rates in forested areas during the growing season that permit greater storage of summer rainfall and consequently less runoff and baseflow.

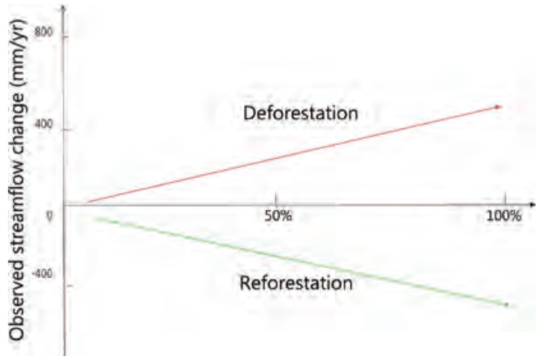

Fig. 3–6. Effects of deforestation and reforestation on watershed water yield. The x axis represents percentage of area converted.

Boggs and Sun (2011) conducted a similar comparison study in the North Carolina piedmont geophysical region by analyzing long-term (2000–2007) monitoring streamflow data collected from an urbanized watershed (UR) (0.70 km^2) that had 44% impervious area and a nearby fully forested watershed (FOR) (2.95 km^2). The UR was much more responsive to rainfall events (Fig. 3–7). The mean annual discharge coefficient (discharge/precipitation) in the UR and FOR was 0.42 and 0.24, respectively. The UR generated about 75% more stormflow than the FOR. The UR had a lower mean ET rate (58%) than the FOR (77%). Peakflow rates in the UR were 13 times higher than in the FOR (e.g., 76.6 vs. 5.8 mm d^{-1}), and so was stormflow volume (77.9 vs. 7.1 mm d^{-1}), and low-flow volume (Fig. 3–8). Growing season precipitation minus discharge normalized by precipitation $(P - Q)/P$ was higher in the FOR than in the UR. Differences between the two watersheds occurred mostly during the growing season and became smaller during the dormant season. They concluded that intensive urbanization elevated watershed peakflow, baseflow, and annual discharge volumes partially due to reduction in ET during the growing season. The Baltimore Ecosystem study (ES) also suggests that the

Table 3–1. Hydrologic responses to urbanization documented in the southern United States (modified from O'Driscoll et al., 2010).

Area/physiographic setting	Total impervious area (TIA) or urban land use	Stormflow	Baseflow	Reference
Raleigh, NC/Piedmont	44% TIA	75% higher in stormflow; 13 times greater peakflow than in forested catchment	Increased low-flow/ baseflow	Boggs and Sun, 2011
Montgomery Co., MD/ Piedmont	~65% urban	3–4 times greater 2-yr peakflows than in forested catchment.	Decreased low flows/baseflow	Moglen et al., 2004
Roanoke River Basin, VA/ Appalachians	6% TIA	9% increase in total flow, 22% increase in 10-yr peak, 73–95% increase in 1–5 yr peaks.	12% watershed decline in groundwater recharge.	Bosch et al., 2003
Watts Branch, MD/Piedmont	32% TIA	2 yr peakflows doubled, greatest increases at confluences.	na	Moglen and Beighley, 2002
Baltimore, MD/Piedmont	30% TIA	Trees interception up to 41%. Runoff decreased by 3.4% when tree cover was increased from 5–40%. Trees could reduce peakflows by 12%.	Doubling TIA reduced baseflow by 17%.	Wang et al., 2008
Baltimore, MD/Piedmont	18% TIA	Simulated streamflow/precip. ratio increased from 0.09 to 0.75 at 80% TIA. Runoff ratio increased rapidly after 20–25% TIA and when soil moisture increased.	Simulated baseflow decline of up to 20%.	Brun and Band, 2000
Baltimore, MD/Piedmont	> 50% TIA	na	Baseflow decreased as TIA increased.	Klein, 1979
GA and MD/Piedmont	> 30% TIA	Significant increase in events exceeding 3 times the median flow for urban streams. Daily % change in streamflow increased from 15% to 19–21% with urbanization.	na	Konrad and Booth, 2005
Accotink Ck, VA/Piedmomt and Coastal Plain	33% TIA	With a historical increase of TIA from 3% to 33% the daily streamflow increased by 48% for periods of normal rain (>6 mm) and by 75% for periods with extreme rain (>35 mm).	Decrease in low flows and increase in flow variability.	Jennings and Jarnagin, 2002
NC; AL, and GA,/Piedmont and Appalachians	Up to 98% urban	More frequent rising events, where total rise is >9 times the median total rise, associated with urban intensity. Relative daily change in stage moderately correlated with urban intensity.	Lack of correlation with low flow and urban intensity.	Brown et al., 2009
NC and AL/Piedmont and Appalachians	Up to 79% urban	Greater flashiness of flow at urban sites (frequency of hourly periods when stage rises/falls by 9–27 cm). Less flashiness when developed land patches are spread out vs. agglomerated.	Shorter duration of low stage flows for urban streams.	McMahon et al., 2003
Greenville, NC/Coastal Plain	Up to 38% TIA	Higher peakflows, and decreased lag times compared with rural. Urban channel incision resulted in deeper water tables.	Baseflow declined from 63% of rural discharge to 35% of urban discharge.	Hardison et al., 2009

Area/physiographic setting	Total impervious area (TIA) or urban land use	Stormflow	Baseflow	Reference
Atlanta GA/Piedmont	>35% TIA	Urbanization increased peakflows. Increased total discharge in wet years, decreased in dry years.	Decreased low flows.	Ferguson and Suckling, 1990
Chattahoochee River, GA/ Piedmont	Up to 40% TIA	Number of times discharge exceeded 9-times the median flow positively correlated with TIA. Number of events discharge increased by 100% in 1 h were positively correlated with TIA.	na	Helms et al., 2009
Atlanta, GA/Piedmont and Blue Ridge	55% urban	Peakflows 30–100% larger than for streams in surrounding less urban catchments. Urban storm recession 1–2 d faster than surrounding streams	Urban low flows 25–35% lower than rural. Urban groundwater levels decreased.	Rose and Peters, 2001
West-central GA/Piedmont	38–48% urban	~100% larger in annual flow.	na	Schoonover et al., 2005
Georgetown Co., SC/Coastal Plain	23% TIA	Runoff 6 times larger for an urban vs. rural watershed and runoff ratio 15% higher for urban.	na	Corbett et al., 1997
NC/Coastal Plain	10% urban	Modeled flow increased 20% total flow	na	Qi et al., 2009
Indian River Lagoon, FL/ Coastal Plain	Up to 35% urban	Event runoff increased up to 55%. Annual runoff increased 49% and 113% for 2 urbanized watersheds.	na	Kim et al., 2002
Miami, FL/Coastal Plain	44% (Directly Connected Impervious Area, DCIA)	Over a 52-yr period 72% of total runoff was generated from the directly connected impervious area (44% of site). Non-DCIA runoff only occurred for large storms.	na	Lee and Heaney, 2003
Econlockhatchee River, FL/ Coastal Plain	Up to 23% urban	River segment draining rural area received 76% groundwater inputs during a storm event, a downstream reach draining up to 23% urban area received only 47% groundwater inputs.	Baseflow decreased along suburban reach.	Gremillion et al., 2000
Barataria Basin, LA/Coastal Plain	13% TIA	For low rainfall events (2.8 cm) and dry soils runoff increased by 4.2 times with 9% TIA.	na	Hopkinson and Day, 1980
Houston Area, TX/Coastal Plain	~8% TIA	For an 88% increase in concrete/asphalt cover, runoff ratio increased ~15%.	na	Khan, 2005
TX and FL/Coastal Plain	17% increase in TIA	Measured precipitation, % TIA changes, and number of individual (>0.5 acres) and general wetland alteration permits were directly related with flood frequency.	N/A	Olivera and DeFee., 2007
Whiteoak Bayou, TX/Coastal Plain	~30% TIA	As watershed urbanized annual runoff increased by 146% (77% attributed to urbanization, 39% attributed to increased rainfall) and peakflows increased by 159% (32% attributed to urbanization, 96% attributed to increased rainfall).	N/A	Brody et al., 2007

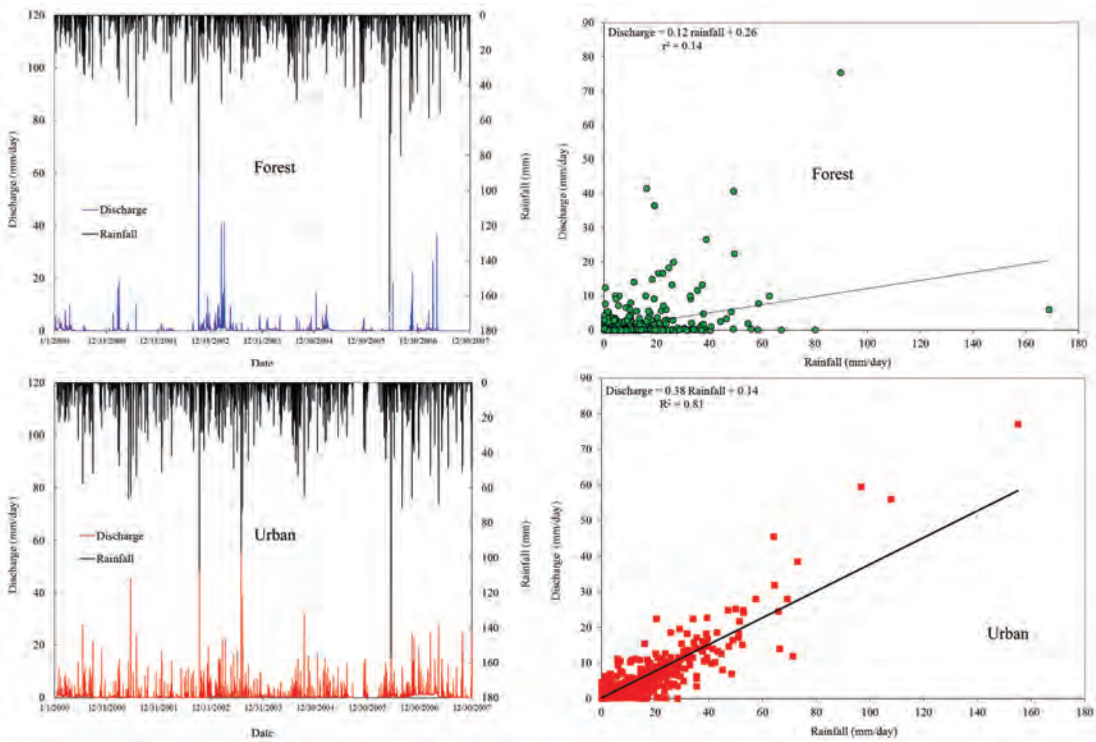

Fig. 3–7. Comparisons between rainfall–runoff relationships for a fully forested watershed (top) and a partially urbanized watershed (bottom) during 2000–2007.

streamflows of urbanized watersheds, with or without storm water management, are highly heterogeneous in space and time, and much flashier and more elevated (up to three times greater monthly flow and higher flow rates) than adjacent forested watersheds, especially in warm seasons (Meierdiercks et al., 2010). Other types of matrices have been developed to evaluate long-term hydrologic effects of urban development and impervious area on stormflow and baseflow (McCray and Boving, 2007).

The observed hydrological responses to urbanization, especially for baseflow, are variable due to complex interactions between natural physical processes (i.e., reduction of ET) and human activities. For example, some studies show that urbanized watersheds have lower baseflow rates due to elevated overland flow and extensive groundwater pumping, and thus reduced groundwater recharge (Barringer et al., 1994). However, baseflow of large watersheds can be augmented by waste water treatment plant effluent in many cases (Paul and Meyer, 2001). Similarly, although some studies have documented that urbanization can elevate peakflow rates by two- to tenfold (Boggs and Sun, 2011), the effects of forest removal on peakflow

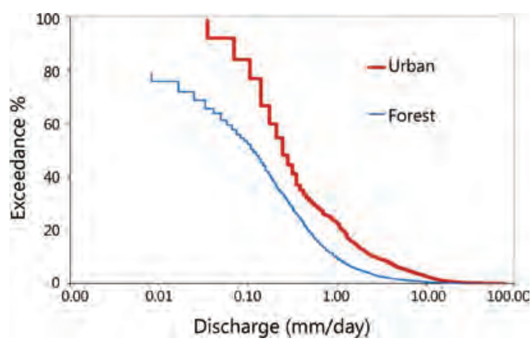

Fig. 3–8. A comparison of the percentage of exceedance curves of daily discharge from January 2000 to December 2007 between an urbanized and forested watershed in the Piedmont of North Carolina. Percentage exceedance is defined as the percent of time that a particular discharge rate occurred during the research period.

rates and floods are more variable, with most literature suggesting forest removal alone does not affect large or atypical flooding that follows large storms (Eisenbies et al., 2007). For large flood events or in a dormant season when ET is low, soils of a forested watershed can be saturated and thus may not differ much from an urbanized watershed. Impervious surface certainly facilitates overland flow moving quickly to streams (Dunne and Leopold, 1978), but due to other factors, such as topography and the locations and sizes of the urbanized areas in a watershed, the true urbanization effects may be masked (Lull and Sopper, 1966, 1969). For example, a recent regional study by Price et al. (2011) found that undisturbed forested watersheds in the humid southeastern United States had higher baseflow rates than other land uses with less forest cover, in spite of the higher ET rates in forests. Those limited regional-scale studies challenge traditional "paired watershed" research results.

The impacts of urbanization and land cover change on streamflow hydrographs depend on the climatic regime and magnitude of disturbance, or so-called total impervious area (TIA) in general. Hydrologic alteration is progressive when converting lands from forests, to grasslands, to urban uses (Fig. 3–9). Urbanization is expected to cause a bigger change in the hydrologic regime (i.e., total flow, peakflow or stormflow) in a wet climate, such as the southeastern United States, than in a drier climate, such as the southwestern United States, because of the likely differential changes in evapotranspiration rates (Zhang et al., 2001). Arid regions naturally have flashy hydrology that is controlled by overland flow generation due to inherent precipitation regimes, and thus urban effects may be obscured on the hydrographs of streams (Grimm et al., 2004). Models (Sun et al., 2011b) are available that can map potential impacts of forest management on annual total water yield is (Fig. 3–10).

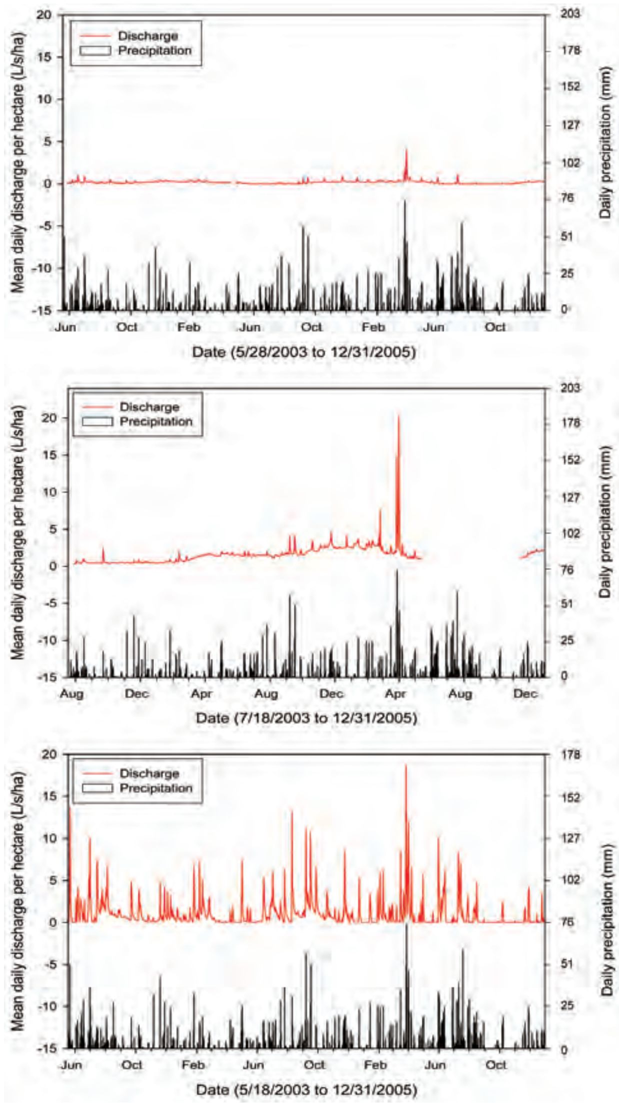

Fig. 3–9. A comparison of stream hydrographs across a land use gradient in west Georgia from (a) forest, (b) grazing, to (c) urban land uses (from Crim, 2007).

Water Quality

As indicated, with minimal disturbance, biogeochemical exports of sediment and nitrogen, phosphorus, and potassium from forests are very low (Schlesinger, 1997; Kimmins, 2004) and in general remain minor in managed forests as well if best management practices are used (Jackson et al., 2004). In terms of mass balances, it is not unusual for accretion (negative export) to occur (e.g., Likens and Bormann, 1995; Swank and Vose, 1997) since many forested systems are highly deficient in these elements and consequently tend to retain inputs. However, reports of Ca and Mg exports from minimally disturbed systems are not uncommon (Kimmins, 2004). Consequently, water quality under most forested conditions is quite good in terms of sediment, N, P, and K concentrations in water. However, there have been reports of elevated fecal coliform in streams of forested catchments as a result of wildlife activity (Fisher and Endale, 1999).

Impacts of urbanization on water quality are primarily derived from two factors: significant production of pollutants and major disruption of the retention capacity of watersheds. The loss of retention capability is heavily associated with the extension of impervious surfaces (Paul and Meyer, 2001), which greatly alter hydrology and consequently reduce the spatial and temporal contact between waterborne pollutants and terrestrial portions of the watershed that might serve as filters. In addition, many pollutants are generated in urban settings, and these are much more diverse than those associated with agriculture or forestry (USGS, 1999; de la Cretaz and Barten, 2007). In addition to sediment and nutrients, urban waters often contain pharmaceuticals such as antibiotics, analgesics, narcotics, and psychotherapeutics; pesticides from lawns and recreational areas; metals from brake linings and industrial activity (Paul and Meyer, 2001); and pathogenic microbial populations associated with sewage leaking from sewage lines or combined stormwater–sewer overflows (Tibbetts, 2005). Paul and Meyer (2001), de la Cretaz and Barten (2007), and Nagy et al. (2011) provided comprehensive reviews of the effects of conversion of forested watersheds to urban land uses on sediment and biogeochemical aspects of water quality and concluded that urban development amplifies water quantity and quality problems primarily as a result of increases in impervious surfaces.

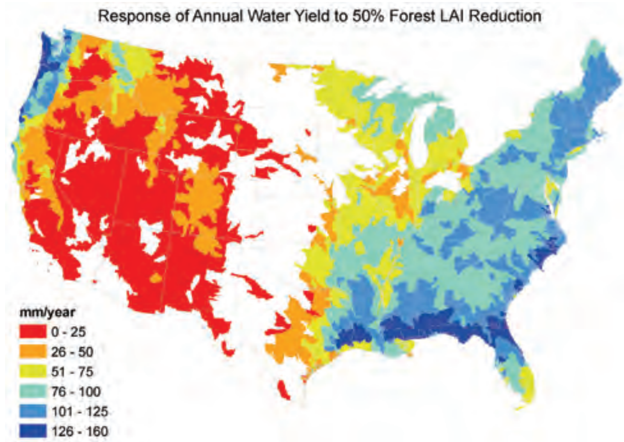

Fig. 3–10. Simulated increase of water yield across the United States in response to forest clearing that reduces 50% leaf area index (LAI) by the Water Supply Stress Index (WaSSI) ecosystem model (Sun et al., 2011b).

In general, the most consistent physiochemical response in streams to urbanization is increased NO_3 concentrations. Total P, K, and SO_4 also often increase, although exceptions occur (e.g., no significant difference in total P between forested and urban streams in Georgia; Schoonover and Lockaby, 2006). The responses of stream NH_4 and dissolved organic carbon concentrations to forest to urban conversion are particularly variable and may increase or decrease. Insights regarding urban signatures on water may be gained from examination of one of the few long-term datasets collected across an urbanizing landscape (i.e., the Altamaha River Basin in GA) as urbanization and population increases occurred over 30 yr (Weston et al., 2009). As the watershed underwent conversion of agricultural lands to urban cover, total N, nitrate or nitrite (NO_x), and P concentrations increased over time while organic C and NH_4 concentrations declined. As an example, the concentrations of total N, NO_x, and total P in the Upper Ocmulgee River (the subwatershed of that basin with the highest population densities) were 1.5, 2.4, and 1.8 times, respectively, those of subwatersheds with lower densities (Weston et al., 2009). Examples of the effects of urbanization on water quality parameters are presented in Table 3–2.

It is well documented that conversion of portions of watersheds from forest to urban cover often significantly increases concentrations and loads of sediment and nutrients in surface waters (Table 3–2) (Paul and Meyer, 2001; de la Cretaz and Barten, 2007; Nagy et al., 2011). The threshold of imperviousness at which these changes often occur is generally reported to be between 10 and 20% of the watershed area (Arnold and Gibbons, 1996; Bledsoe and Watson, 2001). However, noteworthy impacts on water quality have also been shown to occur at levels of imperviousness as low as 5% (Bolstad and Swank, 1997; Schoonover et al., 2005; Crim, 2007). In addition to increasing concentrations and loads, the variability of stream physiochemistry rises as proportions of impervious surface in watersheds increase (Crim, 2007; Fig. 3–11). Reduced stability of the physiochemical environment may have negative implications for some types of aquatic biota as well. The impact of urban land use on water chemistry is sufficiently strong in the southeastern United States to supersede that of physiographic influences (Nagy et al., 2011). However, significant variation occurs among regions in the case of sediment, as areas with steeper topography tend to generate more sediment.

Inputs of other substances to streams as a result of urban land use may be of greater concern than sediment and nutrients. Elevated copper, chromium, lead, cadmium, mercury, zinc, and nickel have been reported in streams or sediments downstream from urban areas (Nagy et al., 2011). Organic pollutants such as PCBs have also been shown to be higher in urban compared to forested streams in South Carolina coast (Sanger et al., 1999). In addition, personal care products such as deodorants and

Table 3–2. Examples of physical, chemical, and biological differences between urban vs. forested streams.

Parameter	Urban compared to forested	Location	References
Sediment	5000× higher export during construction	Baltimore, MD Washington, DC	Wolman and Schick, 1967
	3× higher conc. well after construction period	Appalachian Mtn., NC	Price and Leigh, 2006
Nitrate	2× higher conc.	Columbus, GA	Schoonover et al., 2005
	7× higher conc.	Appalachian Mtn., NC	Price and Leigh, 2006
Phosphorus	9× higher conc.	Northeast USA	de la Cretaz and Barten, 2007
	0× higher	Columbus, GA	Schoonover et al., 2005
Fecal coliform	4–5× higher	Appalachian Mtn., NC	Bolstad and Swank, 1997
	10× higher	Pittsburgh, PA	Gibson et al., 1998
E. coli	6× higher	Columbus, GA	Crim, 2007
Pesticides	Present in 1/3 of urban creeks	Pacific Northwest USA	Weston et al., 2011
Pharmaceuticals	Present in 80–91% of urban creeks	Iowa	Kolpin et al., 2004

pharmaceuticals are often prevalent in urban streams. As an example, under low flow conditions, 86, 80, 40, 40, and 76% of urban streams sampled in an Iowa study contained non-prescription drugs, steroids, fragrances, antibiotics, and prescription drugs, respectively (Kolpin et al., 2004). It has been suggested that conventional water treatment facilities may be limited in their capability to remove these substances (Bolong et al., 2009).

Concentrations of fecal coliform and *Escherichia coli* are frequently found to be high in urban streams (Paul and Meyer, 2001), although, as previously mentioned, there have been reports of elevated fecal coliform in forested streams from wildlife (Fisher and Endale, 1999). In a study near Columbus, GA, Schoonover et al. (2005) found that fecal coliform concentrations in urban streams were consistently elevated over those dominated by pasture and forested land uses by a large margin (Fig. 3–12). Similarly, Crim (2007), working in the same area, estimated average fecal coliform in urban stormflow at 2750 most probable number (MPN) compared to 262 in forested streams.

Conversion of forests to urban land use has been shown to have major implications for the biotic integrity of streams. Often, abundance and diversity of fish, amphibians, reptiles, and invertebrates decline as species that are sensitive to disturbance are replaced by fewer number of species that are more tolerant of altered habitat. Similarly, the health of fish as measured by the presence of eroded fins, lesions, and tumors may decline as well (Helms et al., 2005). Alterations in reproduction patterns, possibly as an adaptation to increased velocity in urban streams, have been reported in salamanders (Price et al., 2006; Barrett et al., 2010). Impacts on stream biota may be driven by increased velocity, changes in stream bed substrate, and increases in physiochemical and/or biological contaminants, among

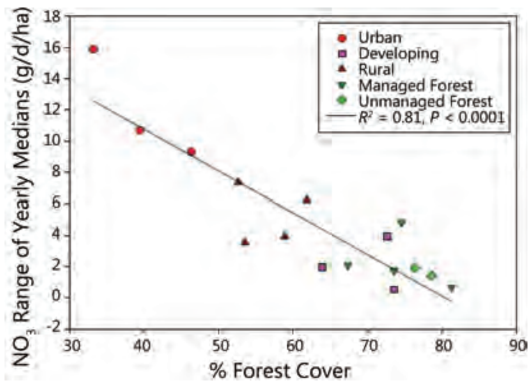

Fig. 3–11. Ranges in nitrate loads (medians) for 2003 through 2005 across a forest cover gradient in the Georgia Piedmont (Crim, 2007).

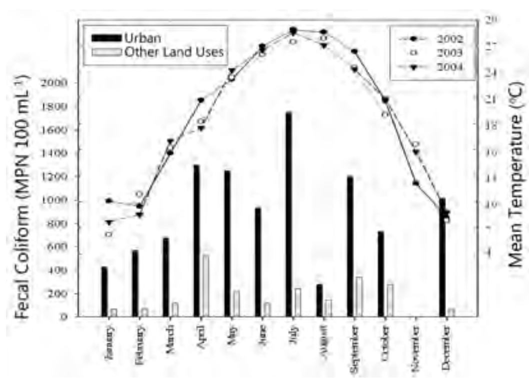

Fig. 3–12. Monthly fecal coliform counts (bars) of urban watersheds versus all other land uses combined in relation to mean annual air temperature (lines) (Schoonover and Lockaby, 2006).

other factors (Paul and Meyer, 2001; Barrett and Guyer, 2008; Barrett et al., 2010).

The hydrologic and water quality changes that often arise following forest conversion to urban areas have negative implications for human health in some instances. As an example, the high concentrations of fecal coliform, *E. coli*, and other pathogenic organisms that are found in many urban streams (Paul and Meyer, 2001; Nagy et al., 2011) may be linked to the occurrence of gastrointestinal illnesses in people coming in contact with contaminated waters (Tibbetts, 2005). Globally, approximately 1.5 million deaths annually are caused by diarrhea related to water supply and sanitation problems (Bjorklund et al., 2009).

In the United States, as a result of the unstable hydrology associated with high levels of impervious surface, combined sewer overflows often overflow during storms and may inject untreated sewage effluent directly into associated streams. As a consequence of the increased pollution and pooling of overflow waters, habitat for disease vectors such as *Culex* spp. mosquitos may be enhanced, thereby stimulating the likelihood of WNV infection in birds and humans (Vazquez-Prokepec et al., 2010).

Water Supply under Multiple Stresses in an Urbanizing Environment

Water supply for human use has been increasingly stressed worldwide as a result of population rise, climate change, land conversions, groundwater overdrawl, and associated water quality degradation (Sun et al., 2008a,b). The U.S. population now exceeds 300 million, and it is projected to almost double within the next 50 yr. Many of the metropolitan areas are expanding, and population is expected to increase at least 50% in the next 20 yr. The South, accounted for more than one-half of the newly developed land in the United States during the period of 1982 through 2007. A recent modeling assessment study (Wear, 2011) concludes that urban land use in the South will double (from ~12,141,000 to 24,282,000 ha, or from 30 million to 60 million acres) by the year 2060, while population will increase by 60 to 80% to about 105 million (Fig. 3–13). The report warns that the general challenge to the sustainability of southern forests, especially in areas where population growth is likely to be concentrated in the Piedmont (portions of the Southern Appalachians bordering Carolinas), the urban areas of Texas, and Peninsular Florida.

In the western United States, highest water stress areas are found in the Midwest, the California Central Valley, and a few major cities (Fig. 3–14), where natural rainfall is insufficient to support intensive agriculture and large human population, as evidenced by groundwater overuse and interbasin water transfer that has caused serious ecological concerns of sustainability. The eastern United States is endowed with high precipitation. However, even in the "water-rich" regions, high water demand by thermoelectric power plants, irrigation, and domestic water withdrawal by metropolitan areas are resulting in some watersheds facing difficulty (Fig. 3–15) in supplying reliable water for both human and aquatic water use and are becoming extremely vulnerable to climatic variability, such as droughts (Lockaby et al., 2011). In the United States, on average, about 545 L (144 gallons) of water are needed for domestic water use per capita. Although past water use surveys suggest that U.S. total water demand has stabilized, large future population growth in certain regions such as the southeastern United States will likely place further stress on water resources, especially under a warming and drying climate regime (Sun et al., 2008a,b).

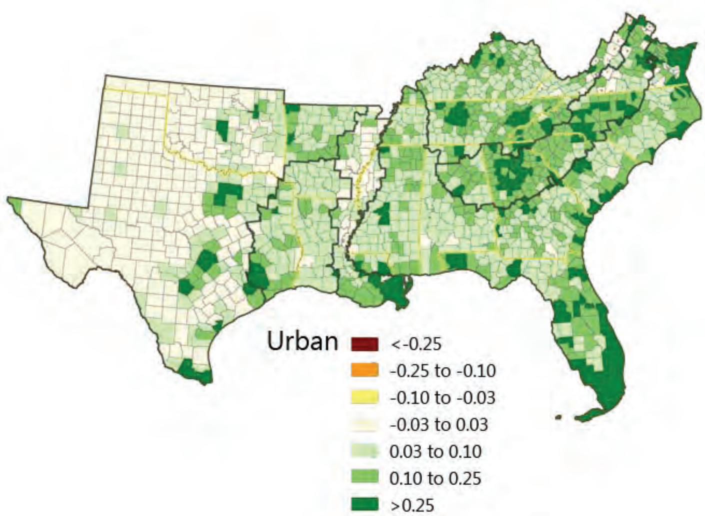

Fig. 3–13. Change in urban land uses, 1997 to 2060, based on an expectation of moderate urbanization gains with increasing timber prices (Wear, 2011).

Management Options for Mitigating Changes in Water Quantity and Quality

Urbanization dramatically alters the ecosystem processes and consequently water quantity, quality, and streamflow timing characteristics of watersheds. Some of the most effective options for protecting water quantity and quality across urbanizing landscapes involve management of land cover at the watershed or subbasin scale. Based on the concept of watershed services (Postel and Thompson, 2005), the general goal is to gain some influence over forest land management within the hydrologic unit of interest and prevent development from occurring in critical locations that are particularly influential in protecting water quantity and quality. If forests can be retained or restored in those locations, the stabilizing influence of forest cover on water supply and on minimization of nonpoint source exports can be maintained (Nagy et al., 2011). The basis for the approach is that costs of retaining forest land in a watershed may be less than those associated with conventional approaches to rising water treatment and/or supply needs (Pires, 2004; Patrick, 2009; Trepel, 2010). As an example, in Georgia, the value of wetland forests for water quality and quantity regulation has been estimated at $11,588 to $20,490/ha depending on proximity to urban areas (Moore et al., 2011).

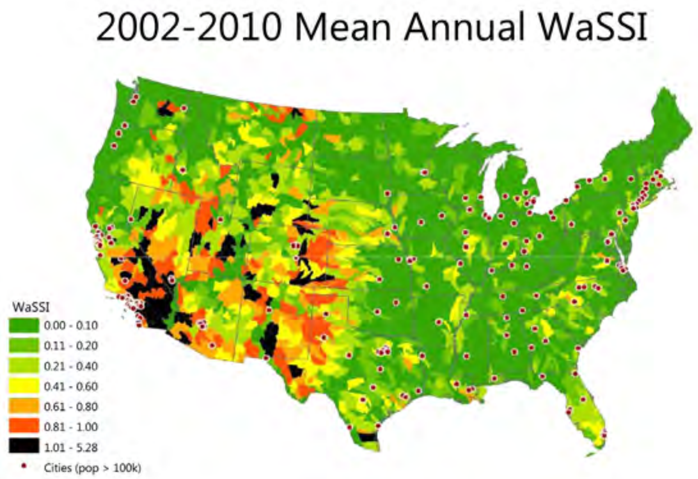

Fig. 3–14. Modeled mean (1996–2005) Water Supply Stress Index (WaSSI = Water Demand/Water supply) across the United States. The higher of the WaSSI value, the bigger the water stress. A value of 0.4 is considered severe stress. The solid dots represent cities with population greater than 100 thousands.

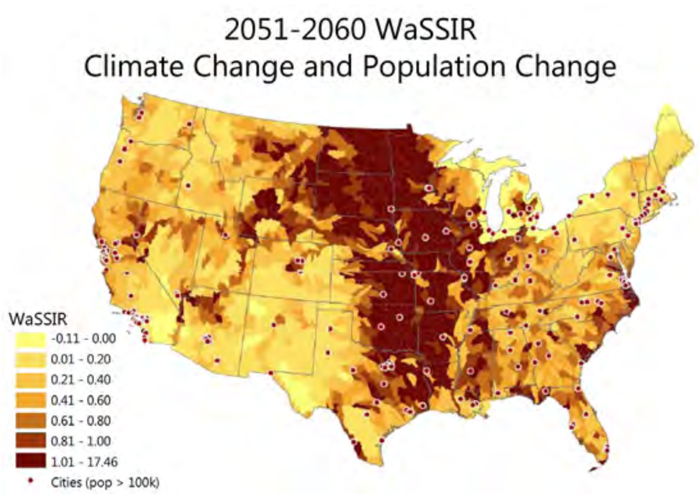

Fig. 3–15. Percentage of change in water supply stress index (defined by the Water Supply Stress Index, or WaSSI = water demand/water supply) caused by population and climate change by 2050.

The costs of implementing conventional approaches are often very high. For example, the cost of building new reservoirs is generally more than $200 million (e.g., $363 million for Palm Beach, FL; www.sun-sentinel.com/community/sfl-flrpfwater- June, 2009) and remedial efforts such as construction of large-scale waste water retention tunnels and refitting water infrastructure for major cities are much more (e.g., $4 billion for Atlanta; Online 6/24/11 www.atlantaga.gov/Government/Watershed). The stable hydrology and greater water availability afforded by forest compared to impervious cover (Nagy et al., 2011) may help reduce the need for such measures.

The ecosystem or watershed services methodology requires integration among socioeconomic, hydrologic, environmental, and other considerations while addressing issues such as fragmented ownership objectives, development of a valuation system, and compliance assessment. These approaches are complex but are becoming better refined as their usage increases in many parts of the world (Jack et al., 2008). Globally, they will be relied on to a greater extent in the future as water becomes a much scarcer commodity (Bjorklund et al., 2009).

Conclusions

Alterations of watershed water cycles are the root causes of many chain reactions of stream ecosystem degradation present in today's urban areas. Knowledge gaps exist regarding interactions among processes of urbanization (land conversion, increasing impervious areas, new pollutants), hydrological (water budget change, infiltration, and ET processes), and ecological (biota change) functions at different temporal and spatial scales (Wenger et al., 2009). Past ecosystem-scale studies have concentrated on small natural watersheds, and long-term monitoring of large basins dominated by human activities would be more beneficial to landscape planners in designing systems (i.e., urban best management practices) that can minimize negative impacts of development (Korhnak and Vince, 2005). Understanding the interactions of human behavior and urbanizing environments is critical to developing sustainability science, an emerging discipline that calls for integrating social and economic values with the physical, chemical, and ecological functions of ecosystems.

It is clear that rising populations and increased development pose major threats to future supplies of clean water. These threats will be exacerbated in many areas by periodic droughts and wider ranges of temperature and precipitation associated with human-caused climate change. An urbanizing environment is likely to be more susceptible to negative climate change impacts due to the loss of the buffering capacity of natural ecosystem services.

Some combination of factors, such as infrastructure renovation, improved design of new water and sanitation systems, and expanded implementation of watershed services management, will need to be aggressively employed to provide clean water for expanding human populations. Prevention of increased human health problems caused by sanitation and hydrologic problems will be dependent on the previously mentioned suite of factors as well.

In the future, innovative implementation of the watershed services concept may represent the primary hope for maintaining water quality and quantity as global populations and urbanization increase (Millennium Ecosystem Assessment, 2005). This will be particularly true in developing countries, where the highest increases in urbanization are expected and where governments may be unable to provide frameworks for protection of clean water supplies (Bjorkland et al., 2009).

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