

Review

Urbanization Effects on Watershed Hydrology and In-Stream Processes in the Southern United States

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Abstract: The southern United States is characterized by a humid, subtropical climate and consists of 16 states (Texas, Oklahoma, Arkansas, Louisiana, Mississippi, Tennessee, Kentucky, Alabama, Florida, Georgia, South Carolina, North Carolina, Virginia, West Virginia, Delaware, and Maryland) and Washington DC. Currently this region is experiencing the largest net population growth in the U.S. Over the last century, the expansion of large urban centers and impervious area in the region has altered the hydrologic cycle. This review synthesizes regional research that shows how watershed hydrology, groundwater recharge, stream geomorphology, climate, biogeochemistry, and stream ecology have been affected by urbanization and the expansion of watershed impervious area.

Keywords: urbanization; TIA; surface water; groundwater; stormwater runoff

1. Introduction

Across the globe human populations are becoming increasingly urban, with approximately fifty percent of the world's population currently residing in urban areas [1]. Urban centers often develop along rivers because of the river's capacity to provide drinking water and transportation [2]. Over the last two centuries, urbanization has caused changes in watershed hydrology that include declines in the natural filtering capacity of river systems (e.g., channelization of headwater streams, loss of floodplains and wetlands) and regulation of flows due to the construction of dams and impoundments. Such changes have resulted in globally altered watershed sediment and solute export [3]. With urbanization comes an expansion of watershed total impervious area (TIA), in the form of parking lots, roadways, lawns, and rooftops. TIA reduces infiltration and surface storage of precipitation and increases surface water runoff [4]. Several recent reviews have synthesized the literature on physical, hydrological, chemical, and ecological effects of urbanization on stream systems [5-7]. Stream degradation is often associated with increased watershed imperviousness and has been referred to as the "urban stream syndrome" [6,8].

In the U.S., the southern region is one of the most rapidly urbanizing areas in the nation. Recent urbanization has increased the potential for hydrological alterations and channel disturbances along streams. Urban stream responses may differ across physiographic regions because of differences in climatic inputs, vegetation, geology, slope, stream geomorphology, and hydrologic processes [9]. There is a growing need to synthesize regional hydrologic responses to urbanization to improve the understanding of stream response to land use change. In the southern U.S. these efforts can guide future research and watershed management efforts in this rapidly urbanizing region. The purpose of this manuscript is to summarize and synthesize the current published literature on effects of urbanization on watershed hydrology and in-stream processes in the southern U.S. Our review aims to answer the following questions:

- What are the predominant physical, chemical, and ecological effects that urbanization has on southern streams and their watersheds?
- What are the challenges and opportunities that exist for improving the understanding of the effects of urbanization on southern streams and their watersheds?

2. Study Area

The southern U.S. is characterized by a humid, subtropical climate and consists of 16 states (Texas (TX), Oklahoma (OK), Arkansas (AR), Louisiana (LA), Mississippi (MS), Tennessee (TN), Kentucky (KY), Alabama (AL), Florida (FL), Georgia (GA), South Carolina (SC), North Carolina (NC), Virginia (VA), West Virginia (WV), Delaware (DE), and Maryland (MD)) and Washington D.C., as defined by the U.S. Census Bureau (Figure 1). Major physiographic settings in the southern U.S. include the Gulf Atlantic Coastal Plains, Gulf Atlantic Rolling Plain/Piedmont, Appalachian Highlands (Blue Ridge, Valley and Ridge, Appalachian Plateau), Midcontinent Plains and Escarpments, and the Lower Mississippi Alluvial Plain [10] (Figure 1). Most of the larger cities in the southern U.S. have developed in the Gulf Atlantic Rolling Plain/Piedmont or Gulf Atlantic Coastal Plain settings (Figure 1). Seven of

the nation’s most populous metropolitan areas (>2.5 million residents) are found in these settings: Dallas, Miami, Washington D.C., Houston, Atlanta, Baltimore, and Tampa metropolitan areas.

The southern U.S. has experienced large changes in population growth and land use over the last century. Regionally, a large shift of population from rural to urban areas has occurred. For example, in 1900, 18% of the population resided in urban areas but by 2000, urban residents made up 72% of the total population of the southern U.S. (Figure 2). Since the 1940s rapid population growth has been occurring and continues in the region. Most recently (2000–2008), the southern U.S. experienced the largest regional population increase (11.5 million) in the nation [11]. The expanding population in the southern U.S. has resulted in rapid urbanization; from 1982–2007 a 79% increase (21 million acres) in the extent of developed land occurred (based on analysis of urban and transportation networks) [12] (Figure 3). The southern region accounted for more than half of the newly developed land in the contiguous U.S. during the period of 1982–2007 (USDA, 2009). Future projections for the region suggest more extensive urbanization [13] with possibly a 3–4 fold increase in the extent of urban areas over the next 40 years [14].

Figure 1. Physiographic settings and major cities of the southern U.S. (modified from Hammond, 1970) [10].

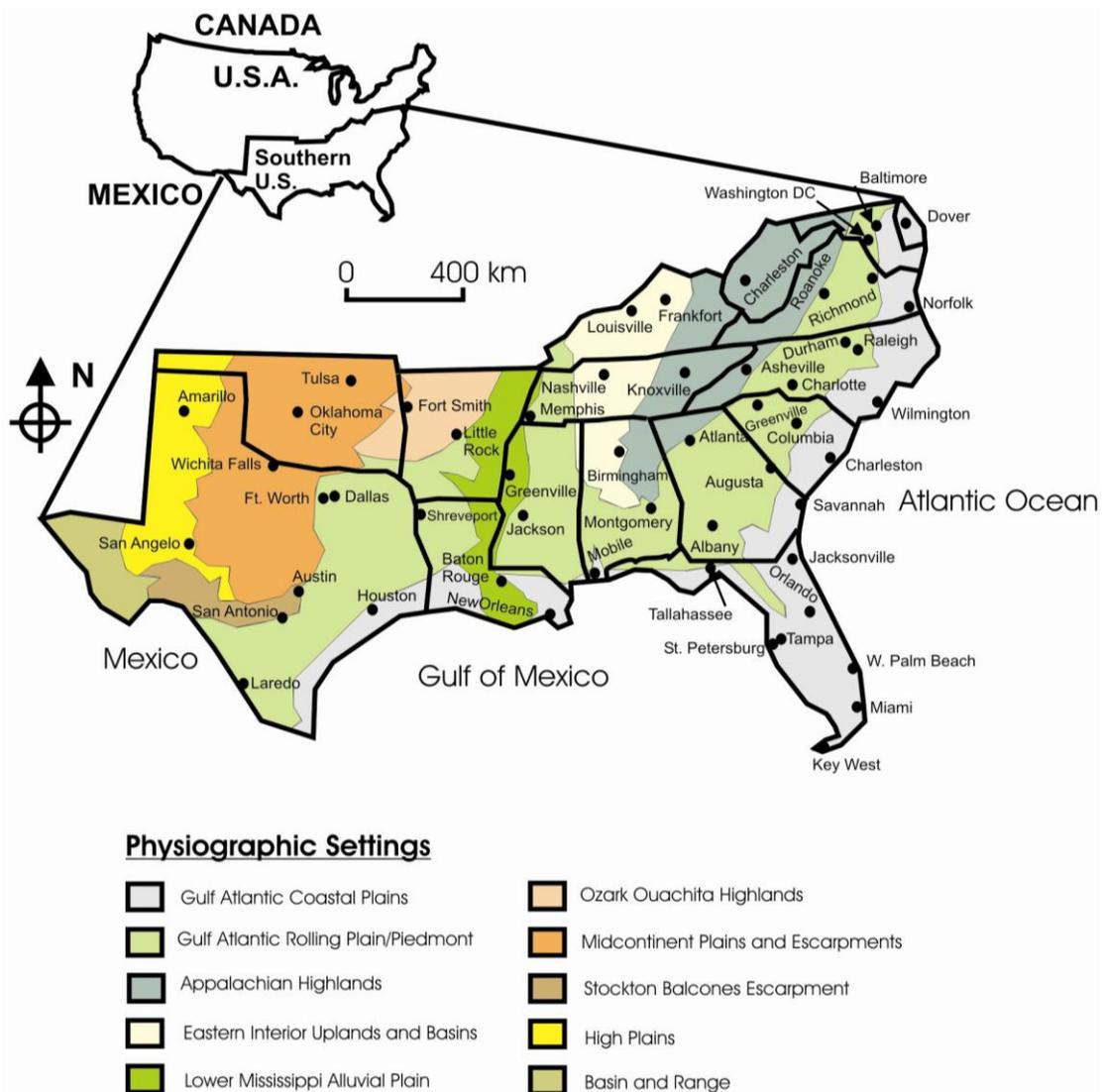


Figure 2. (A) Population change for the period 1900–2000 in the southern U.S. and (B) the percentage of U.S. and southern U.S. population residing in urban areas (source: U.S. Census).

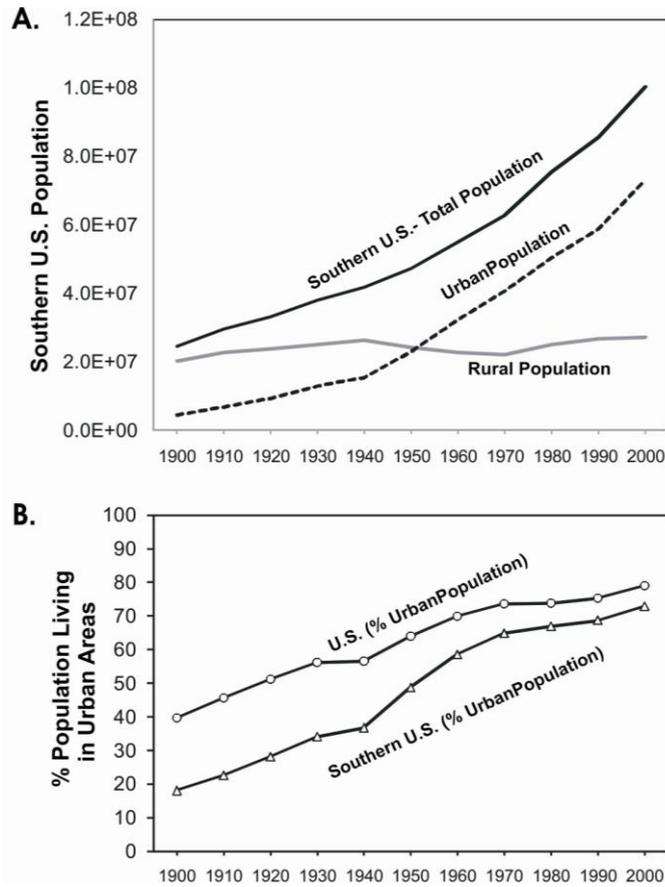
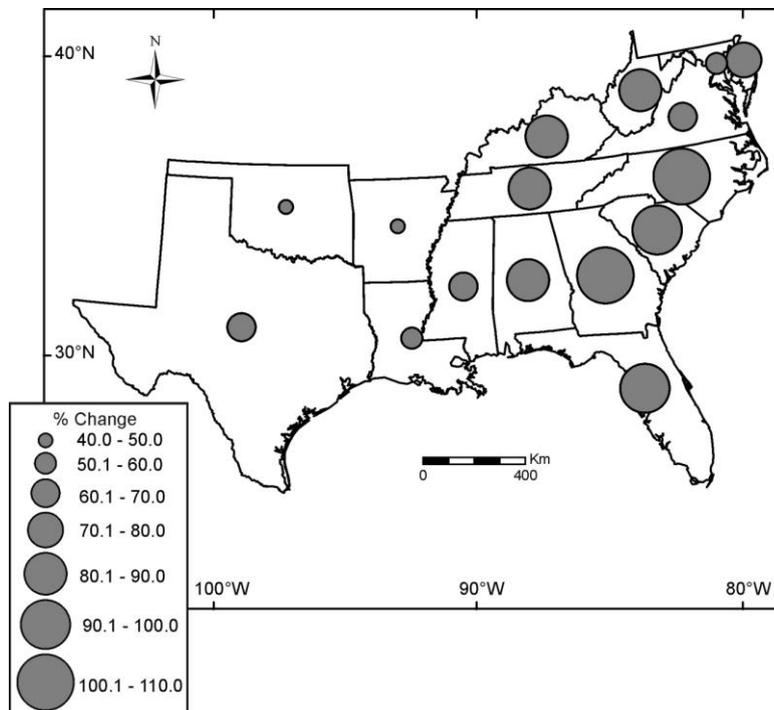


Figure 3. Changes in the extent of developed land (urban and transportation lands) in the southern U.S. from 1982–2007 (source: 2007 National Resources Inventory; USDA, 2009).



3. Urbanization and Watershed Hydrology

3.1. Urbanization Effects on Precipitation and Evapotranspiration

For decades, it has been observed that urbanization commonly increases the storm runoff response to precipitation (Table 1) due to greater stormwater peaks generated by impervious surfaces. Only recently has it been shown that urban land use may also influence the timing and magnitude of precipitation inputs to urban watersheds. Urban induced rainfall can be a result of the urban heat island effect (a warming of the local climate due to changes in land cover, drainage, shading, and albedo). Urban surfaces are generally drier and release more heat than surrounding rural areas. The urban heat island can alter convection of air masses in urban areas. In addition, urban surface roughness and the urban canopy (buildings, infrastructure, or trees) can affect air circulation. The presence of enhanced aerosols in urban areas may also influence local climate. However, the role of aerosols (providing cloud condensation nuclei sources) is still under debate and in some cases aerosols may reduce rainfall [15]. In a literature review of urban induced rainfall, Shepherd (2005) found that warm season rainfall increases were common downwind of major cities in the U.S. In addition, urban heat island effects have resulted in a decrease in the frequency of freezing-rain occurrences in large U.S. cities [16]. Over the past decade, numerous studies on urban induced rainfall have been conducted in the southern U.S., with most focused in the Atlanta, GA and Houston, TX regions.

Table 1. Hydrologic responses to urbanization documented in the southern U.S.

Study Area/Physiographic Setting	TIA or urban land use	Runoff	Baseflow	Ref.
Montgomery Co., MD/Piedmont	~65% urban	3–4 times greater 2 yr peakflows than in forested catchment.	Decreased low flows/baseflow	[27]
Roanoke River Basin, VA/Appalachians	6% TIA	Low density development had greatest hydrological impact due to highest per capita TIA. 9% increase in total runoff, 22% increase in 10 yr peak, 73–95% increase in 1–5 yr peaks.	12% watershed decline in groundwater recharge.	[29]
Watts Branch, MD/ Piedmont	32% TIA	2 yr peakflows doubled, but within the watershed could range from approximately 1.25–3 times larger due to urbanization, greatest increases at confluences.	N/A	[30]
Baltimore, MD/ Piedmont	30% TIA	Trees reduced runoff for small rain events and could intercept 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%.	[31]
Baltimore, MD/Piedmont	18% TIA	Simulated runoff ratio (precipitation/streamflow) 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%.	[32]
Baltimore, MD/ Piedmont	Over 50% TIA	N/A	Baseflow decreased as TIA increased.	[33]

Table 1. Cont.

Study Area/Physiographic Setting	TIA or urban land use	Runoff	Baseflow	Ref.
GA & 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.	N/A	[34]
Accotink Ck, VA/ Piedmont & 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.	[35]
NC; AL, & GA, /Piedmont & Appalachians	Up to 98% urban	More frequent rising events, where total rise is >9X 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.	[36]
NC & AL/ Piedmont and Appalachians	Up to 79% urban	Greater flashiness of flow at urban sites (frequency of hourly periods when stage rises/falls by 0.3–0.9 ft). Less flashiness when developed land patches are spread out vs. agglomerated.	Shorter duration of low stage flows for urban streams.	[37]
Greenville, NC/ Coastal Plain	Up to 38% TIA	Urban storm hydrographs had higher peakflows, lower base flows, 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.	[38]
Atlanta GA/ Piedmont	>35% TIA	Urbanization increased peakflows. Increased total discharge in wet years, decreased in dry years.	Decreased low flows.	[39]
Chattahoochee River, GA/ Piedmont	Up to 40% TIA	# of times discharge exceeded 9-times the median flow positively correlated with TIA. # of events discharge increased by 100% in 1 h were positively correlated with TIA.	N/A	[40]
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 days faster than surrounding streams	Urban low flows 25–35% lower than rural. Urban groundwater levels decreased.	[41]
West-central GA/ Piedmont	38%–48% urban	Greater annual runoff (approximately 100% larger for urban streams).	N/A	[42]
Georgetown Co., SC/ Coastal Plain	23% TIA	Runoff 6X larger for an urban vs. rural watershed and runoff ratio 15 % higher for urban.	N/A	[43]
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.	N/A	[44]
Miami, FL/ Coastal Plain	44% 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.	N/A	[45]
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 inputs decreased along suburban reach.	[46]

Table 1. Cont.

Study Area/ Physiographic Setting	TIA or urban land use	Runoff	Baseflow	Ref.
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.	N/A	[47]
Houston Area, TX/Coastal Plain	~8% TIA	For an 88% increase in concrete/asphalt cover, runoff ratio increased approximately 15%.	N/A	[48]
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	[49]
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	[50]

Bornstein and Lin (2000) showed that the urban heat island effect caused convective activity in Atlanta that was responsible for the occurrence of three out of six summer storm events studied [17]. Dixon and Mote (2003) found urban induced precipitation in Atlanta occurred frequently in the summer and was most common in July [18]. Over a five year period, they observed as many as 15 urban induced precipitation events in a year. A direct relationship was observed between the number of urban induced precipitation events and the distance from major highways in the Atlanta metropolitan area. Factors influencing urban induced precipitation included low level moisture, urban heat island intensity, and atmospheric instability. Using ground-based weather radar, Mote *et al.* (2007) showed that urban induced rainfall resulted in a 30% increase in rainfall in eastern metropolitan Atlanta, relative to the western metropolitan area [19]. The effects of urban induced rainfall could be measured up to 80 km to the east of the center of Atlanta.

Similar urban induced precipitation has been observed in Houston, TX. Burian and Shepherd (2005) analyzed data from 19 rain gauges in the Houston area and compared pre- and post-urban data [20]. They found that during the warm season the urban area had 59% greater rainfall amounts between noon and midnight when compared to an upwind control area. The urban area also had 80% more occurrences of warm season rainfall events between noon and midnight (relative to the surrounding area). The mean warm season precipitation amount increased by 25% in the urban area from a pre- to post-urban time period. These studies suggest that spatial and temporal variability in runoff generation in urban areas in the southern U.S. (particularly in the summer months) may also be related to urban induced precipitation inputs.

Although evapotranspiration (ET) can be the dominant water flux in catchment water budgets in the southern U.S. [21], few published studies have directly evaluated the effects of urbanization on ET in the southern U.S. or have attempted detailed urban water budgets (except see [22,23]). In a study of seven North American cities (including Miami, FL), Grimmond and Oke (1999) showed that ET in urban areas generally increased with an increase in vegetated area [24]. In several U.S. cities, ET was

found to exceed the precipitation inputs, which was possible because water was also supplied by irrigation and the piped urban water supply. Irrigation restrictions in cities were shown to greatly influence urban ET rates. However, there were no consistent patterns relating ET to urban land use.

Dow and DeWalle (2000) used a water budget and meteorological approach to evaluate the effects of urbanization on watershed ET for 51 watersheds in the eastern and southern U.S. [25]. They found significant decreases in watershed ET that were linked with increases in urban/residential development. Based on their empirical relationships between degree of watershed development and watershed ET, a watershed that reaches 100% urban land use would exhibit a reduction of ET of approximately 22 cm/yr. These studies suggest that urbanization can alter ET and urban catchment water budgets in the southern U.S., but more work is needed to determine if observed patterns are consistent or if urban ET rates vary throughout the region based on factors such as topography, vegetation, development density and urbanization age. Published studies directly measuring ET rates and processes in urban areas of the southern U.S. are lacking, although Sun *et al.* (2005) used a modeling approach to evaluate the effects of deforestation on ET in the southern U.S. [26]. They found that deforestation could result in a decrease in ET and increase in water yield, up to 45 cm/yr. In general, removal of evergreen forest resulted in a greater decrease in ET, because deciduous forests were assumed to use approximately 20% less water than coniferous forests.

3.2. Urbanization Effects on Stream Hydrology and Peakflows

Numerous studies of urbanization effects on stream runoff have been conducted in the southern U.S. (Table 1). Much of this research has been focused within the Piedmont setting, particularly in the Washington DC-Baltimore, MD and Atlanta, GA metropolitan areas. Urban areas studied typically consisted of greater than 5% total impervious area (TIA).

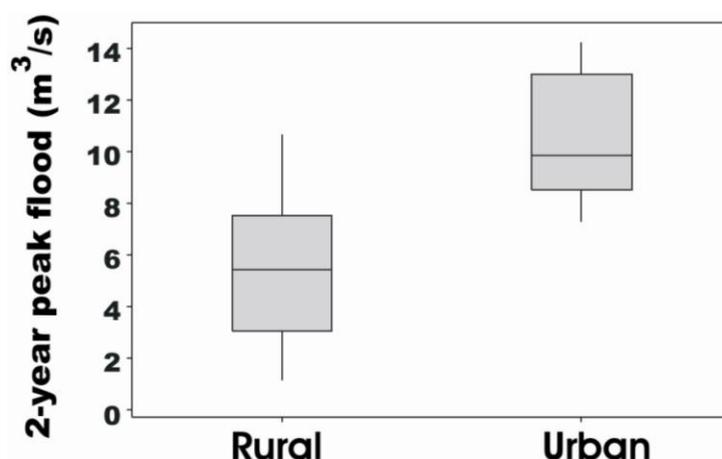
Increased stream stage and discharge variability are common responses to urbanization in the southern U.S. (Table 1). Generally, urban streams behave in a more flashy fashion than their rural counterparts and have a greater occurrence of extreme flow events (flows that are three or more times larger than median flows). Urbanization in the region consistently increases runoff ratios (precipitation/streamflow) and peakflows. Peakflows up to 400% higher than pre-development have been reported [27].

Changes in peakflows due to urbanization vary with the degree of urban development, recurrence interval of the peakflow, and location within the watershed. Empirical urban peak flow regression models have been developed for the majority of the southern states by the U.S. Geological Survey's National Streamflow Statistics Program [28]. Similar to the studies presented in Table 1, model simulations demonstrated consistent increases in peak stream flows in urban catchments when compared to rural catchments in similar physiographic settings throughout the southern U.S. (Figure 4). For shorter recurrence intervals (1–2 yrs), the increase in peakflow due to urbanization is more pronounced than for longer recurrence interval peakflows (10 yrs or greater) [29].

Although the general pattern of increased peakflows associated with increased TIA has been well-established, the spatial variability of peakflow increases due to urbanization is less understood. In a modeling study, in Watts Branch watershed (MD), urban peakflow (2 yr) increases were shown to vary significantly within the watershed [30]. Urbanization resulted in a doubling of peakflows at the

watershed outlet. However, throughout the drainage network, at discrete locations along the channel, the peakflow increases could range from 1.25 to 3 times the pre-urban levels. This within watershed variability may be partially explained by the fact that peakflow increases along stream channels directly downstream from confluences. Urban development in the southern U.S. has consistently resulted in increased urban runoff (Table 1), but caution should be taken when analyzing long-term trends. For example, Olivera and Defee (2007) demonstrated that in the Houston, TX area, changes in rainfall distribution over time may also contribute to trends of increased urban runoff [49].

Figure 4. A boxplot (box represents, 25th, 50th, and 75th percentiles) comparison of urban and rural 2 year peak streamflows for 10 small watersheds (1 mi² or 2.6 km²) in 10 states (AL, DE, FL, GA, KY, LA, NC, SC, TN, TX) within the southern U.S. (using the National Streamflow Statistics Program (Turnipseed and Ries, 2007)). Simulated watershed impervious area was 25% TIA.



Most urban runoff studies in the southern U.S. have focused on the hydrological response to increased impervious area (Table 1), but many recent studies have focused on the hydrological response to parameters such as wetland extent and alteration, tree cover, style of development, and connectivity of impervious area. In FL and TX, Brody *et al.* (2007) showed that the degree of wetland alteration (as indicated by the number of wetland permits) in a watershed may also alter a stream's hydrological response to rainfall and can contribute to increased flood frequency [50]. Wang *et al.* (2008) evaluated the importance of urban tree cover on peakflows [31]. Through model simulations, they found that increased tree cover could reduce peakflows by up to 12%. Trees in urban catchments could intercept up to 41% of the precipitation input during storm events. In addition to vegetation, the connectivity and density of urban roads, infrastructure, and stormwater networks has been shown to influence peakflow responses. In a modeling study, Bosch *et al.* (2003) evaluated the effects of a variety of low and high density development scenarios on watershed hydrology [29]. They found that low density development had the greatest hydrological impacts on peak flows at the watershed-scale because of the greater per capita impervious area. Effective impervious area (EIA, impervious area directly connected to the stream) has been shown to be a robust metric of the urbanization effect on stream hydrology [51,52] in Australia and other settings. This is an emerging area of focus, particularly in the rapidly urbanizing areas in the southern U.S. where research has shown the

importance of the spatial connectivity between impervious areas and streams [37,45,53]. Impervious surfaces allow runoff and associated pollutants to flow through piped systems directly to the stream, even during small storm events. By sending this water through stormwater BMPs and riparian zones for infiltration, evaporation or pollutant removal (e.g., sedimentation), urban streams can more closely mimic natural hydrologic conditions. Effects of disconnection would likely follow a threshold type response in which interception efficiency decreases as storm size increases. However, the strength and shape of this correlation is largely untested. This has great implications for management of urban watersheds, particularly as restoration efforts seek to reestablish lost hydrologic and ecosystem functions.

3.3. Urbanization Effects on Baseflows and Groundwater Recharge

It is frequently presumed that urbanization reduces groundwater recharge because increasing impervious surface area reduces infiltration and increases stormwater runoff [54]. Indeed, baseflow declines with increased impervious area have been measured and modeled within the southern U.S. [32,41], but other studies reveal that urbanization redistributes and often increases recharge [23,55]. These contradictions demonstrate that watershed hydrologic impacts of urbanization are not simply a straightforward function of impervious area, and argue for the need to develop a better mechanistic understanding of the human and landscape controls on urban groundwater recharge and discharge to streams (Figure 5).

Data and modeling results illustrate the range of variation of baseflow response to urbanization. In Atlanta, GA, Rose and Peters (2001) [41] contrasted baseflow recessions in urbanized *versus* less urbanized streams. Low flow declines of approximately 30% were attributed to urbanization and corresponding declines in groundwater levels indicated a decrease in upland recharge associated with an increase in TIA. Using stable isotope hydrograph separations along the Econlockhatchee River, FL, Gremillion *et al.* (2000) [46] showed a change in groundwater inputs occurred downstream as land-use changed from rural to suburban, suggesting an alteration to flowpaths and decreased groundwater inputs associated with greater impervious area. In cases where total annual streamflow increases due to urbanization, it is possible that even though the percentage of total annual discharge comprised of baseflow declines, the total annual volume of baseflow could increase (Hardison *et al.* 2009) [38]. Similarly, in a watershed hydrologic modeling study in coastal NC, Qi *et al.* (2009) found that moderate urbanization ($\leq 10\%$) has the potential to increase total streamflow ($>20\%$) [56]. A modeling study in the coastal Roanoke River Basin, VA, estimated a 12% watershed decline in groundwater recharge as a result of suburban development [29], but such watershed models frequently presume that impervious surface area allows no groundwater recharge and do not account for changes to recharge mechanisms and sources that accompany urbanization.

Recharge in urban areas occurs by one of four mechanisms: direct, localized, indirect, and artificial. Direct recharge happens when infiltration occurs at the point where precipitation hits the ground, typically observed in parks and lawns. Localized recharge occurs where water moves laterally for a short distance before infiltrating in a small-scale feature. In urban areas, such features might include pavement fractures, swales adjacent to uncurbed roads, and gutter downspouts. Larger scale recharge is termed indirect recharge and occurs when lateral movement brings water to infiltrate from within a

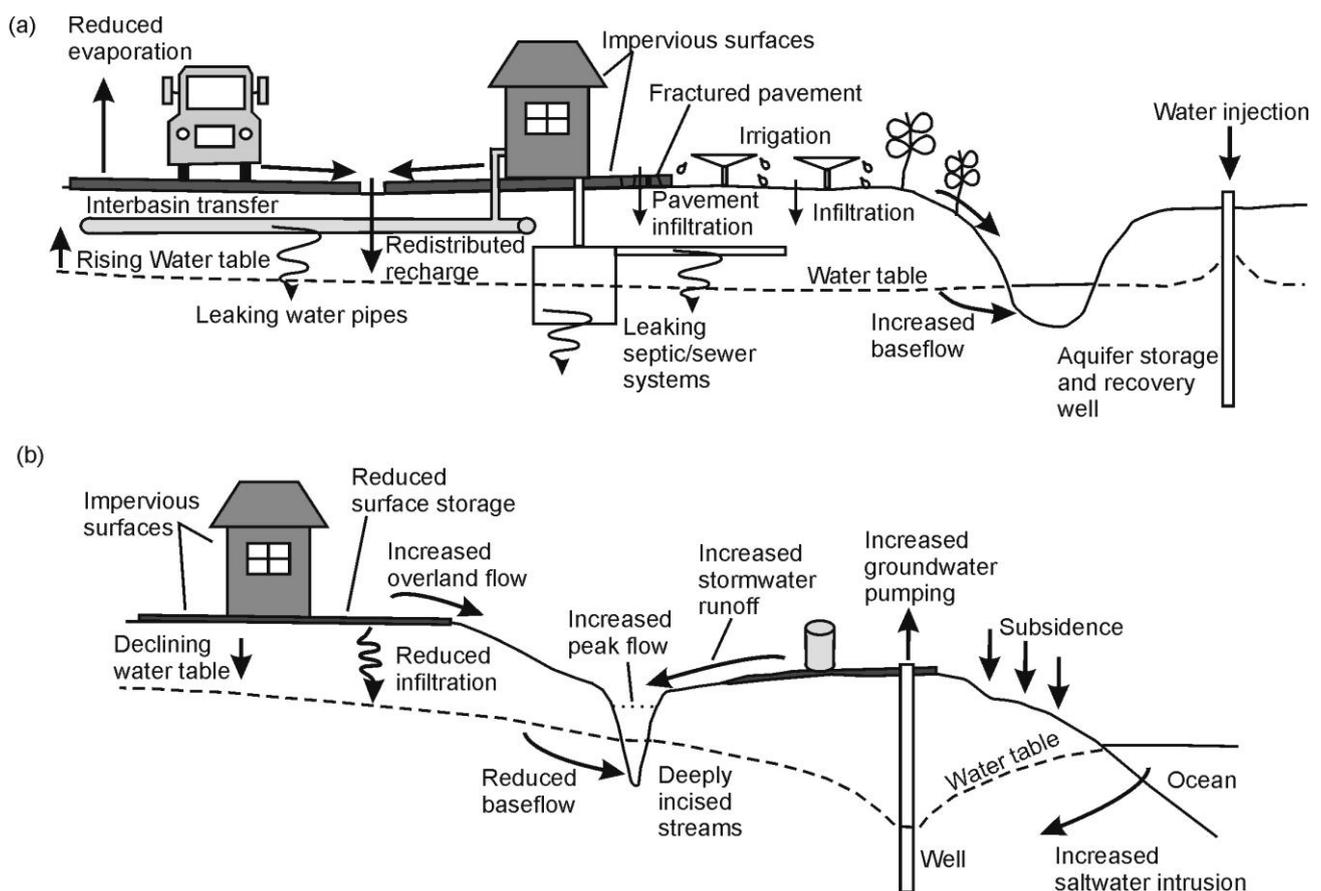
mappable feature such as a sinkhole or lake. In the urban environment, these features include stormwater infiltration basins and detention ponds. Finally, leaking water mains and sewer pipes create a novel artificial mechanism for recharge [55,57]. Quantifying each of these recharge mechanisms or their collective influence at the scale of a metropolitan area is a challenge because multiple mechanisms and land uses are superimposed on pre-existing soil and rock heterogeneity and surface drainage systems, resulting in extreme heterogeneity in the spatial distribution of urban groundwater recharge (Figure 5).

While urbanization is generally assumed to lower recharge, a study conducted in Austin, TX illustrates how urbanization can actually increase recharge through leaky water infrastructure and irrigation [23]. The city sits above the Edwards Aquifer, one of the most productive aquifers in the U.S. Austin grew rapidly through the latter half of the 20th century and had a population of >650,000 in the 2000 Census. A spatial analysis in GIS combined with a water balance method based on municipal water use data and reference evapotranspiration rates were used to estimate the effects of urbanization on various sources of groundwater recharge. Pre-urban direct recharge was estimated at 53 mm/yr, whereas post-urbanization (year 2000) direct recharge was estimated at 31 mm/yr. By comparing the supply of drinking water to the amount of wastewater treated each year, Garcia-Fresca (2006) estimated that 85 mm/yr of treated water was potentially available for recharge [23]. Approximately 31 mm/yr recharges groundwater through leakage from water supply and sewer lines, and the remaining 54 mm of water is assumed to be used for irrigating lawns and parks. Of the irrigation water, 22 mm/yr was estimated as evapotranspiration, leaving 32 mm/yr as recharge from irrigation return flow. While direct recharge was considerably less following urbanization (31 mm/yr compared to 53 mm/yr), combined urban groundwater recharge was 94 mm/yr from direct recharge, utility leakage, and irrigation excess [23].

Decreased direct groundwater recharge is also the most commonly cited cause for lowered stream baseflow in urban streams, as a result of increased TIA [54]. However, recharge from impervious surfaces (e.g., rooftops and paved surfaces) may indirectly contribute to urban recharge. Rooftops preclude direct recharge, but water is routed to gutters and downspouts which often empty into drainage swales or pervious lawns where substantial infiltration may occur as localized recharge. Data on the amount of groundwater recharge generated from rooftops is unavailable for any sites in the southern U.S., but could approach 100% of precipitation if infiltration was rapid enough to limit evapotranspirative losses. Pavements on roadways and parking lots are riddled with fractures that form preferential flow paths for indirect recharge. Pavement infiltration rates of 1.8 to 27 mm/hr and infiltration of 6%–41.6% of rainfall have been recorded in irrigation and infiltrometer studies compiled by Wiles and Sharp (2008) [58]. In Austin, TX, double ring infiltrometer measurements of concrete and asphalt pavement fractures were used to estimate whole pavement hydraulic conductivity of 6×10^{-5} cm/s [58]. Using that hydraulic conductivity and the percent of the time the pavement is saturated, it was estimated that 21% of mean annual precipitation is available for recharge. This equates to 170 mm/yr estimated for “impervious” pavements only, which is greater than the overall recharge (94 mm/yr) for Austin estimated by Garcia-Fresca (2006), who made the conservative assumption that impervious surfaces did not contribute to groundwater recharge [23]. Further, impervious surfaces have much lower evaporation rates than evapotranspiration rates from vegetated surfaces, leaving most of the precipitation to runoff or infiltrate, and pavement studies have shown infiltration rates to be much larger than evaporation rates [59]. Finally, as illustrated in the Austin

water budget case study, over-irrigation of lawns and gardens can increase direct recharge in vegetated parts of the landscape [23]. Thus, the available data suggest that impervious surfaces may contribute more than conventionally assumed to localized groundwater recharge through the routing of water toward pervious areas or fractures. Further work is needed to determine the city-wide significance of these phenomena and how their importance varies across the landscapes, climates, and styles of urban development in the southern U.S.

Figure 5. (a) Factors that may contribute to increased groundwater recharge and/or increased baseflow in urban settings. (b) Factors that may contribute to decreased groundwater recharge and/or decreased baseflow in urban settings (Modified from Welty *et al.* 2007) [22].



In the urban environment, indirect recharge occurs through basins and devices designed to detain and retain runoff from impervious surfaces before it enters the stream. Collectively known as stormwater best management practices (BMPs), these devices include wet detention ponds, filter strips, and bioretention cells. BMPs are ubiquitous in urbanizing areas of the southern US and are often a requirement for new development. In many cases, the primary purpose of a BMP is to reduce peak flows in urban streams, often with the goal of replicating pre-development hydrographs. Peak flow reduction can be accomplished by detaining water for slow release to the stream, increasing evapotranspiration, and increasing infiltration. The relative efficacy of each of these mechanisms depends on BMP design and local conditions. There have been many studies on BMP performance with sites in the southern U.S., and several of these studies have quantified infiltration from BMPs of

various designs. For example, bioretention cells in MD and NC allowed 8% of inflow to infiltrate the subsurface and 19% of inflow to evapotranspire, as measured over a series of precipitation events [60]. Similarly, a level spreader-vegetated filter strip system in Charlotte, NC was effective at eliminating runoff from storms producing less than 13 mm of precipitation. Presumably some fraction of the flow into the BMP infiltrated the ground, but the relative rates of infiltration *versus* evapotranspiration were not measured. As these examples show, research is often limited to inflow-outflow monitoring at individual BMP sites, and more work needs to be done to understand the landscape, climatic, and engineering factors controlling groundwater recharge from stormwater BMPs at scales from the individual BMP to the metropolitan area.

Artificial groundwater recharge takes the form of leaking water supply and sewer pipes. Pipe leaks release water that is usually imported into the watershed where the leak occurs and represent a source and quantity of groundwater recharge that would be unavailable in undeveloped watersheds. Leakage rates for water mains range from 5% to >60% of the supplied water, and water main leakage rates for five TX cities ranged from 8.5% to 37% [57]. Leakage from water distribution systems occurs because flow within the pipes is pressurized and it represents a source of high water quality recharge to the aquifer. Leakage from wastewater collection systems is generally lower in quantity than leakage from water mains, but it can be a much more significant water quality problem for cities. Unlike water supply systems, sewer lines are generally not pressurized. This results in sewer pipes above the water table contributing to recharge and impairing water quality, while those below the water table experience water leaking into them [55]. Leakage rates for water distribution systems can be estimated from minimum nighttime flows and such data may be available for southern U.S. cities from municipal water supply agencies. However, leakage rates from wastewater collection systems are more difficult to determine, and a value of 5% of sewage flow is often assumed [55]. Of course, unsewered cities or portions of cities, including those areas with septic tanks, contribute most of their supplied water to groundwater recharge. To generate estimates of the rates of artificial groundwater recharge in southern U.S. cities, researchers will need to consider the age and condition of water infrastructure, depth to the water table, and proportion of the population served by septic systems. In addition, aquifer storage and recovery (ASR) projects, which can increase local recharge are becoming increasingly common in urban settings in the southern U.S., particularly in FL [61].

In addition to changes in recharge and discharge dynamics, urbanization can have other important impacts on groundwater systems, each with consequences for streams. These impacts have been divided into seven categories: overexploitation; subsidence; saltwater intrusion; contamination; changes in recharge and discharge; alteration of the permeability structure; and destruction of environmental resources, including wetlands and streams [57]. Each of them has been documented in the southern U.S., with the types and scopes of the impacts being partially determined by the geographic setting of the urban area. For example, along the southern US coastline, the Houston-Galveston metropolitan area has experienced land subsidence [62] while saltwater intrusion is occurring in Fort Lauderdale, FL [63], Tampa and Miami, FL, Savannah, GA, and Hilton Head Island, SC [64]. Changes to shallow groundwater and stream baseflow chemistry consistent with wastewater and fertilizer contamination have been observed in moderate relief areas like Austin [65] and Atlanta [66].

4. Urbanization and In-Stream Processes

4.1. Urbanization Effects on Channel Geomorphology and Sediment Transport

Changes in land-cover associated with urbanization alter surface and subsurface flowpaths and the transport of water and sediment to stream channels, which in turn can alter the channel's geomorphology. Some of the earliest work on the effects of urbanization on stream channel morphology was focused in the Baltimore, MD and Washington DC area (Piedmont setting) [67,68]. This geographical region has been the focus of the majority of published studies addressing the effects of urbanization on stream channel geomorphology in the southern U.S. (Table 2) over the last four decades. In his classic study in the Piedmont of MD, Wolman (1967) showed a large increase in sediment yield when bare surfaces were exposed during urbanization [67]. As the urban landscape was built out, the sediment yield declined. Increased runoff due to greater impervious areas and the general decline in sediment yield following urbanization resulted in urban channel erosion. In this case, it was shown that urban channels tended to widen as a result of increased peakflows and a reduction in sediment inputs [67]. Along Watts Branch, MD, Leopold (1973) documented the changes in channel form and floodplain deposition over a period of 20 years during watershed urbanization [68]. This work showed the temporal variability of a stream channel's response to urbanization and the importance of long-term channel morphology and hydrological monitoring to better understand the timing and duration of the urban response and mitigate potential damages.

Table 2. Stream channel geomorphological responses to urbanization documented in the southern U.S.

Study Area/ Physiographic Setting	TIA or urban land use	Effects of urbanization on channel morphology	Enlargement of cross- sectional area	Ref.
Baltimore, MD/Piedmont	Urban (extent N/A)	Increase in channel widths.	N/A	[67]
Watts Branch, MD/ Piedmont	Up to 5 fold increase in houses	The net result of urbanization on channel dimensions after 20 years was a smaller channel, however the trend towards the end of the study was towards channel enlargement. Floodplain deposition ranged from 0.6–1 ft /13 yrs.	Early period of channel aggradation and decrease in channel cross-sectional area followed by a period of channel enlargement.	[68]
PA, MD, DE/ Piedmont	Up to 66% TIA	Increase in channel width and cross-sectional area for urban streams, wider channels and greater cross-sectional areas along forested reaches indicating the importance of vegetation to channel form.	Urban reaches had greater width and cross-sectional area but not depth.	[69]
NC/Piedmont	Up to 80% TIA	Increase in channel width, depth, and cross-sectional area.	2.6 times larger than rural.	[70]
NC/Coastal Plain	Up to 67% TIA	Increase in channel width, depth, and cross-sectional area.	1.8–3.4 times larger than rural.	[71]

Table 2. Cont.

Study Area/ Physiographic Setting	TIA or urban land use	Effects of urbanization on channel morphology	Enlargement of cross- sectional area	Ref.
NC/Coastal Plain	Up to 37% TIA	Increase in channel incision ratio and channel depth.	N/A	[38]
Baltimore, MD/ Piedmont		Approximately 20% loss of stream length due to burial for the broad study area, in Baltimore City 70% of stream length in catchments <260 Ha were buried.	Decrease in bankfull cross-sectional area due to burial by infrastructure.	[72]
OK/Great Plains	Up to 12% TIA	Lack of urban response attributed to geological (cohesive channel sediments and shale and sandstone bedrock) and hydrogeological controls (groundwater).	N/A	[73]
Knox Co. TN/Valley and Ridge	Urban (extent N/A)	Increase in bed particle size and anthropogenic particles. Propagation of channel changes is often prevented by infrastructure.	Majority of urban reaches undergoing channel enlargement.	[74]
Fayetteville, AR/Ozark Mountains	Urban (extent N/A)	Decrease in channel depth.	N/A	[75]
Baltimore, MD/Piedmont	Up to ~32% TIA	Channel reaches were classified as aggrading, early erosional, or late erosional stage. Aggradational sites (7/19) tended to show a decrease in cross-sectional area from 1987–2000, whereas erosional sites tended to show channel cross-sectional area increases of 15–24% over the same period.	Channel cross-sectional area enlargement, channel incision, and reduced channel width variation were common at erosional sites.	[76]
Montgomery Co, MD/ Piedmont	Up to 20% TIA	Erosion from headwater channel enlargement due to urbanization provided approximately 40% of watershed sediment yield.	In catchments without retention basins urban channels were enlarged 2.1–2.5 times greater than rural channels.	[77]
Baltimore Co., MD/ Piedmont	Up to 82% urban	Shear stress and stream power were greatest at high gradient meander bends, however in this urban catchment changes in hydrology have not caused significant migration of the channel since the 1930s.	N/A	[78]
Rockville, MD/Piedmont	~556 homes/mile ²	Increase in channel grain size, increase in floodplain deposition rates (during the first several decades of urbanization), increased channel width, increased w/d ratio, bed aggradation, and lateral migration of channels.	Increase in cross-sectional area due to widening (decrease in channel depth).	[79]
Dallas, TX/Blackland and Grand Prairie	Up to 33% TIA	Channel erosion of up to 4 inches per year in gravel or rock bottom channels.	N/A	[80]

Although many studies have evaluated the effects of urban impervious area on stormwater runoff and channel geomorphology (Table 2), fewer studies have focused on the influence of urban vegetation. In southeastern PA, northern MD, and DE, Hession *et al.* (2003) evaluated the influence of riparian

vegetation on channel dimensions along urban and rural stream reaches [69]. Although the presence of riparian vegetation did not appear to influence channel depth, forested reaches in both urban and rural settings were consistently wider than non-forested reaches, suggesting an influence of riparian vegetation on channel dimensions. The authors suggest this widening is linked to the riparian vegetation influence on floodplain accretion processes.

Numerous studies have focused on using a time-for-space substitution to assess the effects of urbanization on stream channels [69-71], using watershed impervious area (or extent of urban land use) as a controlling variable that can affect hydrological and sediment transport processes that in turn influence channel form. However, the extent of watershed TIA does not provide information on the temporal and spatial patterns of urbanization and the connectivity of impervious area to stream channels (EIA) that may affect changes in discharge, sediment transport, and channel form [52]. Fewer studies have documented changes in channel form along urban streams over long time periods capturing pre- and post-urbanization conditions. Leopold *et al.* (2005) documented changes to Watts Branch (MD) due to urbanization over a four decade period [79]. They noted an increase in channel grain size, increase in floodplain deposition rates (during the first several decades of urbanization), increased channel width, increased width/depth ratio, bed aggradation, and lateral migration of channels in response to urbanization. However, because of the dynamic and interactive nature of these changes, the authors concluded that 40 years may not be sufficient to observe the complete response to urbanization and more long-term monitoring is essential to improve understanding of the urban channel response cycle.

Colosimo and Wilcox (2007) looked at changes in stream channel form due to urbanization in the Baltimore area over a shorter period (1987–2000) [76]. Although most of the 19 channel reaches studied were experiencing erosion and channel enlargement, five reaches were experiencing aggradation and reduction in cross-sectional area. Aggradation was more common in subwatersheds that recently had large increases in the extent of impervious area and at sites downstream from erosional reaches. In addition, vegetation, slope, infrastructure, and natural bedrock controls influenced channel aggradation or degradation. Overbank flows were less common along erosional reaches compared to aggradational reaches. Although a general urban response of aggradation followed by erosion is expected, as illustrated by Leopold (1973), a significant spatial and temporal variability of these processes can occur along a single channel [68]. This is linked to the complex influence of urbanization on water and sediment supply to the channel, changes in vegetation and infrastructure, and their variations within the watershed. In another long-term study, Allmendinger *et al.* (2007) approximated a sediment budget for an urbanizing watershed in Montgomery Co, MD for the period of 1951–1996 [77]. They found that upland erosion, channel erosion, and floodplain storage were of similar magnitude and were all important components of the urban sediment budget. Management of stormwater within the catchment greatly influenced channel size as demonstrated by channels without stormwater retention basins observed to have channel cross sections >2 times larger than their rural counterparts. The amount of sediment generated from urban channel erosion was similar to that generated from upland erosion. Overall, urban watershed sediment yield was greater than that observed for forested watersheds but similar to other rural watersheds in the region.

As urban channels adjust to changes in flow and sediment transport, the channel may migrate and enlarge [68,79]. However, in some cases geological controls may limit the migration and enlargement

of urban channels. The geomorphic effectiveness of floods to alter channels is directly related to the shear stress, stream power, and the resistance of the channel to erosion [78]. Nelson *et al.* (2006) used 2-D hydraulic flow models to account for the spatial and temporal variability in shear stress and stream power in the urban Dead Run catchment in Baltimore, MD during flooding associated with Hurricane Agnes (1972) [78]. They found the greatest shear stresses and stream powers were associated with high gradient meander bends. However, geological controls in this urban watershed including bedrock outcrops and coarse bed and bank materials limited channel erosion. Measured channel migration was minimal since the 1930s. A similar lack of urban channel response was documented in the Great Plains of OK by Kang and Marston (2006) [73]. They looked at the urban channel response in watersheds with up to 12% TIA and did not find any significant changes associated with urbanization. They attributed the lack of response to geological controls including cohesive sediment, resistant bedrock, and groundwater input effects on channel dimensions. These studies suggest that measureable changes in channel form in response to urban runoff may be less likely along bedrock stream channels. However, work in North TX suggests that (shale and limestone) bedrock channels may also exhibit enlargement due to urbanization [80]. Their study showed that 35% of bedrock channels were experiencing moderate to high erosion. In addition, they suggest the type of bedrock can influence erodibility; shale was found to be more erodible than limestone.

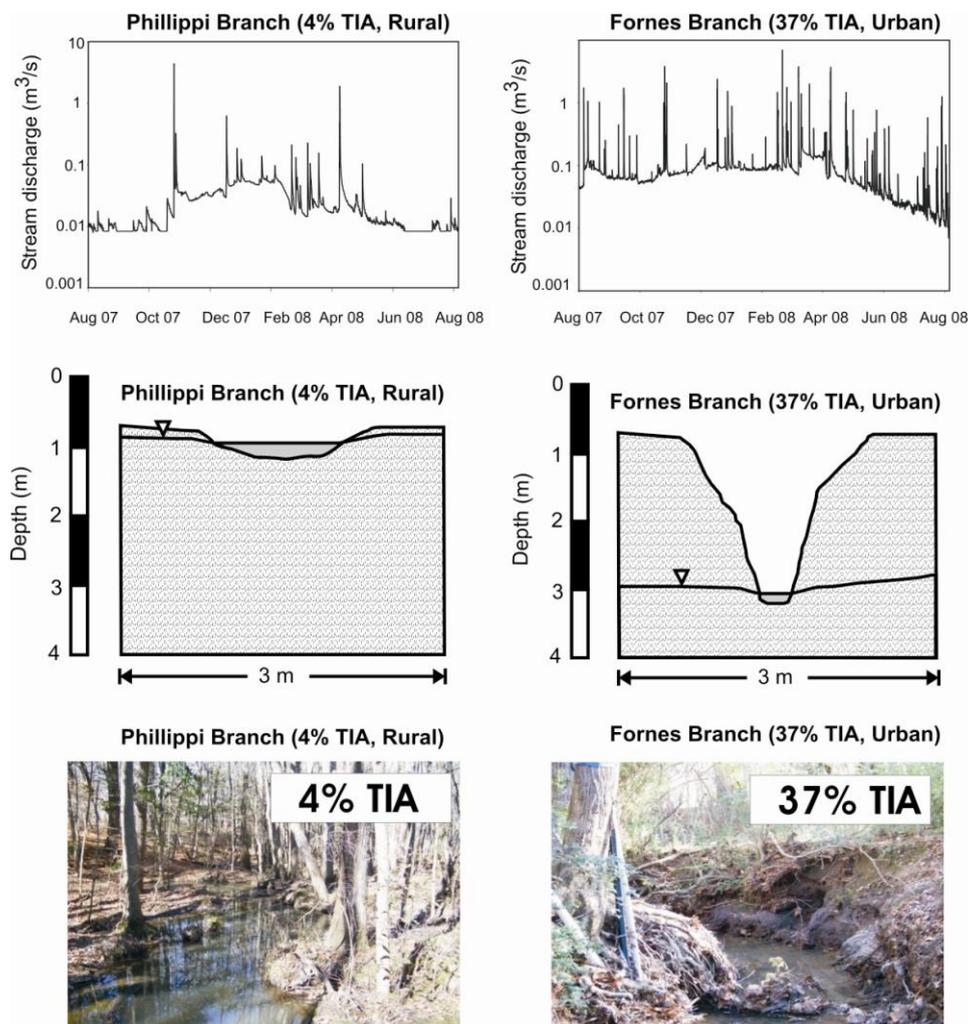
In addition to work in the MD Piedmont region, Doll *et al.* (2002) performed a comparison study of urban and rural channels in the NC Piedmont [70]. Their work showed that urban channels were larger (approximately 2.3 times larger), wider, and deeper than rural channels. The channel widening response was generally larger than the channel downcutting response to urbanization. This is in general agreement with the research in the Piedmont regions of MD and Delaware previously mentioned [67-69]. Channel-widening in response to increased urban runoff may be common in the Piedmont due to the presence of bedrock at shallow depths and bed armoring, which can limit channel incision [81]. However, the silt-clay content of channel banks may also influence the channel erosion response [82].

In contrast, in the Coastal Plain of NC, where streams frequently have sand beds and low-gradients, channel incision has been documented where erodible sediment (sand, silt, clay) underlies channels [38,71] (Figure 6). Urban channel cross-sectional areas for low order Coastal Plain streams in the Greenville, NC area were larger directly downstream of stormwater culverts. A comparison of 20 urban and 20 rural channels revealed that urban channels were approximately 2–3 times larger than their rural counterparts. Watershed TIA explained 65%–72% of urban channel enlargement [71]. In a study of 6 low-order Coastal Plain streams, watershed TIA was found to explain 79% of the variability in stream channel incision [38]. In the most urbanized watershed studied (37% TIA), channel incision of 1.28 m was measured over a 33 year period (1975–2008). These studies suggest that urban channel incision may reduce stream-riparian zone interactions and lower groundwater tables along urban Coastal Plain channels. Groffman *et al.* (2003) documented similar urban channel incision in the Baltimore, MD area (Piedmont) and coined the term “urban riparian drought” to describe the drier riparian zones adjacent to incised urban channels [83].

Fewer studies have addressed stream channel responses to urbanization in mountainous regions of the southern U.S. Although most urban channel studies in the southern U.S. have shown channel enlargement, Keen-Zebert (2007) showed a decrease in channel depth and the common occurrence of aggradation along stream reaches in the Fayetteville, AR region (Ozark Mountains) over an 18 month

period [75]. This was attributed to the fact that much of the urban development in the region was more than 20 years old and bridges, infrastructure, and large woody debris helped promote aggradation. As pointed out by Leopold (1973) the aggradation response may be spatially and temporally variable and time periods of several decades or more would be needed to observe the complete urban response cycle [68]. In an Appalachian Valley and Ridge setting (Knox Co., TN), Grable and Harden (2006) studied the geomorphic response to urbanization [74]. They found that although erosion was the dominant response to urbanization, aggradation could also occur and channel response patterns were complex related to the temporal and spatial variability of urban development and infrastructure. Urban infrastructure and channel alterations could prevent the propagation of channel responses upstream (*i.e.*, upstream migration of erosional knickpoints). In addition, increases in channel sediment grain size and particles of anthropogenic origin were noted. The inputs of anthropogenic particles were geomorphically significant, as they comprised up to 21% of channel sediments.

Figure 6. A comparison of the annual hydrographs of rural (Phillippi Branch, 4% TIA) and urban (Fornes Branch, 37% TIA) streams in the Coastal Plain of NC (Greenville, NC) and the stream channel cross-sections, indicating the channel incision and enlargement response to urbanization documented in the region [38,71]. Photo of Phillippi Branch is courtesy of Dr. Mark Brinson.



It is well-documented that urban infrastructure alters sediment and water transport to urban streams and these changes can bring about changes in channel dimensions, channel migration, and bed sediment grain size. In addition, urban stormwater can alter the channel by reducing channel heterogeneity resulting in a decrease in the diversity of aquatic habitats [84]. Recent work also documents that urban infrastructure can directly and permanently alter the stream network via urban stream burial. Elmore and Kaushal (2008) found that approximately 20% of the stream network was buried by infrastructure in the greater Baltimore area [72]. In Baltimore City, 70% of stream channels in catchments <260 Ha were buried, and in the District of Columbia, 75% of pre-1880's streams have disappeared [85]. Similarly, in Greenville, NC, Hardison *et al.* (2009) found that drainage density in urban catchments was approximately 40% lower than surrounding rural catchments, due to urban stream burial [38]. Urban stream burial, particularly for headwater streams, can affect sediment, nutrient, and water transport to downstream segments, and may be a common occurrence in the southern U.S. More research is needed to document how urban stream burial varies across the southern U.S. due to differences in geology, topography, forest cover, and urban development practices.

Urban infrastructure that regulates in-stream flows can also influence stream hydrology and geomorphology. Dams and water control structures are a common feature of the urban landscape. They are ubiquitous throughout the southern U.S. and affect most drainage basins. The 2009 National Dam Inventory (US Army Corps of Engineers) estimates there are over 35,000 dams in the southern U.S. with many located in urban/suburban areas [86]. Dams in the southern U.S. have been installed for a range of purposes, including recreation, flood control, water supply, hydropower, and navigation. In a census of American dams, Graf (1999) showed that the hydrology of the southern U.S. is highly affected by dams [87]. In a comparison of the volume of water storage by dams across the U.S. he estimated that the South Atlantic region had the greatest water storage per unit land area in the entire U.S. ($345,000 \text{ m}^3/\text{km}^2$). In addition, there are 21 dams in the region with individual capacities of over 1.5 km^3 of storage (Graf 2006) [88]. Dams can have a major influence on downstream hydrology by reducing annual peakflows, decreasing the range of annual discharge, and altering the timing of annual maximum and minimum flows [88]. These hydrologic alterations can in turn modify sediment transport and channel morphology downstream. In a study of 36 paired river reaches (upstream and downstream of dams) adjacent to large U.S. dams, the regulated reaches downstream of dams had larger low flow channels, decreased high flow channels, decreased active floodplain areas, increased extent of inactive floodplain areas, and decreased geomorphic complexity along the channels [88]. It was concluded that the simplified and contracted geomorphology downstream of dams had a direct effect on aquatic ecology as a result of reduced extent and diversity of aquatic habitats and less diverse riparian ecosystems.

4.2. Urbanization Effects on Water Quality

Water Temperature. Hydrologic and geomorphic changes as a result of urbanization can have a direct impact on the thermal regime of urban streams by changing the stream's energy balance. Changes to riparian vegetation, channel geometry, low flow regimes, and temperature of water inputs all influence stream temperature [89]. Where riparian buffers are not protected, clearing of vegetation reduces channel shading, thereby increasing shortwave radiation during the day and reducing

longwave radiation during the night. Heat exchanges at the air-water interface can be amplified in urban streams with increased channel width resulting in increased diurnal stream temperature changes. Lowered baseflows generally observed in urban systems compound the effect as less water is available to absorb incoming energy [89]. The temperature of water inputs to a stream can also change because of discharges of industrial or wastewater treatment effluents and from heating of runoff that flows over impervious surfaces. The typical result of these combined impacts of urbanization is to increase stream temperature during baseflows and some peakflows. Elevated temperature has been shown to enhance rates of biological processes resulting in cascading changes to urban stream ecosystems [89-91].

Data on the effects of urbanization on stream temperatures primarily comes from regions with cold water streams, where thermally-sensitive trout and salmon are found. For example, in the Blue Ridge Mountains of western NC, urban streams had a 3 °C greater late summer diurnal temperature range than forested streams [92]. In the southern U.S., where lowland streams are generally warm water ecosystems, increased stream temperatures could lead to an increased frequency of hypoxic or anoxic conditions during summer months. Model simulations of urban development in a VA Piedmont watershed predict the frequency of maximum temperatures exceeding the state water quality standard to increase from 1.1% to 7.6% of summer days [93].

Urbanization can also affect peakflow temperatures, particularly when storm runoff results from the summer convective thunderstorms common in the southern U.S. Temperature of runoff from paved surfaces is a function of surface properties, rainfall temperature, air temperature, and solar radiation prior to the storm [94] with temperatures and solar radiation maximized before late afternoon, summer storms. In urbanized watersheds in the MD Piedmont, temperature surges associated with summer thunderstorms averaged 3.7 °C and receded back to pre-storm temperatures over 2.8 hours [95]. The maximum temperature surge they recorded was 7.4 °C, with a maximum temperature recession time of 7.6 hours. The frequency of such temperature surges was correlated with deforestation and impervious surface cover in the watersheds [95]. There is very limited data on the effects of stormwater BMPs on stream temperature, but a study in MD showed that dry detention ponds increased stream temperature by up to 2.8 °C [96].

Water Quality. Several excellent reviews describe the effects of urbanization on stream water quality and ecosystem processes across multiple regions [5,6,97-99], thus we will focus on studies throughout the southern U.S. that describe current trends and identify opportunities for improving our understanding of the complex feedbacks in urban ecosystems. The impacts of urbanization to water quality are highly variable and depend upon multiple factors including the age/type of urbanization (established urban core compared to suburban development), presence of concentrated *versus* distributed wastewater treatment, stormwater infrastructure, legacy land use, vegetation, and hydrologic regime. In the southern U.S., relatively recent conversion of historically agricultural and forested lands to urban land uses has led to increased levels of oxygen demand, conductivity, suspended solids, nutrients, metals, hydrocarbons and a range of organic chemicals [5,100-102]. These increases are a result of point source inputs (e.g., wastewater treatment plant effluent, industrial dischargers) and nonpoint sources (e.g., stormwater runoff, stream bank erosion, failing septic systems, leaking sewer systems). In particular, urban stormwater runoff has been cited as a key contributor to water quality impairments [103,104]. A review by Schueler suggests that evidence of stream quality degradation is first detected at approximately 7% imperviousness (range of 2–15% imperviousness) [103].

Similarly, as part of the National Water-Quality Assessment (NAQWA) Program, the U.S. Geological Survey (USGS) conducted a study of urbanization impacts in nine metropolitan areas throughout the U.S., including four in the southern U.S. (Raleigh, NC; Atlanta, GA; Birmingham, AL; Dallas-Fort Worth, TX) [36]. While concentrations of nitrogen (N), phosphorus (P), pesticides and organic compounds generally increased with increasing urbanization, a key finding was that historical and surrounding land use was an important driver in relative concentrations of pollutants, particularly N and pesticides across land uses. Sources of N and P in urban catchments include wastewater discharge, leaky sewer and septic systems and fertilizer application to lawns. Agricultural inputs of N and herbicides confounded detection of relationships between these pollutants and metrics of the intensity of urbanization. Significant relationships were observed between two commonly detected pesticides (diazinon and simazine) and urbanization intensity. Poly-aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) were also consistently detected in urban streams and bioassays assessing toxicity were strongly correlated with levels of watershed urbanization [105,106]. Sources of PAHs include combustion (vehicle exhaust) and petroleum products (oil spills, leaking tanks) that is transported to streams as stormwater runoff.

In the Atlanta area, Gregory and Calhoun (2007) showed that specific conductance, chloride, and sulfate were elevated which can be attributed to widespread input of electrolytes in wastewater [106]. Similarly, Rose (2002, 2007) showed that groundwater discharging into a local urban stream was characterized by high levels of chloride, sulfate, calcium carbonate, and total dissolved solids (TDS) [66,107]. The author speculated that shallow groundwater chemistry in the region may have been influenced by effluent contamination originating from numerous septic tank systems and leaky underground sewer pipes. Schoonover *et al.* (2005) also found elevated levels of chloride, nitrate, sulfate, TDS, potassium and sodium in baseflows derived from watersheds with greater than 5% TIA in western GA [42].

A study of six urban watersheds in Durham, NC highlighted the importance of multiple indicators of urbanization in addition to TIA [104]. Using multiple linear regression analysis to identify patterns in the distribution of phosphorus, total kjeldahl nitrogen, total suspended solids and fecal coliforms, the authors demonstrated that increasing development density (as TIA) correlated with decreased water quality. However other indicators such as house age, EIA and stormwater connectivity explained additional variation in multiple linear regression models [104]. Peters (2009) supported these results with results from 21 urban streams over a range of discharges [108]. Concentrations of major ions, metals, nutrients and coliform bacteria were generally higher in urban streams compared to nearby forested and low-density reference watersheds. In particular, fecal coliform measurements at all urban sites exceeded GA water quality standards for any water usage category [108].

In a review of 135 observations from a wide range of aquatic ecosystems, watershed land use was shown to directly (erosion and runoff) and indirectly (pH driven metal-DOM interactions) affect metal concentrations in freshwaters [109]. Specifically, Fe, Mn and Cu were consistently high in urban watersheds [Fe (12–58 µg/L), Mn (0.20–30 µg/L) and Cu (0.4–15 µg/L)], with only streams draining mining areas having higher concentrations (Das *et al.* 2009) [109]. Additionally, metal concentrations were inversely correlated with pH indicating that at low pH levels, metal solubility increases and metal particles become more mobile. Elevated concentrations of metals in aquatic ecosystems can have long-

lasting effects on ecological health, however limited research has been done to link water quality assessments with biotic integrity.

The broad category of emerging contaminants (EC) includes compounds such as antibiotics, prescription and nonprescription drugs, steroids, hormones, personal care products, and products associated with oil and combustion. Household chemicals, pharmaceuticals and personal care products can enter urban waters through discharge from wastewater treatment plants (WWTPs) as these compounds travel through the treatment process largely unaffected [110,111]. In a comprehensive sampling of 139 streams across 30 states during 1999 and 2000, the USGS detected 82 of 95 target compounds and found at least one compound in 80% of the streams sampled [111]. Measured concentrations were generally low; however little is known about the potential interactive effects (such as synergistic or antagonistic toxicity) that may occur from complex mixtures of ECs in the environment. Similarly, in a study of the occurrence and fate of antibiotics in Ozark streams, AR, 10 different antibiotics were found downstream of the WWTP discharge [112]. The study measured uptake of these compounds along a 3 km reach and found low, but measurable rates of uptake suggesting that these compounds are not conservative in aquatic ecosystems.

Less research has been focused on water quality in streams in coastal urban areas relative to the work done in large cities further inland. Coastal “blackwater” streams have distinctive water chemistry with high dissolved organic carbon (DOC) and dissolved organic N (DON) concentrations. In SC, DON was lower in coastal streams draining urban areas compared to forested reference sites but remained the dominant form of N [113]. Particulate P was the dominant form of total P, which the authors attributed to phytoplankton biomass from stormwater ponds. In coastal streams near Wilmington, NC, fecal coliform bacteria were sampled as indicators for pathogenic bacteria [114]. Through linear regression analysis, the authors demonstrated that the percentage of the impervious surfaces within the watershed explained 95% of the variability in average fecal coliform abundance.

4.3. Urbanization Effects on Ecosystem Processes

Nutrient Cycling. Nutrient cycling and retention in streams has been extensively studied in forested systems and more recently in streams draining agricultural and urban areas [115-118]. Rapid urbanization and the resulting stream degradation has led to increased attention on the retentive capacity of urban streams. Recent work has focused on the role streams play at the watershed scale in serving as mediators of nutrient transport to sensitive downstream ecosystems and the controls on nutrient retention to minimize or mitigate the effects of urbanization on stream ecosystem function [98,119]. Establishment of two urban long term ecological research (LTER) sites in the U.S. (Baltimore, MD and Phoenix, AZ), is a result of the drive to better understand these dynamic and complex human–environment interactions. Results from the Baltimore Ecosystem Study (BES) in particular have greatly advanced our understanding of urban ecology as integrated social-ecological systems in the southern U.S. As part of the BES, integrated studies of streams, riparian zones, and whole watersheds have highlighted the complex interplay between human influences (e.g., water and wastewater infrastructure, impervious surfaces) and ecosystem processes (e.g., biotic integrity, nutrient retention) [120].

Urban and suburban watersheds tend to function as net exporters of nutrients as a result of high inputs and relatively low in-stream retention [121,122]. In the BES study, urban and suburban watersheds had much higher N losses (2.9 to 7.9 kg N ha⁻¹ y⁻¹) than a completely forested watershed (less than 1 kg ha⁻¹ y⁻¹) [121]. However, retention of N in the suburban watershed was surprisingly high, 75% of inputs, which were dominated by home lawn fertilizer (14.4 kg N ha⁻¹ y⁻¹) and atmospheric deposition (11.2 kg N ha⁻¹ y⁻¹) [121]. Nutrient exports were also shown to vary considerably with climate. Nitrogen retention during the 2002 drought was 85% and 99% for suburban and forested watersheds respectively and declined to 35% and 91% during the wet year of 2003 [122]. The vulnerability of urban watersheds to variable climate and hydrology is particularly important as the frequency of severe weather is predicted to increase as a result of climate change [123].

The trend of lower nutrient retentive capacity in urban streams results from a combination of factors, including (1) increased nonpoint source nutrient delivery through stormwater conveyance systems and wastewater treatment plant effluents; (2) increased flashiness of stream hydrology resulting in scouring of benthic sediments; (3) disconnection between the stream and its riparian zone as the majority of stormwater flow enters streams through drainage pipes rather than sheet flow through riparian vegetation; (4) channelization and clearing of woody debris which reduces hyporheic (subsurface) flow through biologically active sediments and removes organic sources for heterotrophic microbes. However, important physical, chemical and biological functions, including NO₃⁻ removal through denitrification (the conversion of dissolved NO₃⁻ to N₂ during the decomposition of organic matter), biotic assimilation of ammonium (NH₄⁺) and soluble reactive phosphorus (SRP), and settling of nutrients sorbed to sediment particles all act to remove and retain nutrients in urban streams.

In a study of 72 streams in eight regions of the U.S., nitrogen uptake was measured using stable isotope tracer experiments [118,124,125]. The LINX II study included forested, agricultural, and urban streams and included a study site in the Southern U.S. near Asheville, NC. Overall, the streams were generally small (discharge 0.2 to 2681 L s⁻¹) and had a wide range of NO₃⁻ concentrations (0.0001 to 21.2 mg N L⁻¹). Nitrate concentrations were higher in urban streams compared to forested reference sites with corresponding increases in uptake rates (*U*, mass of NO₃⁻ removed from the water column on a per area basis) suggesting that elevated NO₃⁻ stimulated retention and removal of NO₃⁻. The study also determined uptake velocity, *V_f*, which can be described as the demand for a nutrient relative to its concentration and an estimate of uptake efficiency relative to availability. Uptake velocity declined exponentially with increasing NO₃⁻ concentrations [124]. Despite increasing *U*, the efficiency at which biological processes removed NO₃⁻ from the water column at high concentrations decreased, which indicated that across multiple regions, the streams became less efficient at removing NO₃⁻ as concentrations increased. Although the LINX II results included agricultural, forested and urban streams across the entire U.S., the study demonstrated important trends of NO₃⁻ dynamics in urban systems that clearly extend to urban streams in the Southern U.S.

In urban streams in Atlanta, GA, concentrations of NH₄⁺ and SRP were significantly higher in urban compared to forested streams [8]. Similar to the LINX II study, uptake efficiency of NH₄⁺ and SRP measured as *V_f* was variable, but overall significantly lower in streams draining urban areas with high nutrient concentrations compared to those in forested catchments [8]. The lowest NH₄⁺ and SRP uptake velocities were measured in the most urbanized catchments suggesting that reduced efficiency of biotic demand for nutrients decreased the stream's overall retentive capacity. Unlike the LINX II

study, nutrient uptake was not correlated with ambient nutrient concentrations, but was directly correlated with sediment organic matter which decreased with urbanization. Organic matter that is stored in backwater areas and hyporheic zones serves as a source of organic matter for heterotrophic microbes that are responsible for nutrient uptake [99]. Enhancing sources of labile carbon (C) through conservation and restoration of riparian areas while increasing residence time in streams may increase nutrient retention in degraded urban streams.

Forested riparian zones contribute an important source of C to in-stream biota but also improve surface water quality as pollutants in shallow groundwater flow through riparian soils. Nitrate in shallow groundwater flows through C-rich surface soils thereby enhancing denitrification and reducing N concentrations that enter stream networks. Stream incision and lowered groundwater tables in urban areas can lead to drying of riparian soils thus inhibiting interaction of groundwater NO_3^- with denitrifying microbial communities along shallow flowpaths [120,126]. As previously mentioned, results from Baltimore, MD showed that urban streams with forested riparian zones had lower groundwater tables, higher NO_3^- pools and enhanced nitrification rates compared to riparian zones in a reference forested watershed [127]. Nitrification is the autotrophic conversion of NH_4^+ to NO_3^- by microbes and therefore a source of NO_3^- . At all sites, denitrification enzyme activity decreased with depth through the soil profile. The capacity for riparian forests to improve water quality can also be reduced in urban systems because of direct connections from stormwater drainage systems to surface waters and bypassing riparian vegetation [128]. Together, these results suggest that hydrologic factors can reduce the ability of riparian zones to act as N sinks.

Measurements of microbially mediated biogeochemistry have led to important advances in understanding the controls on N retention in urban systems, particularly in the context of restoring ecosystem processes. In a study of four urban and suburban streams in the Baltimore, MD area, Groffman and others (2005) highlighted the importance of streambed heterogeneity to ecosystem function [129]. Denitrification was highest in in-stream debris dams and gravel bars with high organic matter content. In-stream structures also supported higher rates of nitrification (in urban compared to forested streams) suggesting that the balance of these processes is critical to the capacity of urban streams to act as either a source or sink of N. While the structures may enhance hot spots for microbial activity, removal and downstream displacement of these structures during high storm flows is likely. Challenges such as these highlight the need for interdisciplinary approaches to urban stream restoration design [84,130].

Production, Respiration and Organic Matter Retention. Nutrient and carbon cycling are intrinsically linked in stream ecosystems with the inputs of both highly dependent upon hydrologic controls. In the LINX II study, NO_3^- uptake lengths (S_w , mean distance traveled by an ion between release from the sediment and subsequent removal from the water column) shortened with increasing gross primary production (GPP), suggesting that autotrophic assimilation was an important mechanism for NO_3^- removal [124]. Land use increased the predictive power of a structural equation model testing controls on NO_3^- uptake length with higher production and elevated NO_3^- concentrations in agricultural and urban influenced streams [124]. Although urbanization acted to increase GPP through enhanced light availability, it was not sufficient to affect cumulative NO_3^- transport. Higher NO_3^- concentrations resulted in no net effect of urbanization on NO_3^- uptake lengths. When NO_3^- removal via

denitrification was considered separately, ecosystem respiration (ER) was significantly correlated with the fraction of total NO_3^- that was removed from the water column [129].

Organic matter inputs to stream ecosystems are generally separated into particulate (e.g., algal biomass, leaf litter) and dissolved components. Across land uses, dissolved organic matter (DOM) accounts for a significant portion of the total organic matter pool and is derived from groundwater flow through riparian soils, terrestrial leaf litter and to a lesser extent in-stream primary production [99,131,132]. Connectivity between streams and riparian corridors is limited in urban systems as stormwater drainage systems discharge directly to streams and riparian forests are cleared. Riparian forests provide C inputs to streams in the form of leaf litter and coarse woody debris, alter solar radiation and lower water temperatures through shading [5,8,133]. Removal of trees along urban streams can lead to negative impacts on many of these functions. The combined result is an overall decrease in the quantity and quality of C that is both delivered to streams and retained in streams for use by heterotrophic communities during respiration.

The effect of disturbance of the riparian corridor on stream metabolism was investigated by Houser *et al.* (2005) [134]. Metabolism is the measure of balance between autotrophic production and combined autotrophic and heterotrophic respiration of organic matter. Metabolism indicates the relative contributions of autochthonous *versus* allochthonous C which forms the basis of food webs [99]. As such metabolism is a metric that is emerging as an integrated indicator of stream ecosystem health. The effect of intense and localized disturbance on metabolism was investigated in a military training facility in the Fort Benning Military Base in west-central GA. In this heavily forested watershed, localized vegetative clearing and erosion resulted in decreased community respiration which was attributed to sedimentation and burial of organic matter. All streams in the study were net heterotrophic with low primary production as a result of riparian shading. Similar results were shown for a study of four urban and two forested streams near Atlanta, GA with net heterotrophy in all streams [8]. While no direct urbanization effect on metabolism was demonstrated between urban and forested systems, urban systems retained significantly less organic matter as measured by mass of FBOM in benthic sediments and tracer studies with coarse and fine particles attributed to high stream velocities during storm events [5].

High velocities in urban systems can also lead to accelerated rates of leaf litter breakdown [99]. In a study of 12 catchments of differing land use (four forested, 3 agricultural, two suburban and three urban), Paul *et al.* (2006) [135] found some of the highest breakdown rates of chalk maple (*Acer babatum*) in urban streams, but also the lowest fungal biomass. Faster breakdown was a result of physical abrasion and fragmentation from higher stormwater runoff. In a study by Sponseller and Benfield (2001), leaf breakdown rate decreased as the percentage of forested riparian vegetation decreased and highlighted the importance of shredding macroinvertebrates to leaf processing [136]. Results suggested that increased sediment inputs from urbanization and stream degradation may limit the distribution of shredders and therefore decrease leaf breakdown rates. Human activities including elevated carbon dioxide levels in the atmosphere can have far-reaching impacts on carbon quality and food web bioavailability. The effect on organic matter quality, particularly leaf litter, is an increasing area of research. Results from laboratory studies show that plants grown in an atmosphere enriched with carbon dioxide may be more recalcitrant towards microbial decomposition as a result of leaf material higher in phenolic, lignin and C:N ratios [137].

Lastly, wastewater infrastructure can affect in-stream function downstream of wastewater treatment plants. Point source discharges of treated wastewater effluent, leaky sanitary sewer pipes and combined sewer overflows can affect ER and dissolved oxygen concentrations as these labile components enter urban streams [5]. Studies of newly designed WWTPs suggest that receiving streams do not display dramatic oxygen depletion downstream as a result of high organic loads; rather, high nutrient concentrations act to increase GPP, ER and enhance nutrient uptake rates [138,139]. In a study of the effect of NO_3^- discharge from a WWTP, researchers in Greensboro, NC showed that while denitrification rates were much higher downstream of the WWTP compared to upstream, the efficiency of denitrification was decreased significantly: only 2.3% of the NO_3^- load was removed downstream of the WWTP compared to 46% upstream [140].

Hyporheic Zone. The impact of urbanization on the biogeochemical processes and ecological function of the hyporheic zone has been virtually unresearched in urban streams. The hyporheic zone is the subsurface area directly beneath and lateral to the wetted stream where surface water and groundwater mix. Researchers have demonstrated that streams with hyporheic zones have higher rates of nutrient retention [141,142] and metabolism [143,144] and that hyporheic zones modulate surface water temperatures [89,145]. Urbanization can have a diversity of impacts on the hyporheic zone and subsequently the surface stream. The input of fine sediments from initial urbanization and in-stream bank erosion can reduce surface water and groundwater exchange and potentially decrease the size and function of the hyporheic zone. While studies from the southern U.S. are few, other researchers have found that land use change results in decreased hyporheic diversity [146,147] and increased degree of subsurface anoxia [147] which can impact both organism health and biogeochemical cycles. Increased pollutant load and reduced water exchange associated with urban streams can reduce the capacity of the zone to retain and transform metals and other toxins. Loss of hyporheic zones in urban streams reduces the functions described above and reduces the overall health of the stream ecosystem and their ability to recover from disturbances. Many restoration plans in the southeast frequently have designed or unintentional impacts on the hyporheic zone [148-150] through activities such as channel modification and placement of in-stream structures. It is often assumed that restoration targeting the surface stream also benefits the hyporheic zone [151]; however, restoration intended to improve groundwater-surface water interactions must incorporate a watershed perspective to ensure that any improvements are not eliminated through other processes, such as the input of fine sediment from upstream sources [150].

4.4. Urbanization Effects on Biological Communities

Stream Communities. In the southern U.S., as in other regions of the country, urban streams are characterized as having degraded biological function showing decreased invertebrate diversity [101,152] decreased invertebrate richness [101,152,153], loss of intolerant taxa [101,152], decreased fish diversity [101], and decreased salamander diversity [154]. A comprehensive review of mussels in the southeastern U.S. describes their overall decline due to a variety of factors, including urbanization [155]. A range of variables and combinations of variables have been used to measure the effects of land use change on macroinvertebrate communities including, but not limited to, abundance, diversity, percent composition, percent (in)tolerant, and EPT (Ephemeroptera/Plecoptera/Trichoptera).

Researchers have recommended several variables as the most responsive for monitoring urban systems, for example (1) taxon richness, EPT richness, and the ICI in GA [156]; (2) Margelef's richness, predator richness, and omnivore richness in Texan streams [105]; and 3) the stream condition index (SCI) in FL, USA [157]. While urbanization is known to decrease the abundance and diversity of organisms living in streams, it is not clear which forcing factors, such as changes in discharge, sediment, and pollutant concentrations, are most important for causing decreased diversity [101]. The NAQWA Program has provided some interesting conclusions relating to the relationship between urbanization and biological diversity. The importance of documenting historic land use was highlighted since stream invertebrate taxa were not resistant to even low levels of urbanization and communities degraded rapidly, particularly if significant historical anthropogenic land use disturbances had occurred [152]. This pattern is further demonstrated in streams in Asheville, NC where there were no differences between fish and benthic macroinvertebrate communities in suburban and urban sites measured using taxa richness [158]. Past studies have suggested that watersheds with agricultural land-use prior to urbanization may have historically altered communities and these historic effects may confound the urbanization response [159].

Benthic macroinvertebrates are key members of the stream community that play a role in the processing of coarse and fine particulate organic matter [160]. Plecoptera (stoneflies) are known to be sensitive taxa often absent in urban streams. These organisms are key feeders on leaf material; however, the impact of their loss on leaf litter breakdown in urban streams is less understood. In 18 FL catchments, snails, dominated this key leaf shredding function, demonstrating the functional flexibility of the system. However, as snails are hard-bodied they will play a different functional role in the overall food web compared to the soft-bodied Plecoptera that may be preyed upon by fish. Thus, while loss of sensitive taxa has been documented across the southern U.S., fewer studies have related these losses to changes in stream ecosystem function. A common pattern seen in the effects of land use change on southeastern stream communities is a decreased diversity accompanied with a replacement by a more homogeneous biological community. This pattern was exemplified in a study of 36 streams in the southern Appalachians where unique endemic fish species were decreased in streams with lower canopy cover, a surrogate measure of disturbance by agriculture and urbanization [161]. In GA streams, benthic macroinvertebrate communities were more homogeneous and composed of tolerant taxa in urbanized catchments [156]. This simplification of the macroinvertebrate community may also impact higher trophic levels where the nutritional value of the altered community may not be equivalent to the pre-urbanization community. Elemental stoichiometry has been a useful tool for investigating food web interactions and nutrient recycling by organisms [162] and there is some evidence to suggest that an organism's nutrient:C ratio relates to their pollution tolerance [163]; however the influence in southern U.S. urban systems is not well understood. Finally, loss of key taxa from urbanization not only affects the aquatic food web but also impacts the adjacent riparian food webs. The cross-habitat flux of nutrients and carbon between the stream and riparian zone is well documented [164] and emerging benthic macroinvertebrates are food for spiders [165], birds, bats [166], and salamanders [167]. Similarly, terrestrial insects fall into the stream and become food for fish [168]. How urbanization potentially alters these reciprocal food webs and their resultant trophic cascades is an important area of future research.

As described earlier, wastewater treatment plants alter the downstream environment by discharging water higher in nutrients and dissolved organic carbon and increasing stream discharge immediately below the discharge point. Wastewater often has a different chemical signature and researchers have exploited this difference to measure the effects of the anthropogenic nutrients discharged into stream communities. In the Cape Fear Basin, anthropogenic nutrients were transferred up the food chain from emerging insects into the foraging bat population [169]. Interestingly, insect abundance and diversity was higher upstream of the wastewater plant than downstream and while bat foraging activity was equal at the two sites, bat community composition differed [169]. Movement of anthropogenic nutrients into the food web was also documented in headwater streams in the Piedmont of NC [170,171]. In VA, freshwater mussels were absent below a wastewater treatment plant, most likely due to sensitivity to the domestic effluent; however, snails and clams were more tolerant [172]. As wastewater treatment plants are often embedded in the urban landscape, it can be difficult to tease apart the most significant factor causing reduced ecosystem health. Further studies that link increased nutrient, carbon, and pollutant conditions to food web interactions, macroinvertebrate and fish growth and production are needed to determine any potential harm to these communities.

Hyporheic Organisms. Streams with hyporheic zones have higher invertebrate production [173] and diversity [174-176] relative to streams with sparse hyporheic zones. While a significant amount of hyporheic zone research has occurred in the western United States, streams in the southern U.S. show similar patterns. For example, streams in VA were shown to have higher invertebrate production and diversity when the hyporheic zone was included in the analysis [173,177]. In 14 unpolluted eastern sites that were surveyed, all newly discovered hyporheic copepod species were found in the unglaciated southeast [178], indicating both the potential capacity for streams in this region to support hyporheic communities and the unknown diversity that remains to be described.

As a group, hyporheic invertebrates possess diverse morphologies and life histories. Stygobionts, the permanent members of the hyporheic zone, are pigmentless, eyeless, have elongated bodies and are very similar to cave fauna. While the importance of the invertebrate community to ecosystem function has been debated [179-181], hyporheic invertebrates can significantly affect biogeochemical cycling rates. Nitrate uptake/regeneration, respiration rates, and particulate organic matter accumulation increased in microcosms with higher hyporheos biomass compared to microcosms with lower biomass [182]. These interactions in turn may affect the concentrations and form of dissolved ions and organic carbon/nitrogen returned to the surface stream. Other members of the hyporheos are considered “temporary” and are represented by organisms that spend some portion of their life cycle in the surface stream. For example, amphibitic Plecoptera (Stoneflies) in the Flathead River (MT) have been collected from hyporheic water at least 4.2 m below and 50 m laterally to the river [183]. These organisms return to the surface stream to emerge, mate, lay the eggs of the next generation [183,184], and are a source of food for fishes and riparian birds and bats. Hyporheic zones may also serve as refugia for benthic macroinvertebrates during high [185] and low [186] flow events and surface water—subsurface water connections are often associated with fish spawning locations [187,188]. Increased pollutant loads associated with urban streams can potentially harm the hyporheic microbial and invertebrate communities, this is an important focus area for future research.

5. Challenges and Opportunities for Future Work

Streams throughout the southern U.S. have experienced rapid urbanization over the last quarter century and changes in the water cycle have resulted in various alterations to watershed hydrology, water quality, and ecosystem processes in urban streams (Table 3).

Table 3. Common alterations to watershed hydrology and in-stream processes in response to urbanization in the southern U.S.

Watershed Hydrology

Precipitation and Evapotranspiration

- Increased summer rainfall due to urban heat island in large cities
- Evapotranspiration may decrease locally but more work is needed

Stream Hydrology and Peakflows

- Increased peak flows and discharge/stage variability

Baseflow and Groundwater Recharge

- Decreased baseflow as a % of total annual streamflow, more work is needed to understand inconsistent streamflow responses
 - Changes in groundwater recharge mechanisms and spatial distribution
-

In-stream Processes

Channel Geomorphology

- Increased channel dimensions and channel homogeneity
- Decreased headwater stream length/ drainage density
- Decreased extent of active floodplain and geomorphic complexity of the channel downstream of dams

Water Quality

- Increased stream water temperature and diurnal temperature variability
- Increased pollutant export from urban catchments
- Increased concentrations of fecal coliforms, hydrocarbons, and organic compounds in urban streams
- Increased oxygen demand, conductivity, total suspended sediments, and metals in urban streams

Ecosystem Processes

- Decreased in-stream nutrient retention and removal
- Decreased organic matter retention and processing
- Changes in balance between production and respiration greatly impacted by extent of riparian forest
- Altered food web interactions with resultant impacts on energy flow

Biological Communities

- Decreased abundance of intolerant macroinvertebrate taxa
 - Decreased diversity of invertebrates, fish, mollusks, and salamanders
 - Decreased size and function of hyporheic zones
-

Physically, streams have commonly responded by increasing their storm peakflows and flow variability. The inconsistent response of in-stream baseflow to urbanization reveals a need to better understand

and characterize the drivers affecting baseflow changes, particularly since low flows can exacerbate water quality impairments and negatively affect in-stream communities. In response to increased urban runoff, stream channels often enlarge their dimensions to accommodate the increased stormflow. Sediment transport from uplands to channels may decline in the urban environment and therefore bank erosion becomes a more important source of urban channel sediment. Inputs of fine sediments can reduce exchange between surface and groundwater and potentially decrease the size and function of the hyporheic zone which affects the viability of stream communities. These alterations can impact nutrient and organic matter processing by changing redox conditions which dictate which biogeochemical pathways dominate. Sediment accumulation can result in decreased oxygen penetration into sediment layers which results in reducing conditions that favor denitrification, sulfate reduction, methanogenesis and others. These shifts are an increasingly important area of research as the end products (e.g., N_2O , CH_4) are powerful greenhouse gases. In addition, stream channel burial in urban areas has altered the headwater stream network in a number of southern cities. These changes influence hydrologic response, water quality and channel form by delivering high volumes of water carrying a suite of pollutants directly to streams, even during relatively small rain events. Recent work has shown the importance of headwaters channels for nutrient uptake [119] and more work is needed to establish the links between alterations to headwater streams in urban areas and restoration of key ecosystem functions including nutrient processing and organic matter retention [84,130].

Our review of the literature on urbanization effects on watershed hydrology and in-stream processes suggests several areas where gaps in understanding should be addressed by future work. These opportunities include understanding the variability in responses within and between physiographic provinces in the southern US; developing comprehensive water budgets for urban watersheds; evaluating the combined impacts of land-use and climate change; understanding how pre-urbanization land-use history affects stream response; understanding and integrating measures of impervious surface connectivity into observational and predictive studies of urban watersheds; integrating hydrologic connectivity with biogeochemical cycling; developing predictive models and other tools to aid decision makers involved in restoration efforts; developing a clearer understanding of the complex interactions between catchment and in-stream processes in urban systems. Such studies would benefit from interdisciplinary approaches that involve hydrologists, soil scientists, geochemists, engineers, planners, ecologists, economists, social scientists, and others.

Although much work has been done on urban stream response in the Piedmont setting, less work has been done in the Coastal Plain setting, and very minimal work has been done on urban stream response in mountainous settings in the southern U.S. There are many opportunities for advancements in these understudied settings and comparisons of the differences in urban hydrological responses across physiographic regions. Similarly, within each physiographic province, certain urban areas are much better studied than others. Clearly, continued efforts need to focus on understanding how variations in soils, vegetation, styles and history of development, affect transferability of results from the well-studied cities to others in the region.

Various components of urban water budgets have been well-documented, but very minimal work has been done on developing detailed water budgets for urban watersheds [23]. These types of studies would provide great insight into the hydrological responses to urbanization, as suggested by Welty *et al.* [22]. For example, leakage from urban sewer, water, and stormwater networks is common

in urban areas [55], but there are few published studies documenting these contributions to groundwater recharge in urban areas of the southern U.S. [23]. Although urban induced rainfall has been documented in several southern U.S. cities, no studies have yet analyzed the effects of this altered rainfall regime on stream runoff generation. The urban induced rainfall effect is an important component to consider in studies that aim to evaluate the effects of global climate change on rainfall-runoff response or water budgets in large southern cities. The overall effects of urbanization on the water cycle need to be better constrained in order to evaluate the importance of hydrologic changes induced by land use change relative to those that may be linked to climate change. Urbanization impacts are occurring simultaneously with climate change, complicating potential analyses of historical hydrologic datasets to evaluate either phenomenon. More work is needed to understand how these forcing factors (land-use and climate change) are interacting and their combined influence on hydrology, geomorphology, biogeochemical cycling and ecology in the southern U.S. and throughout the globe.

Much of the recent land-use change in the southern U.S. is the result of a conversion from agricultural land-use to urban/suburban development. There is a lack of research documenting the effects of historical land-use change (*i.e.* forest to agricultural conversion) and its influence on the more recent urbanization response, but research suggests that past land-use has confounding effects on water quality and biotic communities [36,159]. The urbanization effects are superimposed on the response to prior land-use change. For example, O'Driscoll *et al.* [71] found that channelization to improve drainage for agriculture has altered the dimensions of numerous channels in agricultural watersheds of eastern NC. Urbanization within these formerly agricultural watersheds results in further enlargement of channels in response to increased stormwater runoff. The high density of dams (many constructed in the 1950s and 1960s) and water control structures in the region may also play an important role on stream channel geomorphology and the diversity of riparian habitats in many urban settings throughout the southern U.S. In addition to the effects of previous land-use, there are also questions remaining as to the duration of a watershed's response to urbanization. As pointed out by Leopold *et al.* [79] and in a global review by Chin [7], the duration of a stream channel's response to urbanization generally takes more than two decades and may vary across watersheds. More studies are needed to document the duration and stages of the urban response and to better isolate the responses to land-use changes that occurred prior to urbanization, because that history will influence the timing and duration of the urbanization response. Simon's channel evolution model [189] provides a general framework for a stream reach's geomorphological response to disturbance, particularly for stream channelization. There is a growing need for channel evolution models that are specifically designed to capture the complexities of urban watersheds and account for the differences in physiographic setting [9].

Watershed TIA has been shown to serve as a reliable indicator of the degree of urban stream response, however more work is needed to evaluate the effects of the connectivity of impervious area to stream channels [52]; only a few studies in the southern U.S. have evaluated the effective or directly connected impervious area [38,45]. Improved understanding of how stormwater management can reestablish natural flowpaths thereby enhancing infiltration, evapotranspiration and pollutant removal will contribute to improvements in urban stream physical and ecological functioning. Rapid delivery of surface runoff and associated pollutants to streams from highly connected stormwater drainage

systems during rain events often results in large loads transported downstream as a result of flashy urban hydrology. In a recent review, Schueler *et al.* [103] developed a broad impervious cover model that predicts stream impairments across a range of imperviousness. Future work could also adapt this type of model to specific physiographic settings or ecoregions common in the southern U.S. More work is needed to address the thresholds of watershed impervious area or urbanization that are exceeded prior to impairment and to understand the cumulative effects of multiple stormwater management practices on groundwater recharge, stream water quality, and ecosystem functioning [5].

Nutrient cycling is increasingly being studied in urban systems with results pointing to higher rates of microbial assimilation and transformation as a result of higher in-stream concentrations [121,122]. However, saturation of these processes can lead to decreased efficiency [125] and overall net increases in loads transported downstream. Urban streams also generally exhibit higher concentrations of other pollutants, such as metals, hydrocarbons and organics which can adversely affect the health of aquatic communities during baseflow conditions. Better understanding of the effects of individual and combinations of toxic pollutants on community structure and function is needed as well as their effect on biogeochemical cycling and organic matter processing. It is not surprising, given the diversity of stream channel responses to urbanization with respect to channel geomorphology, sediment, streamflow, and groundwater recharge, that there is a weak understanding of the causal relationship between the effects of urbanization on stream health. Most likely there is no one variable that is most accountable for determining ecological effects; however, the apparent disconnect between cause and effect reflects the difficulty in restoring the biology of urban streams along with their physical structure.

Streams in urbanizing landscapes provide social and economic benefits to communities through enhancement of aesthetic and recreational opportunities. Restoration and conservation practices that integrate neighborhoods with stream corridors offer city dwellers a glimpse of nature and a way for their residents to explore the natural world [190]. However, conflicting interests and views regarding urban streams and their watersheds inherently exist and can greatly impact human behaviors concerning stormwater drainage which in turn directly contribute to pollutant loading and the ecological health of urban streams [5]. Recently, Wenger *et al.* (2009) provided an excellent review of urban stream ecology and a thorough assessment of key research questions for the future [159]. The emerging discipline of urban stream ecology offers opportunities for integrating social and economic values with physical, chemical and ecological functions. Development of tools and spatial models that simplify the complex interactions among the many variables influencing urban stream hydrology and ecology are greatly needed to assist resource managers with the challenging decisions of where to focus conservation and restoration activities.

Population and urban areas in the southern U.S. are projected to continue to grow rapidly over the next half century. Increased water demands, stormwater runoff, wastewater generation, and the impacts of aging urban infrastructure will require advances in watershed management to allow for sustainable development of the region. Great challenges remain in mitigating and reversing the damages that have been done to stream ecosystems as a result of urbanization throughout the southern U.S. and the globe. Storm water runoff as a result of increased impervious cover, disconnection of streams from their floodplains, and removal of riparian vegetation have interacted to produce urban streams that function more like pipes than dynamic ecosystems. Restoration projects are being constructed while ongoing research seeks to better understand complex interactions between catchment

hydrology and ecosystem response. An adaptive strategy should be employed that allows for changes in approach and implementation to occur as knowledge develops. Numerous opportunities exist to develop meaningful and measurable indicators of restoration success, develop tools for placement of projects within communities, and assist resource managers with strategies for implementing cost effective projects.

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