Chapter 13. Water and Forests

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Key Findings

- Forest conversion to agriculture or urban use consistently causes increased discharge, peak flow, and velocity of streams. Subregional differences in hydrologic responses to urbanization are substantial.
- Sediment, water chemistry indices, pathogens, and other substances often become more concentrated after forest conversion. If the conversion is to an urban use, the resulting additional increases in discharge and concentrations will produce even higher loads.
- Although physiographic characteristics such as slope and soil texture play key roles in hydrologic and sediment responses to land use conversion, land use (rather than physiography) is the primary driver of water chemistry responses.
- Conversion of forest land to urban uses may decrease the supply of water available for human consumption and increase potential threats to human health.

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- Increases in urbanization by 2060 in the Appalachians, Piedmont, and Coastal Plain will increase imperviousness and further reduce hydrologic stability and water quality indices in the headwaters of several major river basins and in small watersheds along the Atlantic Ocean and Gulf of Mexico.
- On average, water supply model projections indicate that water stress due to the combined effects of population and land use change will increase in the South by 10 percent by 2050.
- Water stress will likely increase significantly by 2050 under all four climate change scenarios, largely because higher temperatures will result in more water loss by evapotranspiration and because of decreased precipitation in some areas.
- Approximately 5000 miles of southern coastline are highly vulnerable to sea level rise.

Introduction

Compared to all other land uses, southern forests provide the cleanest and most stable water supplies for drinking water, recreation, power generation, aquatic habitat, and groundwater recharge (Sun and others 2004, Jackson and others 2004, Brown and others 2008). Forests are unique among land covers because they are long-lived and relatively stable. However, they are subject to substantial structural and functional alterations by management practices and/or natural disturbances, the intensity of which determines whether alterations are short or long-term. Water resources in the South are at risk of degradation from a growing population, continued conversion of forests to other land uses, and climate change. Urban and agricultural lands can impair water resources by introducing nutrients, sediment, bacteria, and other pollutants to streams. Additionally, altered hydrology—including higher peak flows, and lower baseflows and hydroperiods (Amatya and others 2006)—is common with forest conversion to other land uses. Together, these changes can modify the habitat and consequently the composition of aquatic (and riparian) communities.

Historical land use practices have dramatically changed the landscape of the South. Soil erosion and sedimentation were prevalent throughout the region during the period of agricultural expansion in the 18th and 19th centuries. Evidence can still be seen today in the sediment deposits of floodplains. This massive topsoil erosion and depleted soil productivity was followed by a period of agricultural abandonment and reforestation throughout much of the South. Today major land uses include forestry, agriculture, and increasing urbanization, each of which has its own signature on water resources. Table 13-1 shows the total impervious area as a percent of the total land area for eight southern States—all indications are that the amount of impervious area is likely to increase significantly in the coming decades. Much of the increase in impervious surfaces will be derived from losses of forest land through conversion to urban land uses (ch. 4). One implication for these losses is likely to be an intensification of management activities on remaining forest lands to provide sufficient wood products from a shrinking land base (ch. 9). In addition, expanding wood-based biofuels markets may also increase management intensity and may shorten rotations, increase the use of irrigation and fertilization, and change species composition to favor fast growing species (ch. 10). This intensification and alteration of management practices could have important implications for water resources.

Although population growth, land use change, and intensification and expansion of managed forests are the most obvious sources of impacts on southern water resources, other factors, including climate change, increasing climatic variability, and climate change induced sea level rise could have large impacts as well. For example, some climate models project more frequent El Niño-like conditions (Thompson and others 2003) resulting in more extreme rainfall events. Climate change and increased climate variability will both directly and indirectly affect water resources. Higher temperatures could decrease streamflow by increasing evapotranspiration, although this outcome may be buffered by increased annual precipitation (Sun and others 2005, Oki and Kanae 2006). Subsequently, lower streamflow decreases water supply, degrades aquatic communities, and diminishes water quality. Extreme rainfall events increase flood severities and frequencies that negatively impact human safety and welfare and the functioning of aquatic communities. Changes in hydroperiod (Ernst and Brooks 2003, Ford and Brooks 2002) that result from disturbance- and climate-induced rises in sea level (Ross and others 1994) will have significant direct effects on ecosystem processes in forested wetlands (Amatya and others 2006) and potentially devastating impacts on human welfare in urban and rural areas. For example, the Intergovernmental Panel on Climate Change fourth assessment report (AR4) estimated global mean sea-level rise between 0.28 m and 0.43 m by end of the 21st century (Parry and others 2007). Those estimates excluded dynamic ice changes such as massive movement in the Greenland ice cap. Before 1990, thermal expansion was the largest contributor to sea-level rise but since then, its importance has been eclipsed by a combination of melting glaciers, ice caps, and ice sheets (McCullen and Jabbour 2009). Since the AR4 report there have been other estimates of sea-level rise using non-dynamic modeling techniques and models that include dynamic ice changes. These models predict that sea-level may rise from 0.4 to 2.0 m by the end of the 21st century (Rahmsorf 2007, McMullen and Jabbour 2009, Solomon and others 2009).

Climate change and variability and sea level rise do not act alone to affect water resources. They interact with land use change and exacerbate the impacts on water quality and quantity. An understanding of the relationships between forest cover and water resources, and how these relationships interact with climate change and growing water demand, is critical to crafting actions that minimize the detrimental effects of land conversion now and in the future.

The goals of this chapter are to (1) outline the surface-water consequences of forest conversion to urban and agricultural land uses and highlight differences among physiographic subregions, (2) evaluate and discuss the water-resource implications of intensifying and altering forest management practices, (3) discuss the implications of climate change, growing demand for water, and land use change on water resources, and (4) to discuss the potential impact of sea-level rise on the southern US.

Methods

Literature review and syntheses

An extensive literature synthesis was conducted to address our research questions. We began by evaluating general relationships between forest cover and water resource, with a primary focus on water quality and quantity. Then we explored specific practices in greater detail including forest harvesting, intensification of forest management, agriculture, and urban land use. Finally, we examined studies of particular physiographic subregions to isolate and compare the geographic nature of responses to these land uses and management activities.

Regional Modeling

Water resources are influenced by many complex factors such as climate variability, land use/land cover change, groundwater availability, surface water storage, population growth, and economics (fig. 1). To account for these factors, we applied a water accounting model, WaSSI or Water Supply Stress Index (Sun and others 2008), to examine future changes in water stress induced by humans, biological factors, and climate.

The scale of the WaSSI model can encompass and entire system from watershed to basin or any portion thereof, depending on the research question and availability of data to examine human water use and demand. The model simulates full monthly water balance, including evapotranspiration, soil moisture content, and water yield. Within each basin, spatially explicit land cover and soil data are used to account for evapotranspiration, infiltration, soil storage, snow accumulation and melt, surface runoff, and baseflow processes; and discharge is routed through the stream network from upstream to downstream watersheds. Evapotranspiration is estimated by applying an empirical equation to multi-site eddy covariance-based evapotranspiration measurements, using MODIS derived Leaf Area Index (a measure of the amount of leaf cover), potential evapotranspiration, and precipitation as independent variables (Sun and others 2010). Estimations of infiltration, soil storage, and runoff processes are derived by integrating algorithms from the Sacramento Soil Moisture Accounting Model and STATSGO-based soil parameters (Koren and others 2003).

Water Supply Stress Index is defined as water demand divided by water supply (WaSSI=Water Demand/Water Supply). Monthly water supply was defined as the total potential water available for withdrawal from a basin including total surface water supply, groundwater supply (Kenny and others 2009), and return flow from each of seven water sector users including commercial, domestic, industrial, irrigation, livestock, mining, and thermoelectric (Solley and others 1998). Return flow rates vary among watersheds and water use sectors. For example, in the South return flow rate for the domestic use sector averages 68 percent but ranges from 1.6 to 90 percent across HUCs. Similarly, the thermoelectric sector averages 76 percent but ranges from 0.1 to 100 percent. Water demand represents the sum of all water use by each of the seven sectors and public supply or water withdrawal by public and private suppliers for use by domestic and Industrial sectors. Historic annual water use values reported by USGS (Kenny and others 2009) were redistributed to each month across the South for the irrigation and domestic sectors by applying a series of monthly water use functions.

WaSSI uses the Natural Resources Conservation Watershed Boundary Dataset 8-digit Hydrologic Unit Code (HUC) watershed as the working scale (NRCS 2009) There are approximately 2100 8-digit HUC watersheds in the lower 48 States, 674 of which are in the South.

The databases required for WaSSI include historic water use and return flow rates by water use sectors, groundwater withdrawal, historic and projected climate, population, and land use and other remote sensing products such as leaf area index. Because these databases had different temporal and spatial scales, conversion to the 8-digit HUC watershed level was necessary before scenarios could be developed to individually and collectively quantify the impacts of climate, land use, and population changes on water supply and demand (table 13-2).

Data Sources

Regional Modeling Databases

Historic water withdrawals and use—The 2005 National Anthropogenic Water Use Survey datasets published by the U.S. Geologic Survey were used to determine historic water demand (Kenny and others 2009). Return flows from each water use sector came from the 1995 U.S. Geologic Survey water use survey dataset (Solley and others 1998), the most recent water use dataset to include return flow information. The U.S. Geologic Survey grouped national water users into one of seven categories: domestic (1.1 percent), industrial (5 percent), irrigation (37 percent), livestock (0.6 percent), mining (41 percent), thermoelectric (0.7 percent), and aquaculture (3 percent). For the purposes of this analysis of southern water use, an eighth category, public supply, was added to represent the water withdrawal by public and private utilities for general distribution to domestic and industrial sectors (12 percent of total freshwater). In contrast to national usage, thermoelectric water withdrawal dominates (54 percent) in the South, followed by irrigation centered in the Mississippi valley and western Texas. However, return flow rates from power plants are typically high (> 90 percent), because water is returned to the ecosystem shortly after being withdrawn, thus making irrigation the largest consumptive user (47 percent) followed by public supply (34 percent) and thermoelectric (10 percent). Over half of the water withdrawn is from groundwater in the Mississippi valley, western Texas, and the coastal plain.

Climate data— The full climate data cover the 48 conterminous states at a county scale and range from 1950 to 2099. For this analysis, hydrologic simulations were conducted through 2050 for the southern states defined by the Futures Project ecoregions. Data included monthly air temperature and precipitation as predicted by three General Circulation Models (Hadley Centre for Climate Prediction and Research UKMO-HadCM3, Australian Commonwealth Scientific and Research Organization Atmospheric Research CSIRO-Mk2.0 and CSIRO-Mk3.5, and the Center for Climate System Research (The University of Tokyo) National Institute for Environmental Studies and Frontier Research Center for Global Change MIROC3.2) and under two Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios (SRES A1B and B2). The county climate data were scaled to the 8-digit HUC watersheds, and the WASSI modeling effort used the following climate change and emissions scenarios: CSIROMK35A1B, MIROC32A1B, CSIROMK2B2, and HadCM3B2. It should be noted that climate models are often calibrated to best address climate within a specific geographic region (e.g. MIROC32A1B was specifically developed for Japan). The application of any general circulation model (GCM) will not be universally reliable (i.e. accurately able to predict future climate). In this study the MIROC32A1B predicts climatic conditions for the southern US that are extreme for most parts of the globe. Other models could be considered more moderate in their predictions of future climate. The results of all for model predictions are presented in this chapter but further discussion of the GCM can be found in Chapter 3.

Population and land use change data—The U.S. Census Bureau records indicate that population increased about 30 percent from 1980 to 2000. Population projections at the census block level were aggregated to the 8-digit HUC watershed scale for each year from 1967 to 2050 (NPA Data Services Inc. 1999). For the WaSSI model simulation, we selected the land use data (ch. 4) belonging to Cornerstone A (Intergovernmental Panel on Climate Change storyline A1B, level crop prices, high timber prices). For model simulations, the land use classes described in chapter 4 were grouped into eight categories as: crop, deciduous forest, evergreen forest, mixed forest, grassland, shrub land, savanna, and water/urban/barren. In addition, the land use data were rescaled from the county to the 8-digit HUC watershed for model input. The representative year was 2000 for historic baseline model simulations, and 2050 was selected for future model simulations.

Sea Level Rise

We used the analysis of Titus and Richman (2001) to identify land area 1.5 m, 1.5 to 3.5 m, and >3.5 m above sea level and generated 66 maps (based on 1-degree digital elevation models) to outline coastal areas on the Gulf of Mexico and the southern Atlantic States from Virginia to Florida. The vulnerability of coastal regions (coastal vulnerability index, or CVI) was calculated using the analyses of Hammar-Klose and Thieler (2001), who incorporated geomorphology, coastal slope, rate of relative sea-level rise (mm yr^{-1}), shoreline erosion and accretion rates (m yr^{-1}), mean tidal range (m), and mean wave height (m) into calculations of CVI for the Atlantic, Pacific, and Gulf of Mexico coasts. A ranking was applied to each variable for ~5 km segments (or 3 minutes) of coastline and then combined to form an index of risk using the following equation:

$$\text{CVI} = \sqrt{\left(\frac{a*b*c*d*e*f}{6}\right)}$$

Where a is geomorphology, b is coastal slope, c is relative sea-level rise, d is shoreline erosion/accretion rate, e is mean tide range, and f is mean wave height, and the final CVI was broken into to four risk categories: low (<8.7), moderate (8.7 to 15.6), high (15.6 to 20.0), and very high (>20.0).

Data sources used to calculate CVI included state geologic maps and 1:250,000-scale topographic maps to determine geomorphology; a combination of U.S. Navy ETOPO5 and National Geophysical Data Center digital topographic and bathymetric elevation databases to determine coastal slope; National Ocean Service data to determine relative sea-level rise and mean tidal range; a combination of the May and others (1982) Coastal Erosion Information System dataset and more recent state and local regional studies to estimate shoreline erosion and accretion rates; and the U.S. Army Corps of Engineers Wave Information Study to estimate mean wave height (Hammar-Klose and Thieler 2001). After all data were compiled and rescaled to a 5 km grid, each variable was ranked from 1 (very low vulnerability to sea level rise) to 5 (very high vulnerability to sea level rise).

Results

Physical environment of the southern region

The southern climate is predominantly humid subtropical; however, the western most areas, such as Texas and Oklahoma, are semi-arid. The average annual temperature range is 15 to 21 °C and the precipitation range is 1010 to 1520 mm yr^{-1} (Bailey 1980). Ultisols, the predominant soil order of the South, are strongly leached and nutrient poor with a subsurface accumulation of clay (Bailey 1980, USDA Natural Resource Conservation Service 2009). The relief is mostly level along much of the Atlantic and Gulf of Mexico, however, the upper Coastal Plain of Alabama and Mississippi is moderately to gently rolling (Martin and Boyce 1993). Coastal Plain soils are sandy. The Piedmont has gently rolling to steep terrain with clayey surface and subsurface soils. Consequently, the potential for erosion is high throughout the Piedmont and even higher toward the Blue Ridge subregion of the Southern Appalachians (Trimble 2008). The Southern Appalachians have steep topography and elevation ranges from 225 to 900 m. The three major river basins of the South are the Mobile, Tennessee, and Cumberland (World Wildlife Fund 2010). Southern streams support a diversity of freshwater species and are thus are a high conservation priority (World Wildlife Fund 2010). Physiographic subregions, and the landscape components of watersheds within them, are connected through the flow of energy and materials, the movement of species, and the movement of insects, disease, and other disturbance agents. Unlike many of the exchanges, the movement of water across most of the South is fairly predictable because water follows hydrologic flowpaths that are primarily driven by elevation gradients.

Exceptions occur in the lower Coastal Plain and other systems where hydrology is dominated by groundwater hydrology. Understanding how changing landscapes will alter the quantity, quality, and value of surface water and groundwater requires analyses at expanding spatial scales to examine how rapid urbanization affects forest practices such as cutting, road building, and drainage.

Functions of forested wetlands and riparian forests

Forested wetlands can be described by hydrogeomorphic considerations such as landscape position, water source, and hydrodynamics are dominant process regulators (Ainslie 2002). The three most common classes of southern forested wetlands are riverine, depressional, and flat with mineral or organic soil (table 13-3). In general, forests and hydrological cycles are connected through the processes of evapotranspiration (Amatya and others 2008). Hydrological functions of southern forested wetlands may include flood mitigation or short-term surface water storage; and to a lesser extent than forested wetlands in other regions of the United States, they abate storms and recharge groundwater (Walbridge 1993, National Research Council 1995). Biogeochemical functions of wetlands, including cycling of elements and retention and removal of dissolved substances, serve to improve surface, subsurface, and ground water quality (National Research Council 1995, Blevins 2004). Regardless of type, all forested wetlands contribute to food web maintenance by providing habitat for plants and animals (Walbridge 1993, Faulkner 2004). Some forested wetlands may provide unique functions based on their distinctive characteristics and structure. For example, Carolina Bays may contain rare and endangered plants and also provide desirable breeding sites and habitat for birds and wildlife (Ainslie 2002). And mineral soil flats can have very high herbaceous species richness in part because of their unique fire regime (Ainslie 2002).

Riparian forests also provide hydrological, biogeochemical, and habitat functions. Many studies have shown that riparian forests help to stabilize stream banks and trap pollutants such as sediment, nutrients, bacteria, fertilizers, and pesticides from runoff (USDA National Agroforestry Center 2008, Anderson and Masters 1992, Klapproth and Johnson 2000, Binkley and Brown 1993, Vellidis 1999, de la Crétaz and Barten 2007, Naiman and others 2005). In particular, Naiman and others 2005 found that "The hydraulic connectivity of riparian zones with streams and uplands, coupled with enhanced internal biogeochemical processing and plant uptake, make riparian zones effective buffers against high levels of dissolved nutrients from uplands and streams, while geomorphology and plant structure make them effective at trapping sediments." However, an intact riparian corridor does not ensure stream protection as this relationship is dependent on other factors including residence time of pollutants in the buffer, depth and variation of water table, upland land use practices, climate, and watershed characteristics such as topography, hydrology, soils, and vegetation (Groffman and others 2003, Tomer and others 2005, Walsh and others 2005, de la Crétaz & Barten 2007).

Habitat functions provided by riparian forests include lower water temperatures for aquatic animals due to shading from trees, along with shelter for birds and wildlife (Anderson and Masters 1992, Binkley and Brown 1993, Vellidis 1999, Naiman and others 2005). Riparia are sources of large woody debris, which creates habitat heterogeneity, acts as a substrate for colonization, and provides nutrients to the aquatic (and riparian) community (Naiman and others 2005). Inputs of organic matter from riparian forests supply an allochthonous energy source to stream ecosystems, there by linking the riparian and aquatic foodwebs. Additionally, if Best Management Practices are implemented appropriately, riparian forests can also provide wood products, pasture for livestock, and recreational opportunities (Anderson and Masters 1992).

Hydrologic effects of forest conversion to other land uses

Harvesting forests reduces evapotranspiration and infiltration; creating impervious surfaces increases overland flow (Paul and Meyer 2001). Similarly, forest conversion to agricultural land may compact soils, reduce evapotranspiration and infiltration, and increase overland flow. Regardless of post-harvesting use, characteristic changes in hydrology following forest removal include greater streamflow and peak flows (Hibbert 1967, Bosch and Hewlett 1982, McMahon and others 2003, Schoonover and others 2006, Crim 2007, de la Crétaz and Barten 2007). Representative hydrographs for typical forested, agricultural, and urban watersheds are shown in figures 13-2 to 13-4. A study of the 13 southern States showed that streamflow increased by 69 to 210 mm yr^{-1} following forest harvesting (Grace 2005). Stednick (1996) found that a 20 percent change in forest cover produces a quantifiable change in water yield in the Appalachians, but that the threshold is about 25 percent higher for the Piedmont and Coastal Plain. Since the 1970s in Houston, impervious surfaces were responsible for 32 percent of the 159 percent increase in peak flows, and 77 percent of the 146 percent increase in annual runoff (Olivera and Defee 2007). Similarly since the 1960s in the White Rock Creek watershed of northeastern Texas, peak flows increased by 20 to 118 percent with varying precipitation intensities in response to dramatic increases in impervious cover (Vicars-Groening and Williams 2007). Stream hydrographs of urban watersheds reflect a flashy hydrology with greater pulses and faster attainment of peak flows during storm events (Beighley and others 2003, Calhoun and others 2003, Schoonover and others 2006, Crim 2007, Boggs and Sun 2011). However, as arid regions naturally have flashy hydrology due to inherent precipitation regimes, urban effects may be obscured on the hydrographs of streams in those parts of the South (Grimm and others 2004). Flow-duration curves also depict changes in hydrology by displaying the percentage of time stream flow equals or exceeds a particular value. The pre-urbanization flow duration curve exhibits more gradual variations while the post-urbanization curve is much steeper (fig. 13-5).

In urban and agricultural watersheds, decreased infiltration produces less groundwater recharge, possibly reducing baseflows (Rose and Peters 2001, Wang and others 2001, Calhoun and others 2003). As an example, in tributaries of the upper Chattahoochee River, Calhoun and others (2003) estimated that every 1 percent increase in impervious surface reduces baseflow by 2 percent. However, this is not always the case as illustrated by the lack of baseflow response to increases in impervious surface in the Florida Panhandle (Nagy, R.C.; Lockaby, B.G.; Kalin, L.; and Anderson, C. Manuscript in preparation. Urbanization of a coastal region and the effects on water resources. Authors can be reached at Brown University, Department of Ecology and Evolutionary Biology, 80 Waterman St., Providence, RI 02912; rachel_nagy@brown.edu.); and by higher median baseflow in pastoral watersheds compared to forested watersheds in the Georgia Piedmont (Schoonover and others 2006). The less responsive baseflow in the Coastal Plain may be explained by differences between the extent of surface water and baseflow recharge zones in very flat terrains where baseflow zones may extend beyond surface catchment boundaries. In the case of the pastoral vs. forested watersheds in the Piedmont, apparently reduced ET and adequate surface infiltration rates in the pastures accounted for the higher baseflows there.

Although historically the South has not experienced a great deficiency of water supply compared to other regions in the United States, with continued forest loss and expanding urbanization, water supply may become a more pressing issue in this region. When modeling land use effects only, Sun and others (2008) predicted reduced water deficits due to increased water yield following conversion of forest to urban land uses. However, they found that water resources would likely be under greater pressure in the future when the effects of climate change and population growth are also taken into account . Additionally, it should be noted that despite increased streamflow following forest removal, available surface water might decline due to unstable flow regimes. After increasing withdrawal rates from Alabama streams near Birmingham, surface water available for human use ranged from about 20 to 45 percent of discharge for urban watersheds and 20 to 60 percent for forested watersheds; and at higher withdrawal rates water availability was significantly higher in forested than urban watersheds (L. Kalin, unpublished data; Auburn University, School of Forestry and Wildlife Sciences, 602 Duncan Dr., Auburn, Alabama, 36849; kalinla@auburn.edu. Therefore, although total water yield is often reduced in forested watersheds compared to urban watersheds, forested watersheds may have a greater percentage of water available for use, suggesting that increasing urbanization contributes to greater stress. Lastly, degradation of water quality from point- and non-point source pollution can also reduce the amount of available water (Sun and others 2008).

Effects of Forest Conversion on Sediment

Forests stabilize soils (Jackson and others 2004); therefore soil is more readily eroded following removal of vegetation, and is transported as sediment into floodplains and other areas of lower topography (Jackson and others 2005, Trimble 2008) and/or directly into stream channels. The effects of historical agricultural use, in particular row-crop agriculture, on soil erosion and subsequent sediment deposition throughout the South were profound (Jackson, C.R. and others 2005, Trimble 2008, Casarim 2009). For example, in the Georgia Piedmont, sediment deposition from historical agriculture was as much as 1.6 meters in the Murder Creek floodplain (Jackson, C.R. and others 2005) and averaged 1.8 meters in Bonham Creek and Sally Branch watersheds (Casarim 2009). It can be difficult to differentiate sediment contributions from current land use versus historical agricultural use within a watershed because the legacy effects of historical land use can be observed decades later. Jackson, C.R. and others (2005) estimated that 25 to 30 percent of the sediment load of Murder Creek consisted of re-suspended legacy sediment. This means that land use conversions in the Piedmont have the potential to re-suspend legacy sediment that accumulated in the stream beds decades ago in addition to generating sediment export from terrestrial sources.

The combined effects of altered hydrology, removal of vegetation, and an increase in impervious surface often cause urban watersheds to exhibit stream sediment concentrations much higher than forested watersheds (Lenat and Crawford 1994, Paul and Meyer 2001, Schoonover and others 2005, Clinton and Vose 2006). In the Southern Appalachians, total suspended solids concentrations were 4 to 5 times greater in an urban compared to a reference stream (Clinton and Vose 2006). In the Georgia Piedmont, total dissolved solids concentrations were twice as high in urban streams compared to forested streams (Schoonover and others 2005, Crim 2007), but total suspended solids did not increase significantly under urban cover. In the Coastal Plain, Wahl and others (1997) found a twofold increase in total suspended solids in urban compared to forested streams (up to 200 mg/l during stormflow in the urban stream). Erosion associated with urban land uses can be particularly high at construction sites and areas of new development (Paul and Meyer 2001, Novotny 2003).For example 100- to 10,000-fold increases over nondeveloped conditions were reported by Paul and Meyer (2001), especially in areas with greater topographic variation such as the Southern Appalachians.

Effects of Forest Conversion on Water Chemistry

Undisturbed forested watersheds are generally associated with low stream-water concentrations of most ions. Since most forests are deficient in one or more elements, forested systems are generally effective in retaining inputs of nutrients. Consequently, net export of macronutrients, or nutrients required in large quantities such as N, P, and K, from undisturbed forested catchments is often negative, indicating an accretion of forest biomass (Swank and Douglass 1977, Likens and Bormann 1995). As an example, the 3600 ha Table Rock Reservoir watershed in the Southern Appalachians of South Carolina has been highly restricted in terms of any human activity since 1930, and water quality there remains unchanged since that time (Okun 1992).

Increased nutrient concentrations and loads have been observed in urban and agricultural streams compared to forested streams. Excess nutrients may arise from fertilizers, wastewater effluent, and industrial waste in urban areas; and from animal waste and fertilizers in agricultural areas. Concentrations of NO_3^-, Cl^-, Na^+, K^+, Ca^{2+}, and Mg^{2+} were all higher in an urban stream than a reference stream in the Southern Appalachians during baseflow and stormflow both (Clinton and Vose 2006). Nitrates had the most pronounced increase with mean concentrations of about 0.01 mg/L in the reference stream and 0.7 mg/L in the urban stream (Clinton and Vose 2006). However, ammonium concentrations were higher in the forested stream during stormflow. Bolstad and Swank (1997), working in the same physiographic subregion, found no increases in nitrate, ammonium, or phosphate concentrations during baseflow as urbanization indices increased. However, during stormflow, slight increases were noted for nitrate (from 0.05 to 0.07 mg/L with a significant regression relationship) and ammonium.

In the Georgia Piedmont, Schoonover and Lockaby (2006) found results similar to those of Clinton and Vose (2006) for dissolved organic carbon, NO_3^-, Cl^-, and K^+, as well as SO_4^{2-} when comparing streams with <5 percent and >24 percent impervious surface. The concentrations in streams of watersheds with >24 percent impervious surface were generally two-to-four times higher than those of less developed catchments. For instance, median baseflow nitrate concentrations were 0.61 mg/L for streams with <5 percent impervious surface and 1.64 mg/L for streams with >24 percent impervious surface; stormwater concentrations were 0.36 and 1.93 mg/L respectively for the same comparison (Schoonover and Lockaby 2006). Although ammonium concentrations have been reported to be higher in forested than in urban streams (Tufford and others 2003, Clinton and Vose 2006), two Piedmont studies produced different results (Schoonover and Lockaby 2006, Crim 2007). Similarly, dissolved organic carbon concentrations are often higher in forested watersheds than in urban streams (Wahl and others 1997) although there are exceptions (Schoonover and Lockaby 2006).

Unlike most studies of land-use/land-cover impacts on water quality, which have substituted space for time, Weston and others (2009) evaluated water quality changes within the Altamaha River Basin of the Georgia Piedmont for more than 30 years. The increases in population during that period exceeded 100 percent in some of the basin's watersheds. During that period, agricultural land use declined as populations rose, producing decreases in stream ammonium and organic carbon concentrations but increases in total nitrogen and nitrogen oxide concentrations and loads. Phosphorus concentrations did not increase with urbanization, which the authors suggest may reflect the elimination of phosphates in detergents after 1972. They also suggest that for the Piedmont, elevated total nitrogen and nitrogen oxides may serve as water quality signatures of urbanization and elevated ammonium, and that organic carbon may be associated with agriculture. Ammonium and organic carbon are also often linked with forest cover but the effects of changes in forest cover were beyond the scope of this chapter.

There are fewer studies of land-use/land-cover associated with the Coastal Plain than with the Piedmont. Wahl and others (1997) compared water quality within two coastal watersheds: one with increasing urbanization (18 percent impervious surface) and the other with predominately forest cover (no impervious surface). Nitrate was consistently higher in the urban stream: 130 ug/L in winter (90 ug/L in summer) versus 42 ug/L in winter (29 ug/L in summer). Ammonium was higher in the forested stream regardless of season (159 ug/L versus 70 ug/L) as were dissolved organic carbon concentrations (27 ug/L versus 13 ug/L). In a study within the Florida Panhandle (Nagy, R.C.; Lockaby, B.G.; Kalin, L.; and Anderson, C. Manuscript in preparation. Urbanization of a coastal region and the effects on water resources. Authors can be reached at Brown University, Department of Ecology and Evolutionary

Biology, 80 Waterman St., Providence, RI 02912; rachel_nagy@brown.edu), median concentrations of nitrate, ammonium, calcium, potassium, and sulfate were higher in watersheds with more urbanization (impervious surface up to 16 percent) than in their forested counterparts. However, nitrate concentrations in the Coastal Plain were well below those generally observed in some studies of urban watersheds of the Piedmont (0.35 mg/L versus 1.78 mg/L). Median concentrations of total phosphorus were high and increased from 0.31 mg/L in watersheds with <5 percent impervious surface to 0.43 mg/L in watersheds with >10 percent impervious surface. Similarly, total suspended solids increased from 1.50 to 2.40 mg/L for impervious-surface levels above 10 percent. In contrast, dissolved organic carbon declined from 36 mg/L in watersheds with low impervious surface to 30 mg/L in watersheds with >10 percent impervious surface. Tufford and others (2003) found that total phosphorus was significantly higher in urban streams than forested streams in the Coastal Plain (concentrations of roughly 0.06 mg/L versus 0.03 mg/L).

In general, increases in stream concentrations of several elements within urbanized watersheds are very common although the magnitude of increase and sometimes the particular ions involved vary considerably within and among physiographic subregions. While nitrate and potassium ions commonly increase (probably due to their mobility in water), responses of total potassium or phosphate are much more variable and may not occur at all. Responses of the other major elements fall in between those of nitrate and phosphate. Since discharge usually increases with urbanization, loads increase as well regardless of physiographic subregion. Consequently, there do not seem to be clear distinctions among subregions in terms of stream chemistry responses, which may indicate that the influence of land use overrides that of physiography in the South.

Higher loads of base cations (K^+, Ca^{2+}, and Mg^{2+}), Cl^-, and total nitrogen were found in agricultural streams than in forested streams in the Coastal Plain (Lowrance and others 1985). Increased nitrate concentrations and loads in agricultural streams compared to forested streams are common in the Appalachians (Hagen and others 2006), Piedmont (Crim 2007), and Coastal Plain (Lowrance and others 1985, Lehrter 2006). For example, nitrate loads were 1.5 to 4.4 times higher in watersheds with greater agricultural land than watersheds with less agricultural land (Lowrance and others 1985) and nitrate concentrations were 2.1 to 4.4 times higher in agricultural versus forested watersheds (Hagen and others 2006).

Effects of Forest Conversion on Human Health

Urban and agricultural land uses contribute to increased bacterial concentrations in stream waters. Connected stormwater and sewer overflow systems or failures in sewer systems (such as broken pipes, mechanical failures, and blockages from tree roots) can directly or indirectly release raw sewage into surface waters. Additionally, pet and wildlife feces can be transported in runoff over lawns and impervious surfaces in urban areas. Fecal coliform bacteria counts were higher in urban than forested or reference streams in the Coastal Plain (Mallin and others 2000, Holland and others 2004, DiDonato and others 2009), Piedmont (Schoonover and others 2005, Crim 2007), and Southern Appalachians (Clinton and Vose 2006). For example, Mallin and others (2000) report that fecal coliform is highly correlated with impervious surface ($r=0.975$, $p=0.005$), percent development ($r=0.945$, $p=0.015$), and population ($r=0.922$, $p=0.026$). Similarly, concentrations of *E. coli* can be much greater in urban watersheds than forested watersheds (Mallin and others 2000, Crim 2007). Urban watersheds in the Georgia Piedmont had the highest median *E. coli* concentrations (as measured in MPN or most probable number), ranging from 135 to 1255 MPN/100 mL, compared to median ranges of 94 to 169 MPN/100 mL for pine covered watersheds and 59 to 170 MPN/100 mL for oak-pine covered watersheds (Crim 2007). Becker (2006) reported that residential areas (2.2 percent of the watershed) did not appear to be a source of bacteria to Travertine Creek subbasin in Oklahoma, but attributed increased bacterial concentrations in the nearby Rock Creek basin to livestock grazing and sewage effluent in periods of high precipitation. Andrews and others (2009) reported fecal coliform counts as high as 10,000 colonies/100 mL with the highest counts in stormflow for the portions (42.17 percent) of the Illinois River basin in Oklahoma and Arkansas that are in agricultural use (pastures for cattle and confined feeding operations for poultry and swine); they also observed rapid urban development on the upper portion of this basin that may further impair water resources.

In addition to the danger of elevated concentrations of bacteria in surface waters, other health risks from urban and agricultural land uses include metals, pesticides, and personal care products (Klapproth and Johnson 2000, Paul and Meyer 2001). Trace metal sediment concentrations can be 2 to 10 times higher in streams near urban and industrial areas than in forested watersheds or suburban watersheds (Holland and others 2004); they tend to accumulate, rather than degrade, over time in sediments and plant and animal tissue (Klapproth and Johnson 2000). Metal concentrations may be inversely related to sediment particle size (Paul and Meyer 2001) and thus we might expect high concentrations of metals to be more problematic in the Southern Appalachians or Piedmont with more silty and clayey soils than in the Coastal Plain. Pesticides enter streams through runoff from agricultural and urban areas (Klapproth and Johnson 2000, Paul and Meyer 2001) again underscoring the importance of riparian buffers and other forested areas in slowing runoff and enhancing infiltration before contaminants can reach the stream. Personal care products including deodorants, perfumes, and pharmaceuticals may not be removed by traditional water treatment methods and are not as widely regulated as other substances (Kolpin and others 2002).

Effects of Forest Conversion on Aquatic Communities

Altered hydrology and channel-morphology and higher stream temperatures caused by forest conversion can dramatically affect aquatic communities. With increasing urban and/or agricultural uses, species richness and abundance often decline as reported for algae (Sponseller and others 2001), macroinvertebrates (Lenat and Crawford 1994, Paul and Meyer 2001, Roy and others 2003, Maloney and Feminella 2006, Helms 2008), fish (Onorato and others 1998, Walsh and others 2005), and amphibians (Orser and Shure 1979, Houlahan and Findlay 2003, Price and others 2006, Wang and others 2000). These effects may be offset somewhat for algae by additional stream nutrients (Biggs 1996, Chessman and others 1999) and for macroinvertebrates by perennial flows (Chadwick and others 2006). Mussels have virtually disappeared from some southern streams as a result of increased conversion of land to urban uses (Gillies and others 2003, Gangloff and Feminella 2007). These detrimental effects on aquatic organisms are particularly evident in the Southern Appalachians (Walters and others 2003, Scott et al 2002) where both diversity and endemism are very high (Wallace and others 1992). Cuffney and others (2010) found a high correlation of macroinvertebrate assemblages with urban metrics in eastern metropolitan areas (including Raleigh, NC, Atlanta, and Birmingham, AL), but not in central metropolitan areas such as Dallas-Fort Worth. Additionally, species composition may shift as sensitive species are replaced by more tolerant species or species better suited for the new conditions (Lenat and Crawford 1994, Weaver and Garman 1994, Onorato and others 1998, Sutherland and others 2002, Walters and others 2003, Roy and others 2005, Price and others 2006). For example, in urban streams of the western Georgia Piedmont, reptile species richness increased at the same time that the richness of salamanders and other amphibian species decreased (Barrett and Guyer, 2008).

Fish communities may also be strongly affected by changes in hydrology and water quality that are derived from land use change s. Higher velocities and discharge as well as increased sediment loads may degrade stream habitat to a significant degree (Nagy et al. in press). In particular, higher deposition of sediment in stream channels may reduce diversity of habitat with negative implications for some fish species. Reduced abundance of benthic feeders and lower spawning success in general may accompany such changes (Nagy et al. in press). In addition, near Columbus, GA, Helms et al. (2005) noted indicators of reduced fish health such as occurrence of lesions and tumors in fish from urban streams and a negative correlation between biotic integrity of fish and the proportion of impervious surface within watersheds.

Implications of Land Use Change Projections on Water Quality

The decreases in forest cover and increases in urbanization that are projected by 2060 carry important implications for water resources in the region. Losses of forest cover across much of the Piedmont of North Carolina, South Carolina, Georgia, and, to a lesser extent, Alabama (ch. 5) imply that further degradation of water quality and destabilization of surface water hydrology are likely in localized catchments within that subregion. In addition, it is likely that the alterations in the hydrologic cycles within the headwaters of major river basins (fig. 13-6) will affect conditions downstream. River basins and watersheds will undergo reductions in evapotranspiration due to lower leaf area indices as well as increases in impervious surfaces. Consequently, responses to deforestation—reduced infiltration, increased runoff, reduced baseflow, and increased discharge and velocity—that already exist on the Piedmont to some extent (but are more often associated with steeper terrains) will likely be exacerbated.

These hydrologic responses, combined with the increased quantities of potential pollutants in urbanizing watersheds, will increase streamwater concentrations and/or loads of sediment, nutrients, pathogens, and various chemicals. The result could be significant degradation of water quality within river basins and stream systems, and concurrent negative effects on diversity of aquatic organisms. Although responses will be manifested throughout river basins, cumulative effects will magnify the trends along their lower reaches. If streams remain connected to the floodplain forests that lie below the physiographic fall line, some fraction of pollutant loads may be filtered as sediment is deposited by spreading floodwaters. However, although upper coastal forests are projected to be cleared to a lesser extent than Piedmont forests, they remain at some risk of conversion with subsequent reduction of pollutant filtration potential on floodplains. In addition, the increased velocity of streams and rivers draining highly urbanized upper reaches will tend to increase channel incisement, thereby reducing the filtering benefits of overbank flooding and sediment deposition. Although reservoirs created by dams may trap significant amounts of sediment and other substances, they have a finite capacity for sediment filling, which is already being approached in some areas. Consequently, loads of sediment, nutrients, and pathogens will likely increase in lower reaches of river basins, with exports of these materials expected to elevate levels in coastal estuaries as well.

Another trend is the large loss of forest cover that is expected within Gulf of Mexico and Atlantic coastal counties from southern Texas, through parts of Louisiana, Mississippi, Alabama, Florida, Georgia, South and North Carolina, to southern Virginia (ch. 5). Although many of these counties are already experiencing varying degrees of urbanization, additional development is projected, bringing major changes to freshwater resources as well as new hydrologic, biogeochemical, and other inputs to coastal estuaries. Outputs from large rivers are mainly driven by large-scale interactions between land-use/land-cover and climate throughout basins that may extend far northward and touch multiple states. Meanwhile, numerous lower order coastal streams will be directly impacted by reductions in forest cover and increased impervious surfaces at local scales—placing them at risk to increased levels of nitrate, pathogens, and other substances (Nagy, R.C.; Lockaby, B.G.; Kalin, L.; and Anderson, C. manuscript in preparation. Urbanization of a coastal region and the effects on water resources. Authors can be reached at Brown University, Department of Ecology and Evolutionary Biology, 80 Waterman St., Providence, RI 02912; rachel_nagy@brown.edu.). Given the proximity of groundwater tables to the surface in some coastal areas, the risk may extend to those waters as well. Elevated exports of nonpoint source pollutants from large rivers and lower order coastal streams can significantly increase eutrophication of coastal waters and risks to humans from contaminated seafood and direct contact while swimming.

Apart from scattered locations near Dallas, Houston, Little Rock, AR, and Oklahoma City, OK, little change in the proportion of forest land is expected across much of southwestern Alabama, and most of Mississippi, Louisiana, Arkansas, and eastern Texas and Oklahoma (ch. 5). This stabilization of forest area may preclude further declines in water quality in the Lower Mississippi Alluvial Valley, an area that has traditionally been associated with heavy exports of sediment and nutrients into the Mississippi River Basin; however improvements in water quality appear unlikely without the major increases in forest land (15 to 27 million acres) that would be required to substantially reduce nonpoint source pollutant exports (Mitsch and others 2001). Increased forest coverage will also be characteristic on many inland Coastal Plain areas of Virginia, North Carolina, South Carolina, Georgia, Alabama, Mississippi, and Louisiana (ch. 5), a trend that could protect floodplain forests and maintain water quality of those systems.

Another location that is anticipated to undergo major increases in urbanization is the southern half of Florida, where forested wetlands (both riparian and depressional) are prevalent and associated water quality functions are at risk. At a time when additional nonpoint source pollutant exports are originating from newly urbanized landscapes, a portion of the natural systems with potential to filter pollutant loads will disappear with the demise of forested wetlands.

Effects of Expanded Intensive Forest Management on Water

The establishment of pine plantations has resulted in vast acreages of intensively managed pine forests in the South. Plantation based forestry–using pine and fast growing hardwood species–is likely to increase in the future (ch. 9), and demand from a shrinking land base and emerging wood fiber markets for bioenergy is likely to increase management intensity on new and established plantations. Considerable information is available on the impacts of forest management on streamflow throughout the United States (Jones and Post, 2004, Brown and others 2005). For example, removing the forest canopy increases streamflow for the first few years, but the magnitude, timing, and duration of the response varies considerably among ecosystems. In some, streamflow returns to preharvest levels within 10 to 20 years; whereas in others, streamflow remains higher for several decades after cutting, or can even drop lower than pre-harvest levels (Jackson and others 2004). This wide variation in responses is attributable to the complex interactions between climate, which can vary considerably from dry to wet regimes, and vegetation, which can vary in structure and phenology (coniferous versus deciduous forest).

Information on the relationships between specific ecosystems or forest types and streamflow can be inferred from studies quantifying annual evapotranspiration. At annual time scales, streamflow is approximated by the difference between precipitation (PPT) and evapotranspiration (ET), streamflow = PPT - ET. Therefore, for a given amount of precipitation, management actions that alter evapotranspiration will also alter streamflow. It is well established that coniferous forests, with their greater capacity for interception and transpiration, have higher evapotranspiration (and hence lower streamflow) than deciduous hardwood forests (Swank and Douglass 1974, Ford and others *in press*). Averaged across several climate regimes and forest types, the difference between coniferous and hardwood forests is about 55 percent at 1200 mm yr^{-1} precipitation and increasing precipitation widens this difference in the two forest types. Evapotranspiration also varies considerably between managed and unmanaged southern forests (table 13-4). This variation is important for evaluating the implications of increasing pine plantation forests in the South because the magnitude of the effects on streamflow depends on the species, forest type, or land use being replaced. For example, pine plantations may consume nearly twice the water consumed by longleaf pine savannas (table 13-4).

Implications of increasing management intensity on water resources will depend on the specific management activity. Increasing acreages of fast growing species for bioenergy production or carbon sequestration may have negative consequences for water yield (Farley and others 2005, Jackson, R.B. and others 2005). To illustrate, a mature *Eucalyptus* plantation (age 5, 1,111 trees ha^{-1}, leaf area index of 6 m^2 m^{-2}) growing in southwestern GA could potentially consume 882 mm yr^{-1} of water, exceeding other forest types by a factor of 2.5 (table 13-4). Nitrogen fertilization improves productivity primarily though increased leaf area (Vose and Allen 1988), and evapotranspiration is highly correlated with leaf area index (Sun and others *in press*). Shortening rotation times usually increases streamflow by decreasing the amount of time that the stand is at canopy closure, when leaf area index is highest and streamflow is lowest. For any given leaf area, younger or shorter trees also have higher stomatal conductance than older or taller trees (Schafer and others 2000, Moore and others 2004, Novick and others 2009). Although transpiration per unit leaf area is less than for younger forests, older forests have larger leaf area and can intercept more water, and therefore have greater evapotranspiration. This means that managing for older forests is likely to decrease streamflow.

Impacts on water quality will depend on the type of management activity and the effectiveness of established Best Management Practices, which were originally developed for less intensive management. For example, in review of the impacts of forests fertilization, Fox and others (2007) concluded that correctly applied fertilizer rarely degrades water quality. In contrast, increasing the frequency of harvest for shorter rotations may have impacts on sediment yield, especially if the harvests result in greater soil disturbance (Ursic 1986) or require more roads and more frequent road usage (Swift 1988).

Implications of Climate Change, Land Use Change, and Population on Water Resources

Climate change impacts on water resources—Because of the combination of biological and physical controls on hydrologic processes, climate change will both directly and indirectly impact southern water resources (Brian and others 2004, Sun and others 2009). The direct impacts will depend on how the amount and timing of precipitation are altered and how this influences baseflow, stormflow, groundwater recharge, and flooding. Long-term U.S. Geological Survey streamflow data suggest that average annual streamflow has increased and that this increase has been linked to greater precipitation in eastern States over the past 100 years (Lins and Slack 1999, Karl and Knight 1998, Intergovernmental Panel on Climate Change 2007); however, fewer than 66 percent of all General Circulation Models can agree on the direction of predicted precipitation change, whether wetter or drier (Intergovernmental Panel on Climate Change 2007). Annual precipitation within a year or from one year to the next is a natural phenomenon related to large-scale global climate teleconnections, such as El Niño Southern Oscillation, Pacific Decadal Oscillation, and North Atlantic Oscillation cycles. Many regions of the United States have experienced an increased frequency of precipitation extremes over the last 50 years (Easterling and others 2000a, Huntington, 2006, Intergovernmental Panel on Climate Change 2007). As the climate warms in most General Circulation Models, the frequency of extreme precipitation events increases across the globe (O'Gorman and Schneider, 2009); however, the timing and spatial distribution of extreme events are among the most uncertain aspects of future climate scenarios (Karl and Knight 1998, Allen and Ingram, 2002). Despite this uncertainty, recent experience with droughts and low flows in many areas of the United States indicate that even small changes in drought severity and frequency will have a major impact on society, among them a reduction in drinking water supplies (Easterling and others 2000, Luce and Holden, 2009).

The indirect impacts of climate change are related to changes in temperature and atmospheric carbon dioxide. In the short term, higher temperatures have the potential to increase evaporation and plant water use via transpiration (as temperatures increase, the energy available for evapotranspiration increases), and therefore decrease excess precipitation available for streamflow or groundwater recharge. Warmer temperatures will also influence the duration and timing of snowmelt, a critical factor in ecosystems where snowmelt dominates hydrologic processes. The impacts of temperature may be offset (or exacerbated) by changes in other factors that influence evapotranspiration such as vapor pressure (warm air holds more water), wind patterns (which impact boundary layer resistance), increases in carbon dioxide (which decrease stomatal conductance), and changes in net radiation (influenced by changes in cloud cover and aerosols). In the longer term, a warmer climate in combination with changes in precipitation will likely shift distributions of tree species, which differ considerably in the amount of annual and seasonal water they use via transpiration and interception (Ford and others 2011, Sun and others*,* 2011). For example, in some geographic areas, a shift from hardwood to pine forests may result in year-round transpiration and interception and greater water use. Controlled studies have demonstrated that increased atmospheric carbon dioxide reduces transpiration in many tree species, which may translate into increased streamflow (Ainsworth and Rogers, 2007); however, it is not certain that these patterns will persist over the long-term.

Modeled Impacts of Future Climate Change on Water Resources —The impacts of climate change on water resources are complex and variable over space and time (Dale et al. 2001). In addition, changes in land cover and human demands for water resources are likely to be complicating factors. For these reasons, models are often useful for integrating complex interactions across multiple scales. Past forest hydrological studies using small experimental watersheds clearly show that climate variability and land cover can substantially impact water quantity and quality. However, at the large basin or regional scale, the magnitude and areal sizes of disturbances will determine how water resources will be affected by climate change, forest clearing in preparation for urbanization, or disease/bioenergy-induced changes in species distribution. Over the entire southern region, WaSSI model simulations reveal that on average (year 2002-2007) forests represented 27 percent of the land cover but were the source of 34 percent of the total water yield (fig. 7).

Future global climate change is expected to have regional impacts, but the severity depends on the magnitude of changes in both precipitation and atmospheric warming. Key findings from multiple scenario modeling (table 13-2) by the WaSSI model are summarized below to show the projected impacts of climate change and other contributors to water stress, or imbalance between supply and demand, around the year 2050.

1) Average water stress in the South is low (WaSSI = 0.16) but high in southern and western Texas (WaSSI > 0.90) because of naturally low precipitation and high evapotranspiration (fig. 13-8). A few isolated basins also show moderate water stress (WaSSI 0.4 – 0.9), primarily near population centers and other areas of high water demand.
2) The highest water stress occurs during the growing season when ecosystem water use and human water withdrawal are the highest (fig. 13-9). In particular, irrigation, domestic, and thermoelectic uses are greatest during the summer months. All future climate change scenarios will likely increase monthly WaSSI relative to historic levels across the region and may shift the timing of peak WaSSI from late summer to early fall.
3) Population will increase by 104 percent by 2050 in the South as a whole (NPA Data Services 1999). Population growth alone will increase water stress 10 to 50 percent in much of the Piedmont and Coastal Plain, with increases of 50 to 100 percent in the Florida Panhandle (fig. 13-10). On average, population growth will increase water stress by about 12 percent.

4) Land use change alone may increase or decrease water stress, depending on the historic and future use for a given land area. For example, in areas converted from forest to urban use, water stress will likely decrease due to reductions in evapotranspiration. However, areas converted from forest to crop use will likely experience increases in water stress due to higher irrigation water demand. By 2050, land use change alone is not likely to significantly change water stress in the South as a whole (fig. 13-11).

5) Population growth and land use change will produce an array of effects on water stress across the South and may aggravate water shortages at the regional scale (fig. 13-12). On average, water stress due to the combined effects of population and land use change will increase in the South by 10 percent.

6) All climate change scenarios predicted that the South would likely see increases in air temperature in the next 50 years but differed in predictions of precipitation change across the region. The combined effects of changing temperature and precipitation will generally decrease streamflow across the South (fig. 13-13). In addition, streamflow will likely become more variable with lower flows during drought periods and higher flows during wet periods than experienced in the past.

7) Water supply stress would likely increase significantly under all four climate change scenarios (fig. 13-14), largely caused by increases in water loss by evapotranspiration resulting from higher air temperatures, and also because of decreasing precipitation in some areas. The effects of changing climate on water stress will vary significantly across the region (fig. 13-15). For example, the WaSSI model projects that Frankfort, KY will have negligible change in water stress across the four future climate scenarios, while Oklahoma City, OK, Little Rock, AR, and Austin, TX are projected to have significant increases in water stress.

Implications of Sea Level Change on Coastal Areas

Sea-level may rise from 0.4 to 2.0 m by the end of the 21st century (table 13-5) (Rahmsorf 2007, McMullen and Jabbour 2009, Solomon and others 2009). Along the Atlantic Coast in the study region there is approximately7, 297 square miles (~ 4.6 million acres) of coastal land below an elevation of 1.5 meters (North Carolina and Florida have the most coastal area below 1.5 m), with an additional 5,573 square miles (~ 3.5 million acres) of coastal land between 1.5 and 3.5 m. Along the Gulf Coast there is approximately 13,605 square miles (~ 8.7 million acres) of land below an elevation of 1.5 m (Louisiana and Texas have the most coastal area below 1.5 m), with an additional 6,430 square miles (~ 4.1 million acres) of coastal land between 1.5 and 3.5 m (fig. 13-16). If sea level rose 1.5 m we estimate that 2,633 square miles (~1.6 million acres) of forests could be affected along the Atlantic Coast, and 3,352 square miles (~ 2.1 million acres) of forests could be impacted along the Gulf Coast. When physical processes are considered by the coastal vulnerability index, along the Atlantic Coast North Carolina and Virginia have the most coastline in the very high-risk class, and along the Gulf Coast, Louisiana and Texas have the most coastline in the very high-risk class (fig. 13-17).

Projections of sea level changes can help managers identify portions of the coastline that could be monitored more closely. For example, figure 13-18 shows that the entire Louisiana coastline is in the high risk category with coastal area below 1.5 m, but the Gulf coast portion of southern Florida is ranked in the moderate risk category even though its coastal area is also below 1.5 m, suggesting that its response to a rising sea may be slower than if predicted from elevation alone (Thieler and Hammar-Klose, 2000). Figure 13-19 shows that portions of the North Carolina coastline and the Atlantic coast of Florida are in the high risk category, but because those coastal areas are between 1.5 and 3.5 m, a sea-level rise of 1 m may not affect those higher elevation areas.

Discussion and Conclusions

Forest conversion to agriculture or urban land uses consistently causes increases in discharge, peak flow, and velocity of streams. Differences in the nature of hydrologic responses to urbanization among subregions are substantial. As examples, the pronounced effect of urban development on peak flows and stream hydrographs found in the Appalachians and Piedmont may be obscured by natural precipitation regimes in arid regions, such as western Texas where hydrographs from less disturbed streams resemble those of urban streams (Grimm and others 2004). Similarly, the reductions in baseflow that are often observed following increases in impervious area in the Piedmont may not occur in the flatter terrain of the Coastal Plain.

Forest conversions also result in increases in sediment, water chemistry indices, fecal coliform and *E. coli*, and other substances. Because discharge and concentrations increase after urbanization, loads are generally higher. Physiographic characteristics such as slope and soil texture strongly influence hydrology and sediment export, but their impact on water chemistry is less than the impact of urbanization. Conversion of forest land to urban uses may result in health risks for humans as evidenced by large increases in fecal coliform and e. coli, heavy metals, pharmaceuticals, and other substances in stream water. While effective water treatment may overcome this risk to drinking water, there remains significant potential for direct contact with polluted water as streams flow through residential areas prior to treatment.

Each river basin has a unique land use history that may have long-lasting effects. Cuffney and others (2010) reported that the conversion of forest to urban land had more pronounced effects on benthic macroinvertebrates in Atlanta, Birmingham, AL, and Raleigh, NC than the conversion of agriculture to urban land had in Dallas, where natural grassland had already been degraded by agriculture in the recent past (an example antecedent land use impacts taking precedent over historical land use). In fact, their study found that antecedent agricultural land use masked the effects of urbanization in areas of historic forest use as well.

Physiographic characteristics could determine the threshold, or the resilience to change, that each subregion displays in response to changes in land use. For this reason, McMahon and Harned (1998) recommended incorporating measures of both natural physiographic variation and effects of human activity in watershed studies and management plans. Additionally, there have been some indications that impervious surface increases may have higher thresholds for significant water degradation in the Coastal Plain than the Piedmont or Southern Appalachians (Stednick 1996, Roy and others 2003, Morgan and Cushman 2005, Helms and others 2009, Utz and others 2009); but this does not appear to be true for all measures (Nagy, R.C.; Lockaby, B.G.; Kalin, L.; and Anderson, C. Manuscript in preparation. Urbanization of a coastal region and the effects on water resources. Authors can be reached at Brown University, Department of Ecology and Evolutionary Biology, 80 Forest St., Providence, RI, 02912; rachel_nagy@brown.edu.).

The concept of thresholds of imperviousness beyond which significant degradation of water quality occurs is vague and had previously been reported at 10-20% (Arnold and Gibbons 1996, Bledsoe and Watson 2001). However, some reports have noted significant changes in water quality at even lower levels of development, such as <5 percent impervious surface (Crim, 2007, Cuffney and others 2010, Nagy, R.C.; Lockaby, B.G.; Kalin, L.; and Anderson, C. Manuscript in preparation. Urbanization of a coastal region and the effects on water resources. Authors can be reached at Brown University, Department of Ecology and Evolutionary Biology, 80 Waterman St., Providence, RI, 02912; rachel_nagy@brown.edu.). For instance, at 5 percent impervious surface, Cuffney and others (2010) estimated a 13 to 23 percent degradation of macroinvertebrate assemblages compared to background conditions. This suggests that care must be taken from the first stages of development to limit impacts on water resources. Boggs and Sun (2011) suggest that maintaining high ET of vegetation in the growing season is key to reducing stormflow in urban watersheds. Furthermore, once impervious surface cover exceeds 30 percent, deterioration of water quality becomes severe (Paul and Meyer, 2001, Calhoun and others 2003). In areas where development is planned or about to begin, it would be very useful to identify key bioindicators for detecting the onset of significant degradation.

Among the most dramatic impacts associated with forest conversion to urban or agriculture are changes in aquatic populations. The higher velocity and channel scouring associated with urban hydrology creates unstable habitat, and this is compounded by the effects of degraded water quality. Species richness and abundance generally decline; and some groups, such as mussels, may be eliminated from particular locations. These impacts tend to be most severe in the Appalachians. Also, species that are tolerant of the altered conditions may replace those that are intolerant. An example was observed in the Georgia Piedmont as reptile species richness increased after urbanization, while amphibian richness decreased (Barrett and Guyer, 2008).

Increased intensification of forest management on a smaller land base could have impacts on quantity and quality of water, especially at local scales. In general, an increase in pine plantations or fast growing hardwood species may result in greater water use via transpiration (Ford and others, in press); however, the magnitude and significance of greater transpiration on water resources will depend on the community type that is being replaced and on site specific hydrologic processes. In addition, the impact of greater water use may be offset by a net reduction of forest cover. Increased intensification of forest management activities that create more severe or frequent soil disturbance–such as site preparation, increased harvest frequencies, a larger road network, or more traffic–may result in increased sediment and reduced water quality if Best Management Practices are bypassed.

Based on WaSSI model results under the four future climate scenarios considered in this chapter, stream flows and water supply will generally decrease and become more variable over the next 50 to 100 years. However, magnitudes and even the signs of changes in stream flows resulting from climate change will vary considerably across the region, with some small areas, such as western Texas, experiencing increases in water supply. Other areas will likely experience decreases in supply, particularly in Florida, Oklahoma, and northern Texas. Overall, climate-induced decreases in water supply and increased demand from a growing human population will likely result in an increase in water supply stress into the next century.

Considerable variability of water resource predictions among the future climate scenarios and the absence of overlapping predictions for any particular subregion confound the certainty of future projections. Despite these uncertainties, the importance of water resources for human and aquatic life argues for further research and active management.

Our projections indicate a greater risk of sea level rise for many coastal areas in this century. Thermal inertia dictates that once the waters rise, curbs is future greenhouse gas emissions will not produce a quick reversal. Therefore, unlike precipitation driven flood events, flooding due to sea level rise will have long-term consequences. Coastal inundation is one of the most visible impacts of rising sea levels. Areas that were once dry further inland will gradually shift to episodically inundated (during high tides and storms) and then to permanently inundated. The impact of sea level rise to the point of inundation is obvious, but other impacts may be less visible, such as the salt water marshes that exemplify an ecosystem in balance between fresh water and saline environments. These unique places provide important breeding habitat for many terrestrial and aquatic animal species. However, rapidly rising sea levels will permeate non-saline forests and grasslands, causing losses existing vegetation without the possibility of replacement by more salt tolerant species. Once the existing vegetation is dead, the root structure that binds the soil system together and provides a buffer from incoming tides will also be lost, and coastal erosion is likely to accelerate. Although coastal erosion is a naturally occurring process in barrier islands and many other areas, the increase in rate and severity that is likely with rising sea levels could result in a greatly accelerated loss of valuable coastal property.

Finally, a combination of pressure on water resources from increasing human populations and rising sea levels could severely reduce fresh water supplies along coastal areas. As fresh water is drawn out of shallow ground water systems, adjacent brackish water would likely fill the void, thus raising the risk of salt water contamination to drinking water supplies. Rises in sea level will further increase the risk of contamination as saline water levels rise. The loss of ground water supplies in places like Florida would have enormous social and economic implications and may be more significant than coastal inundation in the near to medium future.

Knowledge and Information Gaps

Past studies on forest-water relations that have been conducted primarily in forested watersheds are not sufficient to address issues in more complex, human dominated landscapes. A key issue is the relationship between increasing urbanization and diminishing available water supply for humans in the South. We need to understand more about the nature of this relationship and how it may change across the array of southern physiographic features. Complexity increases with the interactive effects of multiple drivers including land use change, climate change, population growth, and the natural variability in the hydrologic cycle. A better understanding is critically needed in advance of the next major drought, whose impacts may exacerbated by expected increases in human populations and impervious surfaces in many areas of the South(ch. 4).

The ramifications of urbanization on surface water and subsequently, human health is another topic that deserves greater attention. Very high counts of fecal coliform and *E. coli* have been documented in urban streams, but the potential risks to human health have not yet been assessed. Research is needed on coastal areas, which have not been adequately studied and are expected to undergo high population growth and development rates in coming years. Also, the sensitivities of aquatic organisms to urbanization have been demonstrated but not quantified, and should be more fully understood so that they can serve as bioindicators of impending degradation to surface water resources.

The focus of this chapter was on surface water impacts associated with land use conversion, but literature searches produced little information on the relationships of groundwater to land use and land cover. Because many southern communities are considering expanded use of aquifers, believing them to be “drought proof”, they will need to understand the extent to which changes in land use might affect groundwater resources.

The WaSSI model provides a general summary of water supply and demand dynamics across large regions over extended periods of time, requiring assimilation and integration of large datasets and the use of extensive GIS and computing resources. These large data requirements necessitated development of simplifying assumptions to simulate water resource changes in response to climate change. For example, the WaSSI model used for this chapter does not include provisions for water supply reservoir storage or interbasin water transfers; it also assumes that all in-stream surface water is available for human use (no ecological flow is reserved) and that river flows are routed through the river network instantaneously during a given month. It is important to keep in mind that these assumptions may impact water supply stress predictions for some areas across the region. Future land use changes are likely to affect water quality and extreme hydrology such as peakflow rate, issues that are not addressed yet by the WaSSI model. The tradeoffs between water resources and carbon sequestration are not well understood and need to be quantified before embarking on bioenergy development and forest management to mitigate climate warming.

Compared to the physics of oceanic thermal expansion, relatively little is known about the rate of global warming, the changes in ocean surface albedo, or the input of water from snow and ice melts on land. Many of these unknowns are not a function of science gaps, but rather uncertainty about future increases in greenhouse gas emissions. Conversely, the physics of thermal expansion are well understood. As predictions of global warming rates improve, the accuracy of sea-level rise will also improve significantly. Finally, demographic changes and associated pressures on ground water resources are also unknown. These knowledge gaps need to be addressed before a more complete assessment of climate change on sea-level rise is possible.

Acknowledgements

Literature Cited

Ainslie, W.B. 2002. Forested wetlands. In: Wear, D.N.; Greis, J.G., eds. Southern forest resource assessment: summary report. Gen. Tech, Rep. SRS–54. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station: 479–500

Ainsworth, E.A.; Rogers, A. 2007. The response of photosynthesis and stomatal conductance to rising [CO2]: mechanisms and environmental interactions. Plant, Cell and Environment. 30: 258–270. [DOI: 10.1111/j.1365-3040.2007.01641.x].

Allen, M.R.; Ingram, W.J. 2002. Constraints on future changes in climate and the hydrologic cycle. Nature. 419: 224–232.

Amatya, D.M.; Callahan, T.J.; Radecki-Pawlik, A. [and others]. 2008. Hydrologic and water quality monitoring on Turkey Creek watershed, Francis Marion National Forest, SC. In: Proceedings of the

South Carolina water resources conference. Clemson, SC: Clemson University. http://www.clemson.edu/restoration/events/past_events/sc_water_resources/t3_proceedings_presentations/t3_zip/williams.pdf. [Date accessed: October 28, 2010].

Amatya, D.M.; Sun, G.; Skaggs, R.W. [and others]. 2006. Hydrologic effects of global climate change on a large drained pine forest. In: Williams, T., ed. Hydrology and management of forested wetlands: Proceedings of the international conference. St. Joseph, MI: American Society of Agricultural and Biological Engineers: 583–594.

Anderson, S.A.; Masters, R.E. 1992. Riparian forest buffers. Fact Sheet 5034. Stillwater, OK: Oklahoma State University, Cooperative Extension Service. 7 p.

Andrews, W.J.; Becker, M.F.; Smith, S.J.; Tortorelli, R.L. 2009. Summary of surface-water quality data from the Illinois River basin in north Oklahoma, 1970–2007. Rev. Sci. Invest. Rep. 2009–5182. Reston, VA: U.S. Geological Survey. 39 p.

Arnold, C.L., Jr.; Gibbons, C.J. 1996. Impervious surface cover: the emergence of a key environmental indicator. Journal of the American Planning Association. 62: 243–258.

Bailey, R.G. 1980. Description of the ecoregions of the United States. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture Forest Service. 77 p.

Barrett, K.; Guyer, C. 2008. Differential responses of amphibians and reptiles in riparian and stream habitats to land use disturbances in western Georgia, USA. Biological Conservation. 141: 2290–2300.

Becker, C.J. 2006. Water quality and possible sources of nitrogen and bacteria to Rock and Travertine Creeks, Chickasaw National Recreation Area, Oklahoma, 2004. Sci. Invest. Rep. 2005–5279. Reston, VA: U.S. Geological Survey. 24 p.

Beighley, R.E.; Melack, J.M.; Dunne, T. 2003. Impacts of California's climatic regimes and coastal land use change on streamflow characteristics. Journal of the American Water Resources Association. 39: 1419–1433.

Biggs, B.J.F. 1996. Patterns in benthic algae of streams. In: Stevenson, R.J.; Brothwell, M.L.; Lowe, R.L., eds. Algal ecology: freshwater benthic ecosystems. San Diego: Academic Press: 31-56.

Binkley, D.; Brown, T.C. 1993. Management impacts on water quality of forests and rangelands. Gen. Tech. Rep. RM–239. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Forest and Range Experiment Station. 114 p.

Bledsoe, B.P.; Watson, C.C. 2001. Effects of urbanization of channel instability. Journal of the American Water Resources Association. 37: 255–270.

Blevins, D.W. 2004. Hydrology and cycling of nitrogen and phosphorous in Little Bean Marsh: a remnant riparian wetland along the Missouri River in Platte County, Missouri, 1996–97. Sci. Invest. Rep. 2004–5171. Reston, VA: U.S. Geological Survey. 78 p.

Boggs, J. ; Sun, G. 2011. Urbanization alters watershed hydrology in the Piedmont of North Carolina. Ecohydology (in press)

Bolstad, P.V.; Swank, W.T. 1997. Cumulative impacts of landuse on water quality in a Southern Appalachian watershed. Journal of the American Water Resources Association. 33(3): 519–533.

Bosch, J.M.; Hewlett, J.D. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology. 55: 3–23.

Brian, H.H.; Callaway, M.; Smith, J.; Karshen, P. 2004. Climate change and U.S. water resources: from modeled watershed impacts to national estimates. Journal of the American Water Resources Association. 40: 129–148.

Brinson, M.M. 1993. A hydrogeomorphic classification for wetlands. Tech. Rep. WRP–DE–4. Vicksburg, MS: U.S. Army Corps of Engineers, Waterways Experiment Station. 101 p.

Brown, A.E.; Zhang, L.; McMahon, T.A. [and others]. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology 310: 28–61.

Brown, T.C.; Hobbins, M.T.; Ramirez, J.A. 2008. Spatial distribution of water supply in the conterminous United States. Journal of the American Water Resources Association. 44: 1474–1487.

Calhoun, D.L.; Frick, E.A.; Buell, G.R. 2003. Effects of urban development on nutrient loads and streamflow, upper Chattahoochee River basin, Georgia, 1976–2001. In: Hatcher, K.J., ed. Proceedings of the 2003 Georgia water resources conference. Athens, GA: The University of Georgia, Institute of Ecology: 5 p.

Casarim, F. 2009. Legacy sediments in Southeastern United States Coastal Plain streams. Auburn, AL: Auburn University. 155 p. M.S. thesis.

Chadwick, M.A.; Dobberfuhl, D.R.; Benke, A.C. [and others]. 2006. Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness. Ecological Applications. 16: 1796–1807.

Chessman, B.; Growns, I.; Currey, J.; Plunkett-Cole, N. 1999. Predicting diatom communities at the genus level for the rapid biological assessment of rivers. Freshwater Biology. 41: 317–331.

Childers, D.L.; Gosselink, J.G. 1990. Assessment of cumulative impacts to water quality in a forested wetland landscape. Journal of Environmental Quality. 19: 455–464.

Clinton, B.D.; Vose, J.M. 2006. Variation in stream water quality in an urban headwater stream in the Southern Appalachians. Water, Air, and Soil Pollution. 169: 331–353.

Crim, J. F. 2007. Water quality changes across an urban-rural land use gradient in streams of the west Georgia piedmont. Auburn, AL: Auburn University. 130 p. M.S. thesis.

Cuffney, T.F.; Brightbill, R.A.; May, J.T.; Waite, I.R. 2010. Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. Ecological Applications. 20(5): 1384–1401. [DOI: 10.1890/08-1311].

Dale, V.H.; Joyce, J.A.; McNulty, S. [and others]. 2001. Climate change and forest disturbances. BioScience. 51: 723–734.

De la Cretaz, A.L.; Barten, P.K. 2007. Land use effects on streamflow and water quality in the Northeastern United States. New York: CRC Press. 319 p.

Dennis, J.V. 1988. The great cypress swamps. Baton Rouge, LA: Louisiana State University Press. 232 p.

DiDonato, G.T.; Stewart, J.R.; Sanger, D.M. [and others]. 2009. Effects of changing land use on the microbial water quality of tidal creeks. Marine Pollution Bulletin. 58: 97–106.

Duryea, M.L.; Hermansen, A. 1997. Cypress: Florida's majestic and beneficial wetlands tree. CIR 1186. Gainesville, FL: University of Florida, Florida Cooperative Extension Service, Institute of Food and Agricultural Services. 12 p. http://edis.ifas.ufl.edu. [Date accessed: March 15, 2009].

Easterling, D.R.; Evans, J.L.; Groisman, P.Y. [and others]. 2000. Observed variability and trends in extreme climate events: a brief review. Bulletin of the American Meteorological Society. 81: 417–425.

Ernst, K.A.; Brooks, J.R. 2003. Prolonged flooding decreased stem density, tree size and shifted composition towards clonal species in a central Florida hardwood swamp. Forest Ecology and Management. 173: 261–279. [DOI:10.1016/S0378-1127(02)00004-X].

Exum, L.R.; Bird, S.L.; Harrison, J.; Perkins, C.A. 2005. Estimating and projecting impervious cover in the Southeastern United States. EPA/600/R–05/061. Athens, GA: U.S. Environmental Protection Agency, Ecosystems Research Division. 133 p.

Farley, K.; Jobbagy, E.; Jackson, R.B. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology. 11: 1565–1576.

Faulkner, S. 2004. Urbanization impacts on the structure and function of forested wetlands. Urban Ecosystems. 7: 89–106.

Ford, C.R.; Brooks, J.R. 2002. Detecting ecosystem response to increasing river flow in southwest Florida, USA. Forest Ecology and Management. 160: 45–64.

Ford, C.R.; Hubbard, R.M.; Vose, J.M. 2011. Quantifying structural and physiological controls on canopy transpiration of planted pine and hardwood stand species in the Southern Appalachians. Ecohydrology. [DOI:10.1002/eco.136].

Ford, C.R.; Mitchell, R.J.; Teskey, R.O. 2008. Water table depth affects productivity, water use, and the response to nitrogen addition in a savanna system. Canadian Journal of Forest Research. 38: 2118–2127.

Fox, T.R.; Allen, H.L.; Albaugh, T.J. [and others]. 2007. Forest fertilization and water quality in the United States. Better Crops. 91: 5–7.

Gangloff, M.M.; Feminella, J.W. 2007. Stream channel geomorphology influences mussel abundance in Southern Appalachian streams, USA. Freshwater Biology. 52: 64–74.

Gillies, R.R.; Brim Box, J.; Symanzik, J.; Rodemaker, E.J. 2003. Effects of urbanization on the aquatic fauna of the Line Creek watershed, Atlanta - a satellite perspective. Remote Sensing of Environment. 86: 411–422.

Grace, J.M. 2005. Forest operations and water quality in the South. Transactions of the ASAE, 48(2): 871–880.

Gresham, C.A. 1989. A literature review on developing pocosins. In: Hook, D.D.; Lea, R., eds. Proceedings of the symposium: the forested wetlands of the Southern United States. Gen. Tech. Rep. SE–50.

Asheville, NC: U.S. Department of Agriculture Forest Service, Southeastern Forest Experiment Station: 44–50.

Grimm, N.B.; Arrowsmith, J.R.; Eisinger, C. [and others]. 2004. Effects of urbanization on nutrient biogeochemistry of aridland streams. In: Defries, R.; Asner, G.; Houghton, R., eds. Ecosystem interactions with land use change. Geophys. Monogr. Ser. 153. Washington, DC: American Geophysical Union: 129–146.

Groffman, P.M.; Bain, D.J.; Band, L.E. [and others]. 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment. 1(6): 315–321.

Hagen, E.M.; Webster, J.R.; Benfield, E.F. 2006. Are leaf breakdown rates a useful measure of stream integrity along an agricultural landuse gradient? Journal of North American Benthological Society. 25(2): 330–343.

Hammar-Klose, E.S.; Thieler, E.R. 2001. Coastal vulnerability to sea-level rise: a preliminary database for the U.S. Atlantic, Pacific and Gulf of Mexico Coasts [CD–ROM]. Digital Data Ser. DDS–68. http://pubs.usgs.gov/dds/dds68. [Date accessed: October 28, 2010].

Helms, B.S., J.W. Feminella, and S. Pan. 2005. Detection of biotic responses to urbanization using fish assemblages from small streams of western Georgia, USA. Urban Ecosystems 8:39-57.

Helms, B. S. 2008. Response of aquatic biota to changing land use patterns in streams of west Georgia, USA. Auburn University: Auburn, AL. 224 p. Ph.D. Dissertation.

Helms, B.S.; Schoonover, J.E.; Feminella, J.W. 2009. Seasonal variability of landuse impacts on macroinvertebrate assemblages in streams of western Georgia, USA. Journal of the North American Benthological Society. 28(4): 991–1006.

Hibbert, A.R. 1967. Forest treatment effects on water yield. In: Sopper, W.E.; Lull, H.W., eds. Forest hydrology: Proceedings of a National Science Foundation advanced science seminar. New York: Pergamon Press: 527–543.

Holland, A.F.; Sanger, D.M.; Gawle, C.P. [and others]. 2004. Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. Journal of Experimental Marine Biology and Ecology. 298: 151–178.

Houlahan, J.E.; Findlay, C.S. 2003. The effects of adjacent land use on wetland amphibian species richness and community composition. Canadian Journal of Fisheries and Aquatic Sciences. 60: 1078–1094.

Huntington, T.G. 2006. Evidence for intensification of the global water cycle: review and synthesis. Journal of Hydrology. 319: 83–95.

Intergovernmental Panel on Climate Change. 2007. Contribution of working groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change. In: Pachauri, R.K.; Reisinger, A., eds. Climate change 2007: synthesis report. Geneva, Switzerland: 104.

Jackson, C.R.; Martin J.K.; Leigh, D.S.; West, L.T. West. 2005. A southeastern piedmont watershed sediment budget: evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation. 60: 298–310.

Jackson, C.R.; Sun, G.; Amatya, D.M. [and others]. 2004. Fifty years of forest hydrology in the Southeast. In: Ice, G.G.; Stednick, J.D., eds. A century of forest and wildland watershed lessons. Bethesda, MD: Society of American Foresters: 33–112.

Jackson, R.B.; Jobbagy, E.G.; Avissar, R. [and others]. 2005. Trading water for carbon with biological carbon sequestration. Science. 310: 1944–1947. [DOI:10.1126/science.1119282].

Jones, J.A.; Post, D.A. 2004. Seasonal and successional streamflow response to forest cutting and regrowth in the Northwest and Eastern United States. Water Resources Research. 40: W05203. [DOI:10.1029/2003WR002952].

Karl, T.R.; Knight, R.W. 1998. Secular trends of precipitation amount, frequency, and intensity in the USA. Bulletin of the American Meteorology Society. 79: 231–241.

Kenny, J.F.; Barber, N.L.; Hutson, S.S. [and others]. 2009. Estimated use of water in the United States in 2005. Circ. 1344. Reston, VA: U.S. Department of the Interior, U.S. Geological Survey. 52 p.

Klapproth, J.C.; Johnson, J.E. 2000. Understanding the science behind riparian forest buffers: effects on water quality. Virginia Coop. Ext. Publ. Blacksburg, VA: Virginia Tech: 420–451.

Kolpin, D.W.; Furlong, E.T.; Meyer, M.T. [and others]. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999–2000: a national reconnaissance. Environmental Science & Technology. 36: 1202–1211.

Koren, V.; Smith, M.; Duan, Q. 2003. Use of *a priori* parameter estimates in the derivation of spatially consistent parameter sets of rainfall-runoff models. In: Duan, Q.; Sorooshian, S.; Gupta, H. [and others], eds. Calibration of watershed models water science and applications. American Geophysical Union: 239–254. Vol. 6.

Lehrter, J.C. 2006. Effects of land use and land cover, stream discharge, and interannual climate on the magnitude and timing of nitrogen, phosphorus, and organic carbon concentrations in three Coastal Plain watersheds. Water Environment Research. 78(12): 2356–2368.

Lenat, D.R.; Crawford, J.K. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia. 294: 185–199.

Likens, G. E. and F. H. Bormann. 1995. Biogeochemistry of a forested ecosystem, 2nd edition. New York: Springer-Verlag, 159 p.

Lins, H.; Slack, J.R. 1999. Streamflow trends in the United States. Geophysical Research Letters. 26: 227–230.

Lowrance, R.R.; Leonard, R.A.; Asmussen, L.E.; Todd, R.L. 1985. Nutrient budgets for agricultural watersheds in the southeastern Coastal Plain. Ecology. 66(1): 287–296.

Luce, C.H.; Holden, Z.A. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948–2006. Geophysical Research Letters. 36: L16401. [DOI:10.1029/2009GL039407].

Mallin, M.A.; Williams, K.E.; Esham, E.C.; Lowe, R.P. 2000. Effect of human development on bacteriological water quality in coastal watersheds. Ecological Applications. 10: 1047–1056.

Maloney, K.O.; Feminella, J.W. 2006. Evaluation of single- and multi-metric benthic macroinvertebrate indicators of catchment disturbance over time at the Fort Benning Military Installation, Georgia, USA. Ecological Indicators. 6: 469–484.

Martin, W.H.; Boyce, S.G. 1993. Introduction: the southeastern setting. pages 1-46. In: Martin, W.H.; Boyce, S.G.; Echternacht, A.C., eds. New York: John Wiley..

Maurer, E.P.; Brekke, L.; Pruitt, T.; Duffy, P.B. 2007. Fine-resolution climate projections enhance regional climate change impact studies. Eos, Transactions, American Geophysical Union. 88(47): 504.

May, S.K.; Kimball, W.H.; Grady, N.; Dolan, R. 1982. CEIS: the coastal erosion information system. Shore and Beach. 50: 19–26.

McCullen, C.P.; Jabbour, J. 2009. Climate change science compendium. http://www.unep.org/compendium2009/. [Date accessed: October 28, 2010].

McMahon, G.; Bales, J.D.; Coles, J. [and others]. 2003. Use of stage data to characterize hydrologic conditions in an urbanizing environment. Journal of the American Water Resources Association. 39: 1529–1546.

McMahon, G.; Harned, D.A. 1998. Effect of environmental setting on sediment, nitrogen, and phosphorous concentrations in Albemarle-Pamlico Drainage Basin, North Carolina and Virginia, USA. Environmental Management. 22(6): 887–903.

Meyer, J.L. 1992. Seasonal patterns of water quality in Blackwater Rivers of the Coastal Plain, Southeastern United States. In: Becker, C.D.; Neitzel, D.A., eds. Water quality in North American river systems. Columbus, OH: Bartelle Press: 249–276.

Mielke, M.S.; Oliva, M.A.; deBarros, N.F. [and others]. 1999. Stomatal control of transpiration in the canopy of a clonal eucalyptus grandis plantation. Trees - Structure and Function. 13: 152–160.

Mitsch, W.J.; Day, J.W.; Gilliam, J.W. [and others]. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem. BioScience. 51(5): 373–388.

Moore, G.W.; Bond, B.J.; Jones, J.A. [and others]. 2004. Structural and compositional controls on transpiration in 40- and 450-year-old riparian forests in western Oregon, USA. Tree Physiology. 24: 481–491.

Morgan, R.P.; Cushman, S.E. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society. 24: 643–655.

Nagy, R.C., B.G. Lockaby, B. Helms, L. Kalin, and D. Stoeckel. In press. Water resources and land use / cover in a humid region: the southeastern United States. Journal of Environmental Quality.

Nagy, R.C; Lockaby, B.G.; Kalin, L.; Anderson, C. Urbanization of a coastal region and the effects on water resources. Journal of Environmental Quality. R. Chelsea Nagy, Graduate Research Assistant, original manuscript, Brown University, 80 Waterman St., Providence, RI 02912.

Naiman, R.J.; Décamps, H.; McClain, M.E. 2005. Riparia: ecology, conservation, and management of streamside communities. Burlington, MA: Elsevier Academic Press. 430 p.

National Research Council. 1995. Wetlands: characteristics and boundaries. Washington, DC: National Academy Press. 328 p.

Novick, K.; Oren, R.; Stoy, P. [and others]. 2009. The relationship between reference canopy conductance and simplified hydraulic architecture. Advances in Water Resources. 32: 809–819.

Novotny, V. 2003. Water quality: diffuse pollution and watershed management. New York: John Wiley. 888 p.

NPA Data Services, Inc. 1999. Economic databases- mid-range growth projections 1967–2050. Reg. Econ. Proj. Ser. Arlington, VA.

NRCS. 2009. Watershed Boundary Dataset. Available URL: "http://datagateway.nrcs.usda.gov" [Accessed 01/09/2009].

O'Gorman, P.A.; Schneider, T. 2009. The physical basis for increases in precipitation extremes in simulations of 21st-century climate change. Proceedings of the National Academy of Sciences. 106: 14,773–14,777. [DOI:10.1073/pnas.0907610106].

Oki, T.; Kanae, S. 2006. Global hydrological cycles and world water resources. Science. 313: 1068–1072. [DOI:10.1126/science.1128845].

Okun, D. A. 1992. Properties of the Table Rock and Poinsett Reservoirs: their future. A report to the Greenville Watersheds Study Committee, P.O. Box 728, Greenville, SC. 24 p.

Olivera, F.; DeFee, B.B. 2007. Urbanization and its effect on runoff in the Whiteoak Bayou watershed, Texas. Journal of the American Water Resources Association. 43: 170–182.

Onorato, D.; Marion, K.R.; Angus, R.A. 1998. Longitudinal variations in the ichthyofaunal assemblages of the upper Cahaba River: possible effects of urbanization in a watershed. Journal of Freshwater Ecology. 13: 139–154.

Oren, R.; Pataki, D.E. 2001. Transpiration in response to variation in microclimate and soil moisture in southeastern deciduous forests. Oecologia. 127: 549–559.

Orser, P. N. and D. J. Shure. 1972. Effects of urbanization on the salamander Desmognathus fuscus fuscus. Ecology 53: 1148-1154.

Palmer, T. 1994. Lifelines: the case for river conservation. Washington, DC: Island Press. 256 p.

Parry, M.L.; Canziani, O.F.; Palutikof, J.P. [and others], eds. 2007. Contribution of the working group II to the fourth assessment report of the intergovernmental panel on climate change, 2007. Cambridge, United Kingdom; New York: Cambridge University Press.

Paul, M.J.; Meyer, J.L. 2001. Streams in the urban landscape. Annual Review of Ecology, Evolution, and Systematics. 32: 333–365.

Phillips, N.; Oren, R. 2001. Intra- and inter-annual variation in transpiration of a pine forest. Ecological Applications. 11: 385–396.

Powell, T.L.; Starr, G.; Clark, K.L. [and others]. 2005. Ecosystem and understory water and energy exchange for a mature, naturally regenerated pine flatwoods forest in north Florida. Canadian Journal of Forest Research. 35: 1568–1580.

Price, S.J.; Dorcas, M.E.; Gallant, A.L. [and others]. 2006. Three decades of urbanization: estimating the impact of land-cover change on stream salamander populations. Biological Conservation. 133: 436–441.

Rahmstorf, S. 2007. A semi-empirical approach to projecting future sea-level rise. Science. 315: 368–370.

Rose, S.; Peters, N.E. 2001. Effects of urbanization on streamflow in the Atlanta area (Georgia, USA): a comparative hydrological approach. Hydrological Processes. 15: 1441–1457.

Ross, M.S.; Obrien, J.J.; Sternberg, L.D.L. 1994. Sea-level rise and the reduction in pine forests in the Florida Keys. Ecological Applications. 4: 144–156.

Roy, A.H.; Freeman, M.C.; Freeman, B.J. [and others]. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society. 24: 656–678.

Roy, A.H.; Rosemond, A.D.; Paul, M.J. [and others]. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). Freshwater Biology. 48: 329–346.

Schafer, K.V.R.; Oren, R.; Lai, C.; Katul, G. 2002. Hydrologic balance in an intact temperate ecosystem under ambient and elevated atmospheric CO_2 concentration. Global Change Biology. 8: 895–911.

Schafer, K.V.R.; Oren, R.; Tenhunen, J.D. 2000. The effect of tree height on crown level stomatal conductance. Plant Cell and Environment. 23: 365–375.

Schoonover, J.E.; Lockaby, B.G. 2006. Land cover impacts on stream nutrients and fecal coliform in the lower Piedmont of west Georgia. Journal of Hydrology. 331: 371–382.

Schoonover, J.E.; Lockaby, B.G.; Helms, B.S. 2006. Impacts of land cover on stream hydrology in the west Georgia Piedmont, USA. Journal of Environmental Quality. 35: 2123–2131.

Schoonover, J.E.; Lockaby, B.G.; Pan, S. 2005. Changes in chemical and physical properties of stream water across an urban-rural gradient in western Georgia. Urban Ecosystems. 8: 107–124.

Scott, M.C.; Helfman, G.S.; McTammany, M.E. [and others]. 2002. Multiscale influences on physical and chemical stream conditions across Blue Ridge landscapes. Journal of the American Water Resources Association. 38(5): 1379–1392.

Solley, W.B.; Pierce, R.R.; Perlman, H.A. 1998. Estimated use of water in 1995. Circ. 1200. Alexandria, VA: U.S. Geological Survey. http://water.usgs.gov/watuse/pdf1995/pdf/summary.pdf. [Date accessed: July 2007].

Soloman, S.; Plattner, G-k.; Knutt, R.; Friedlingstein, P. 2009. Irreversible climate change due to carbon dioxide emissions. Proceedings of the National Academy of Sciences of the USA. 106(6): 1704–1709.

Sponseller, R.A.; Benfield, E.F.; Valett, H.M. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology. 46: 1409–1424.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology. 176: 79–95.

Stoy, P.; Katul, G.; Siqueira, M. [and others]. 2006. Separating the effects of climate and vegetation on evapotranspiration along a successional chronosequence in the Southeastern US. Global Change Biology. 12: 2115–2135.

Sun, G., M. Riedel, R. Jackson, R. Kolka, D. Amatya, and J. Shepard. 2004. Book Chapter 3: Influences of management of Southern forests on water quantity and quality. In: H.M. Rauscher an d K. Johnsen (Eds.) Southern Forest Sciences: Past, Current, and Future. Gen. Tech. Rep/ SRS-75. Ashville, NC U.S. Department of Agriculture, Forest Service, Southern Research Station. 394 p.

Sun, G.; McNulty, S.G.; Lu, J. [and others]. 2005. Regional annual water yield from forest lands and its response to potential deforestation across the Southeastern United States. Journal of Hydrology. 308: 258–268.

Sun, G.; McNulty, S.G.; Moore Myers, J.; Cohen, E.C. 2008. Impacts of multiple stresses on water demand and supply across the Southeastern United States. Journal of American Water Resources Association. 44(6): 1441–1457.

Sun, G., A. Noormets, M.J. Gavazzi1, S.G. McNulty, J. Chen, J-C. Domec, J.S. King, D.M. Amatya, and R.W. Skaggs. 2010. Energy and water Balance of two contrasting Loblolly pine plantations on the lower coastal plain of North Carolina, USA. Forest Ecology and Management 259: 1299-1310.

Sun, G.; Alstad, K.; Chen, J. [and others]. 2011. A general predictive model for estimating monthly ecosystem evapotranspiration. Ecohydrology.

Sutherland, A.B.; Meyer, J.L.; Gardiner, E.P. 2002. Effects of land cover on sediment regime and fish assemblage structure in four Southern Appalachian streams. Freshwater Biology. 47: 1791–1805.

Swank, W.T.; Douglass, J.E. 1974. Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science. 185: 857–859.

Swank, W.T.; Douglass, J.E. 1977. Nutrient budgets for undisturbed and manipulated hardwood forest ecosystems in the mountains of North Carolina. In: Correll, D.L., ed. Watershed research in Eastern North America. Edgewater, MD: Smithsonian Institute: 343-362. Vol. 1.

Swift, L.W., Jr. 1988. Forest access roads: design, maintenance, and soil loss. In: Swank, W.T.; Crossley, D.A., Jr., eds. Ecological studies, forest hydrology and ecology at Coweeta. New York: Springer Verlag: 313–324. Vol. 66.

Thieler, E.R.; Hammar-Klose, E.S. 2000. National assessment of coastal vulnerability to future sea-level rise: preliminary results for the U.S. Gulf of Mexico Coast. U.S. Geological Survey, Open-File Report 00-179, 1 sheet.

Thomson, A.M.; Brown, R.A.; Rosenberg, N.J. [and others]. 2003. Simulated impacts of El Niño/southern oscillation on United States water resources. Journal of the American Water Resources Association. 39: 137–148. [d65-10.1111/j.1752-1688.2003.tb01567.x].

Titus, J.G.; Richman, C. 2001. Maps of lands vulnerable to sea level rise: modeled elevations along the US Atlantic and gulf coasts. Climate Research. 18(2): 205–228.

Tomer, M.D.; Dosskey, M.G.; Burkart, M.R. [and others]. 2005. Placement of riparian forest buffers to improve water quality. In: Brooks, K.N.; Folliot, P.F., eds. Moving agroforestry into the mainstream. Proceedings of the 9th North America agroforestry conference. St. Paul, MN: University of Minnesota, Department of Forest Resources: 11 p.

Trimble, S.W. 2008. Man-induced soil erosion on the southern Piedmont. Soil and Water Conservation Society. 80 p.

Tufford, D.L.; Samarghitan, C.L.; McKellar, H.N., Jr. [and others]. 2003. Impacts of urbanization on nutrient concentrations in small southeastern coastal stream. Journal of the American Water Resources Association. 39(2): 301–312.

Ursic, S.J. 1986. Sediment and forestry practices in the South. In: Proceedings, fourth Federal interagency sedimentation conference. Washington, DC: U.S. Government Printing Office: 28–37. Vol. 1.

U.S. Census. 2001. Census 2000 summary file 1 United States, product ID V1–D00–S1S1–08–US1. http://www.census.gov/geo/www/gazetteer/places2k.html. [Date accessed: October 28, 2010].

U.S. Department of Agriculture, National Agroforestry Center. 2008. Working trees for water quality. http://www.unl.edu/nac/workingtrees.htm. [Date accessed: June 18, 2010].

U.S. Department of Agriculture, Natural Resource Conservation Service. 2009. Distribution maps of dominant soil orders. http://soils.usda.gov/technical/classification/orders/ultisols.html. [Date accessed: June 15, 2009].

Utz, R.M.; Hilderbrand, R.H.; Boward, D.M. 2009. Identifying regional differences in threshold responses of aquatic invertebrates to land cover gradients. Ecological Indicators. 9: 556–567.

Vellidis, G. 1999. Water quality function of riparian ecosystems in Georgia. In: Hatcher, Kathryn, J., ed. Proceedings of the Georgia water resources conference. Athens, GA: The University of Georgia, Institute of Ecology: 207-210.

Vicars-Groening, J.; Williams, H.F.L. 2007. Impact of urbanization on storm response of White Rock Creek, Dallas, TX. Environmental Geology. 51: 1263–1269.

Vose, J.M.; Allen, H.L. 1988. Leaf area, stemwood growth, and nutrition relationships in loblolly pine. Forest Science. 34(3): 547–563.

Vose, J.M.; Sun, G.; Bredemeier, M. 2011. Forest ecohydrological research in the 21st century: what are the critical needs? Ecohydrology (n press)

Wahl, M.H.; McKellar, H.N.; Williams, T.M. 1997. Patterns of nutrient loading in forested and urbanized coastal streams. Journal of Experimental Marine Biology and Ecology. 213: 111–131.

Walbridge, M.R. 1993. Functions and values of forested wetlands in the Southern United States. Journal of Forestry. 91(5): 15–19.

Wallace, J.B.; Webster, J.R.; Lowe, R.L. 1992. High-gradient streams of the Appalachians. In: Hackney, C.T.; Adams, S.M.; Martin, W.A., eds. Biodiversity of the Southeastern United States: aquatic communities. New York: John Wiley: 133-192.

Walsh, C. J.; Roy, A.H.; Feminella, J.W. [and others]. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society. 24: 706–723.

Walters, D.M.; Leigh, D.S.; Bearden, A.B. 2003. Urbanization, sedimentation, and the homogenization of fish assemblages in the Etowah River basin, USA. Hydrobiologia. 494: 5–10.

Wang, L.; Lyons, J.; Kanehl, P.; Bannerman, R. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environmental Management. 28: 255–266.

Weaver, L.A.; Garman, G.C. 1994. Urbanization of a watershed and historical changes in a stream fish assemblage. Transactions of the American Fisheries Society. 123: 162–172.

Weston, N.B.; Hollibaugh, J.T.; Joyce, S.B. 2009. Population growth away from the coastal zone: thirty years of land use change and nutrient export in the Altamaha River, GA. Science of the Total Environment. 407(10): 3347–3356.

World Wildlife Fund. 2010. U.S. Southeast rivers and streams: safeguarding America's richest source of freshwater. http://www.worldwildlife.org/what/wherewework/sers/index.html. [Date accessed: June 18, 2010].

Wullschleger, S.D.; Hanson, P.J.; Todd, D.E. 2001. Transpiration from a multi-species deciduous forest as estimated by xylem sap flow techniques. Forest Ecology and Management. 143: 205–213.

Tables

Table 13-1–Total impervious area for eight southern States reported by Exum and others (2005)

	Percent land area classified as impervious				
	>20	10-20	5-10	2-5	<2
Alabama	0.8	1.9	6.0	29.6	61.8
Florida	7.0	7.2	12.1	23.7	50.1
Georgia	2.4	4.0	7.9	30.1	55.7
Kentucky	0.8	2.2	7.0	40.2	49.8
Mississippi	0.1	1.2	2.9	20.5	75.2
North Carolina	1.9	5.2	11.0	38.9	43.1
South Carolina	1.7	4.0	10.1	37.7	46.7
Tennessee	1.9	3.1	6.9	37.7	50.4

Table 13-2–Scenarios for water supply stress index simulations (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand), based on inputs from historic and projected estimations of population, land use, and climate

Scenerio name	**Annual WaSSI averaging period**	**Population scenario**	**Land use input**[2]	**Climate Input**
Historic climate, land use, and population	1995 to 2005	2000	1997	NCAR[3] 1960 to 2007
Historic climate and land use; future population	1995 to 2005	2050	1997	NCAR 1960 to 2007
Historic climate and population; future land use	1995 to 2005	2000	2050	NCAR 196 to 2007
Historic climate; future population and land use	1995 to 2005	2050	2050	NCAR 1960 to 2007
Csiromk35a1b climate; historic population and land use	2045 to 2055	2000	1997	csiromk35a1b
Miroc32a1b climate; historic population and land use	2045 to 2055	2000	1997	miroc32a1b
Csiromk2b2 climate; historic population and land use	2045 to 2055	2000	1997	csiromk2b2
Hadcm3b2 climate; historic population and land use	2045 to 2055	2000	1997	hadcm3b2
Csiromk35a1b climate; future population and land use	2045 to 2055	2050	2050	csiromk35a1b
Miroc32a1b climate; future population and land use	2045 to 2055	2050	2050	miroc32a1b
Csiromk2b2 climate; future population and land use	2045 to 2055	2050	2050	csiromk2b2
Hadcm3b2 climate; future population and land use	2045 to 2055	2050	2050	hadcm3b2

[2] See chapter 4.

[3] National Center for Atmospheric Research in Boulder, CO.

Table 13-3–Southern wetland types and characteristics

Wetland type	Characteristics	Sources
Riverine (bottomland hardwood wetlands)	Occurring in floodplains or riparian corridors Many connections between wetland and stream channel (overbank flow and subsurface connections) Including cypress stands, sloughs, and hardwood swamps associated with brown-, black-, and redwater streams in Atlantic and gulf Coastal Plains	Ainslie 2002, Brinson 1993, Palmer 1994, Childers and Gosselink 1990, National Research Council 1995, Naiman and others 2005, Walbridge 1993, Meyer 1992, Dennis 1988
Depressional	Named for their depressional topography which promotes surface water accumulation Cypress trees predominant Including cypress domes (gulf Coastal Plain) and Carolina bays (Atlantic Coastal Plain)	Ainslie 2002, Brinson 1993, Duryea and Hermansen 1997, Dennis 1988
Wet flats	Can have either organic soils (pocosins) on plateaus or mineral soils in the Atlantic Coastal Plain between rivers or floodplain terraces Pocosins characterized by dense evergreen shrub vegetation Mineral flats characterized by a closed canopy of hardwoods or an open savannah with some pines	Ainslie 2002, Brinson 1993, Gresham 1989

Table 13-4–Mean annual transpiration (mm yr^{-1}) for southern forest types

Vegetation type	Transpiration	Source
Longleaf pine savanna	244	(Ford and others 2008)
Old field	250	(Stoy and others 2006)
Oak-pine-hickory forest	278	(Oren and Pataki 2001)
Upland oak forest	313	(Wullschleger and others 2001)
Mixed pine hardwood	355	(Phillips and Oren 2001)
Mixed pine hardwood	442	(Stoy and others 2006)
Planted loblolly pine	490	(Stoy and others 2006)
Mixed pine hardwood	523	(Schafer and others 2002)
Slash pine flatwoods	563	(Powell and others 2005)
Eucalyptus hybrid planatation	882	Estimated for Baker County, southwester n Georgia in 2006 for an average climate and rainfall year1
Planted loblolly pine (early rotation)	328	Domec et al. unpublished data; USFS Raleigh, NC 27606. Sun et al., 2010
Planted loblolly pine (mid-rotation)	777	Domec et al. unpublished data; USFS Raleigh, NC 27606; Sun et al., 2010

1 Derived from a model that used data collected in 2006 by the Joseph W. Jones Ecological Research Center, 3988 Jones Center Drive Newton, GA 39870; model assumed no soil water limitation, all trees at age 5, 1,111 trees ha-1, and a leaf area index of 6 m2 m-2 (Mielke and others 1999).

[Type text]

Table 13-5–Sea-level rise estimates by the end of 21st century

Author	Estimated rise	Model characteristics
Parry 2007	0.28 m to 0.43 m	Excludes dynamic ice changes
Rahsmorf 2007	0.5 m to 1.4 m	Semi-empirical (relationship: sea-level rise and surface temperature)
Soloman and others 2009	0.4 m to 1.9 m	Limited to oceanic thermal expansion
	Could increase above estimate by several meters	Includes glacier melts and ice sheet melts
McCullen and Jabbour 2009	0.8 m to 2.0 m	Includes ice changes

Figures

Figure 13-1—Key controls of on water supply and demand and their interactions (Sun et al., 2008).

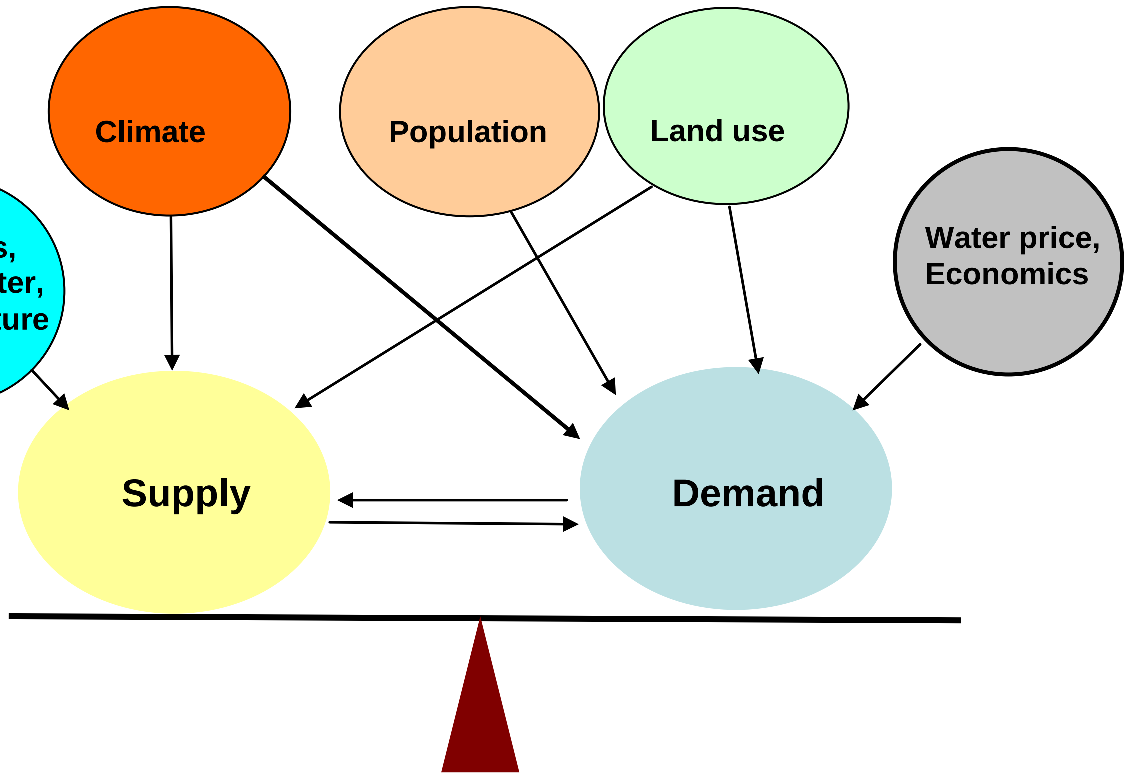

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Figure 13-2–Representative hydrograph of a forested watershed (Crim 2007). Discharge units are liters per second per hectare.

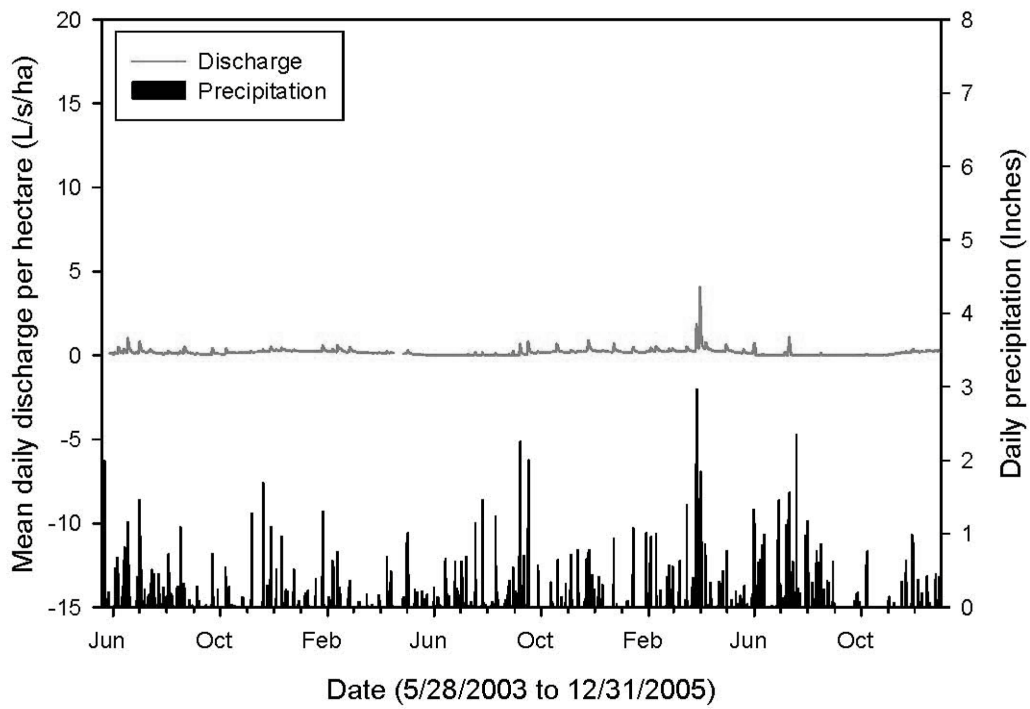

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Figure 13-3–Representative hydrograph of a pastoral watershed (Crim 2007). Discharge units are liters per second per hectare.

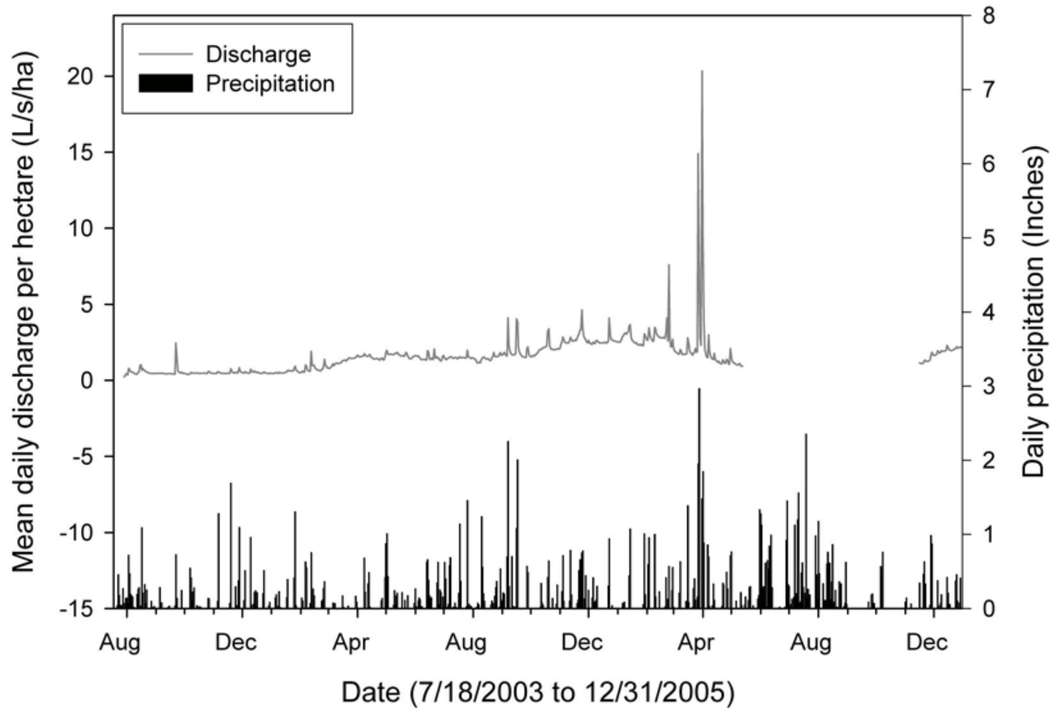

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Figure 13-4–Representative hydrograph of an urban watershed (Crim 2007). Discharge units are liters per second per hectare.

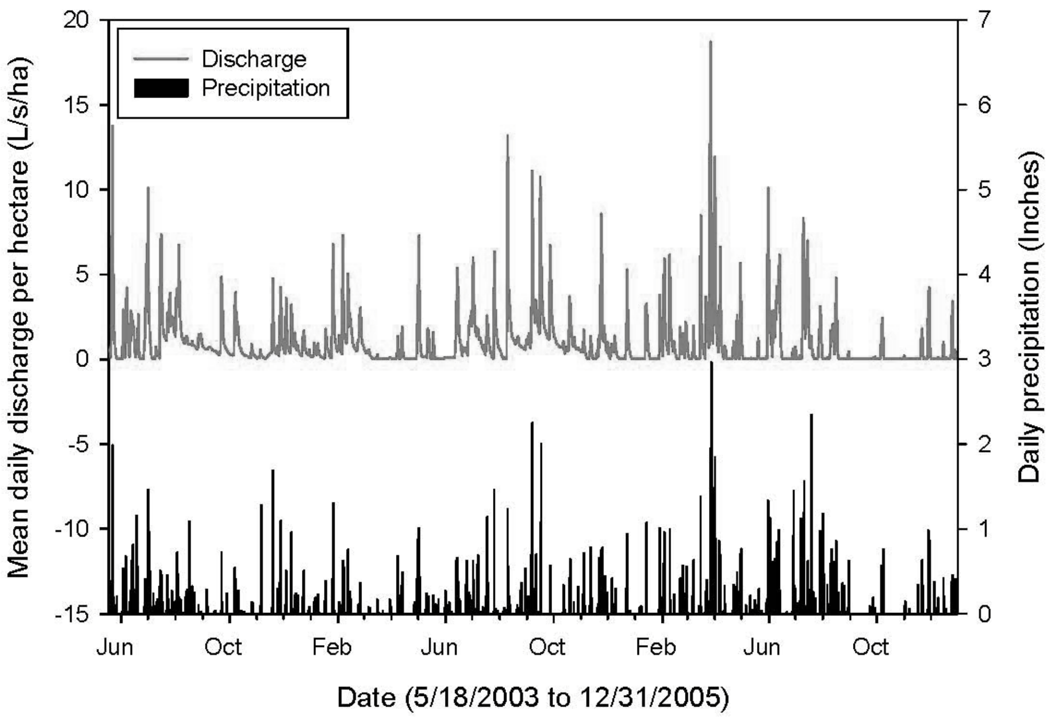

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Figure 13-5—Stream flow-duration curves in cubic feet per second before and after urbanization (L. Kalin, unpublished data; Auburn University, School of Forestry and Wildlife Sciences, 602 Duncan Dr., Auburn, Alabama, 36849; latif@auburn.edu)

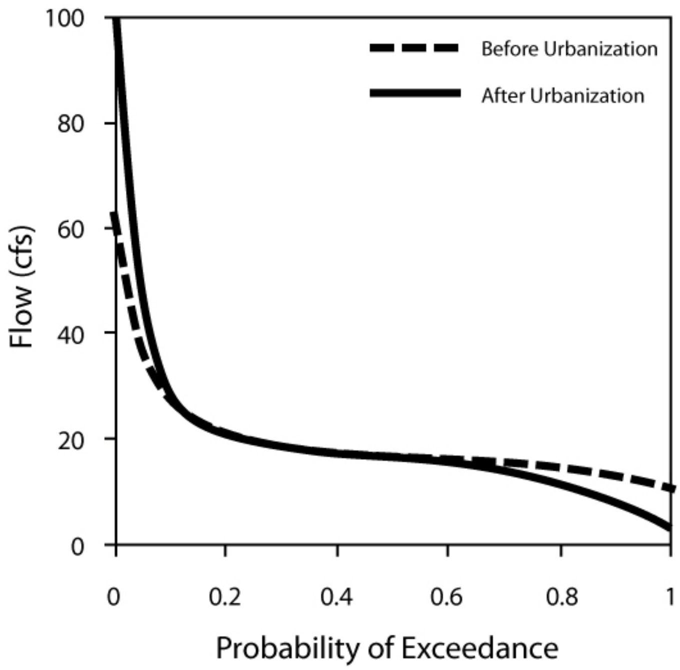

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Figure 13-6–Projected increases in urban cover within three major river basins of the South from 1997 to 2060.

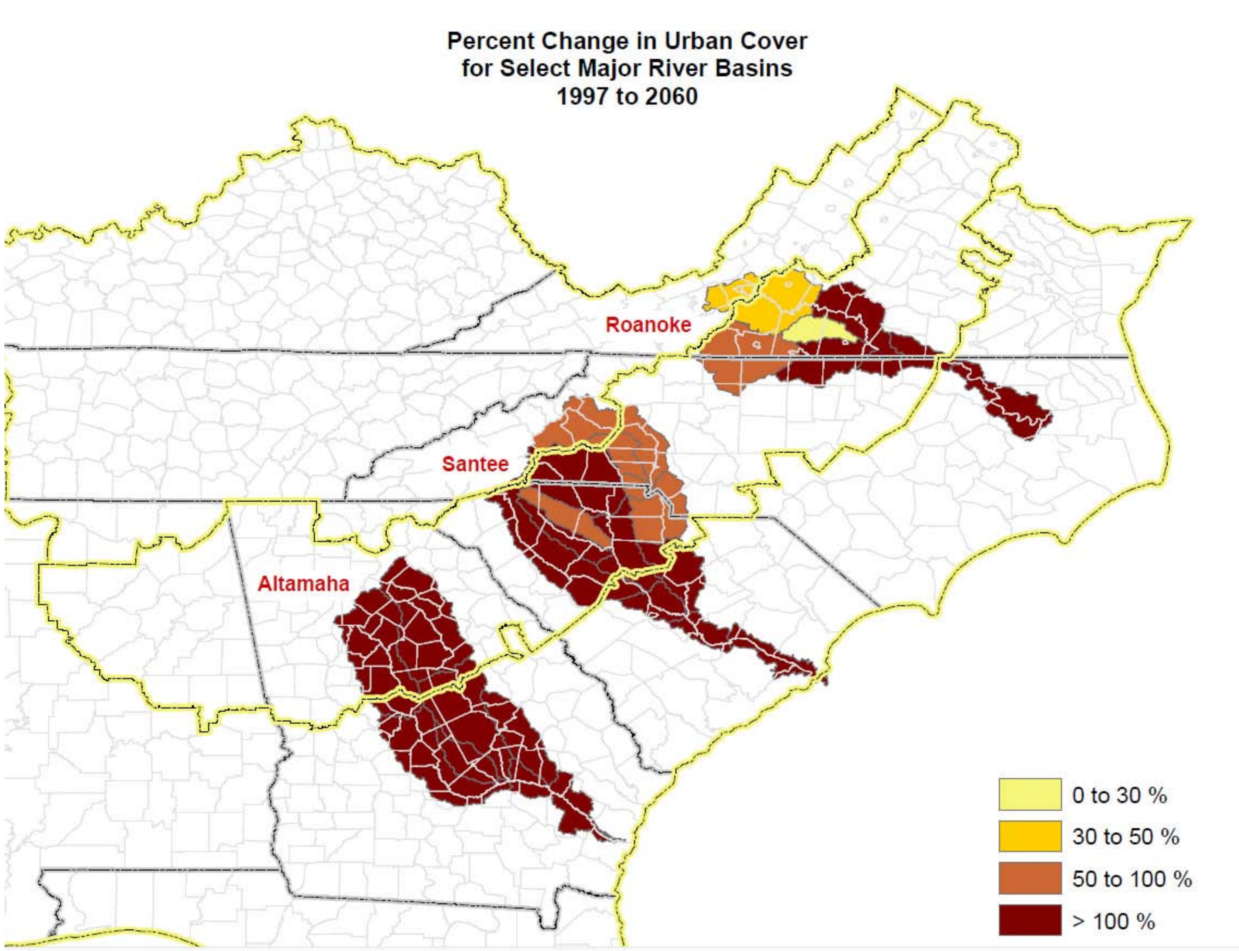

Figure 13-7– (A) 2001 MODIS percent land cover and (B) simulated mean water yield by land cover in the South from 2002 to 2007, showing that forests cover 27 percent of the land area but produce 34 percent of total water yield in the region.

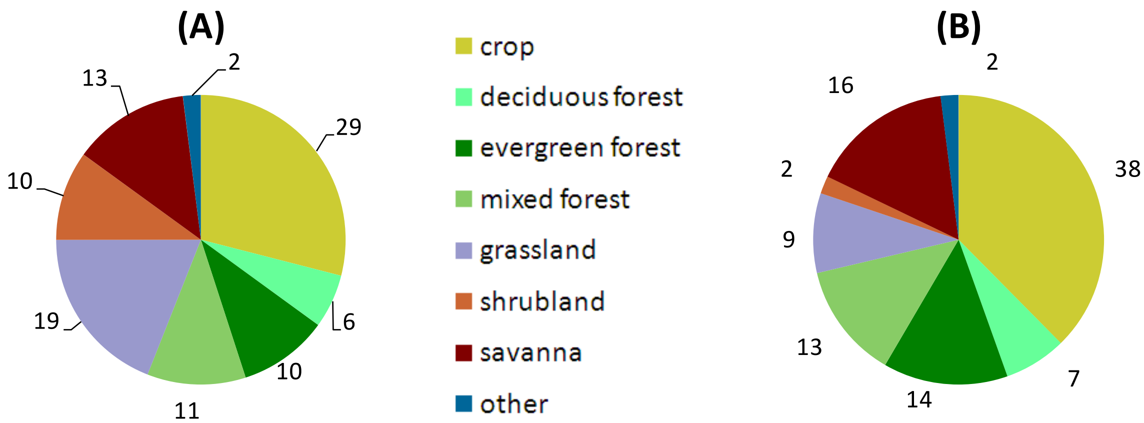

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Figure 13-8–Water supply stress index (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) under baseline, 1995 to 2005, conditions.

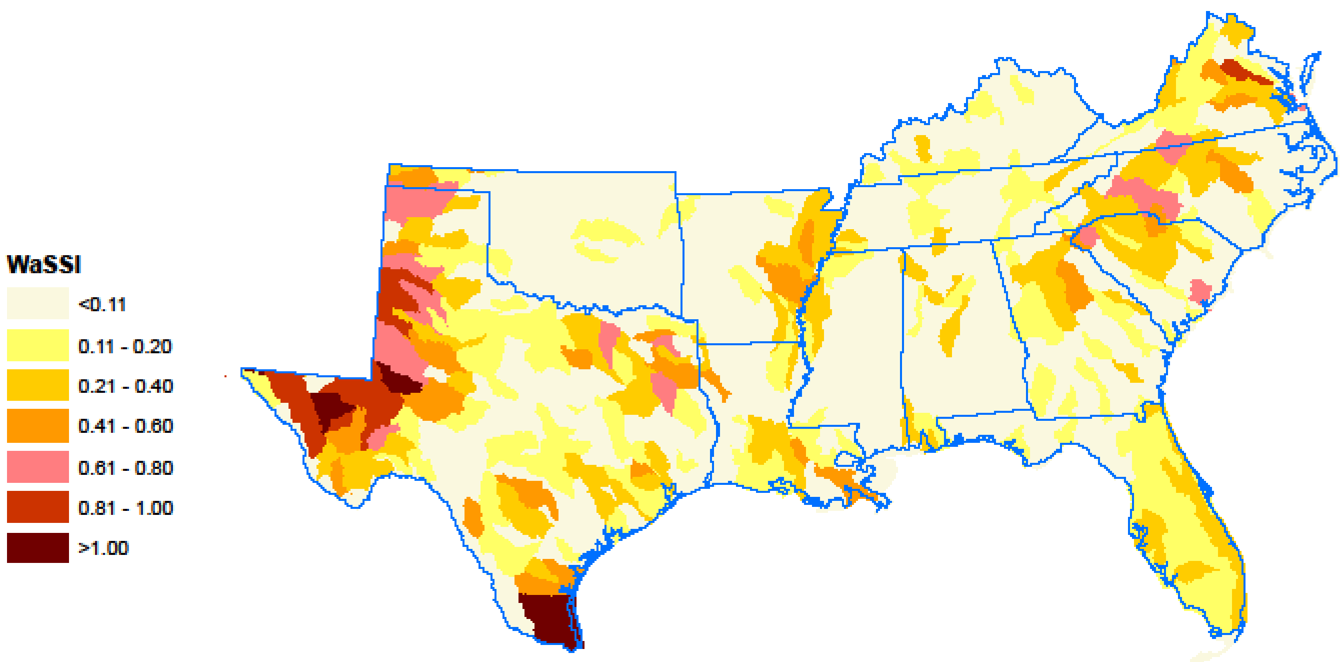

Figure 13-9–Average monthly water supply stress (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) among all Natural Resource Conservation Service Watershed Boundary Dataset Hydrologic Unit Code watersheds (HUCs) in the South under historic and four future climate scenarios.

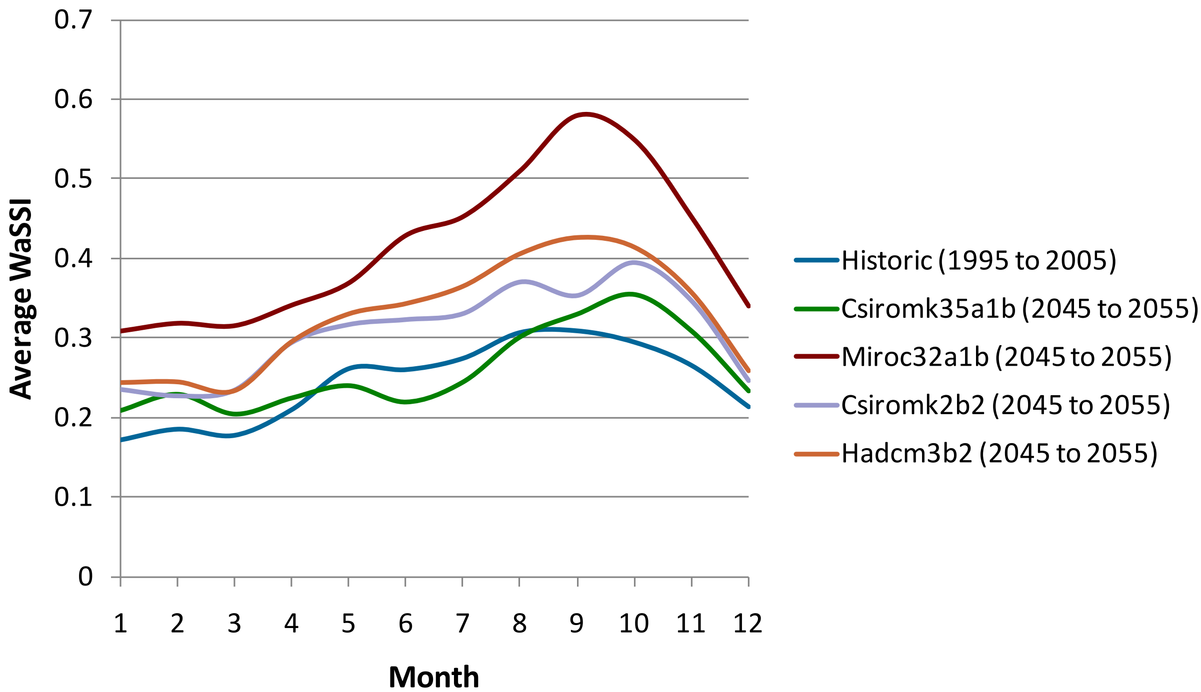

Figure 13-10–Percent change in water supply stress (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) caused by population change by 2050.

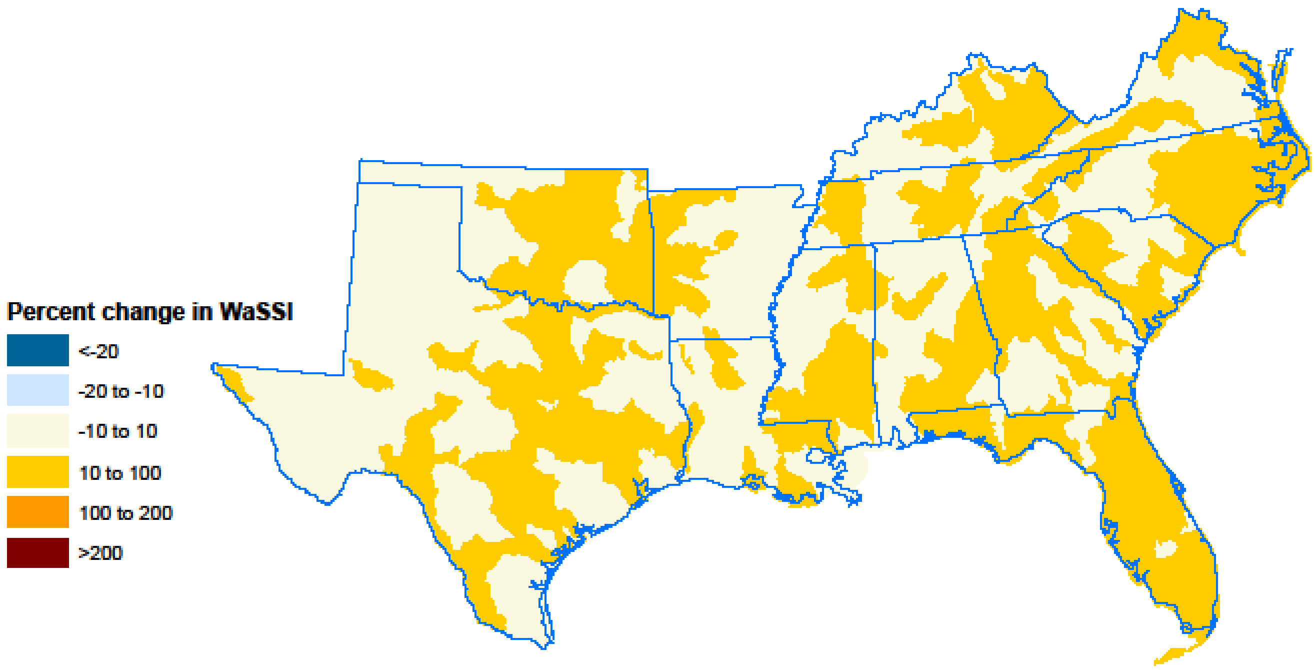

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Figure 13-11–Percent change in water supply stress index (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) due to land use change by 2050.

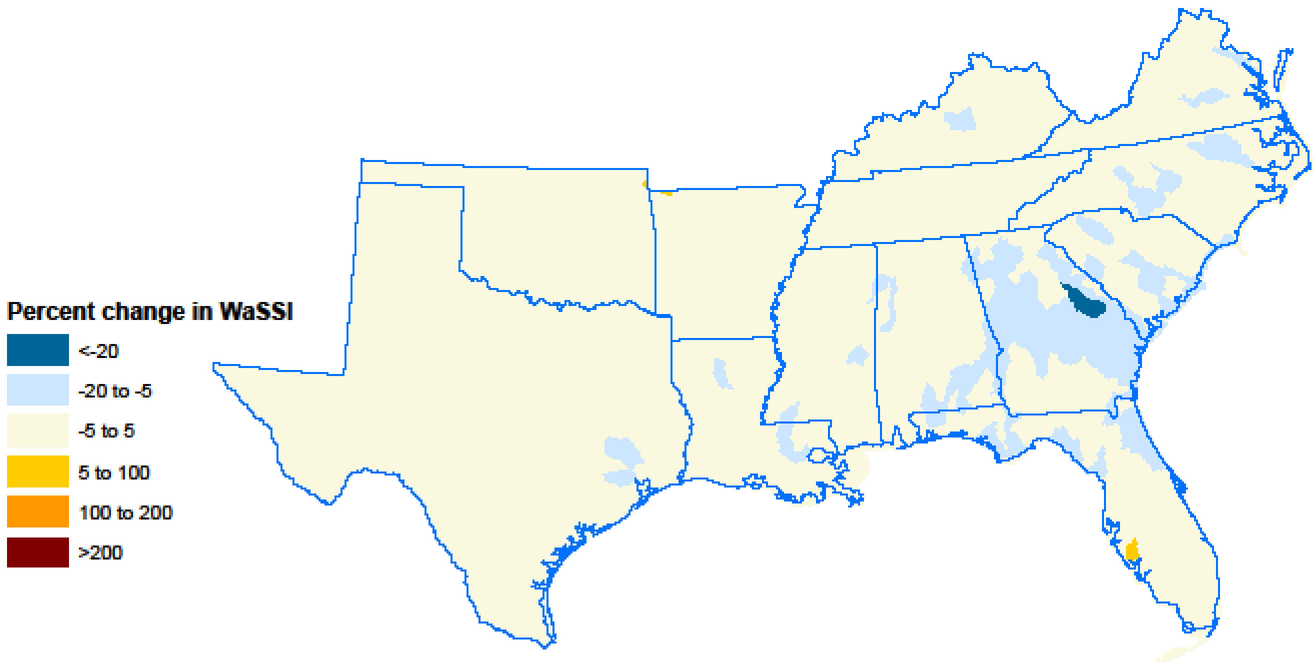

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Figure 13-12–Change in water supply stress (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) by 2050 due to the combined effects of population and land use change.

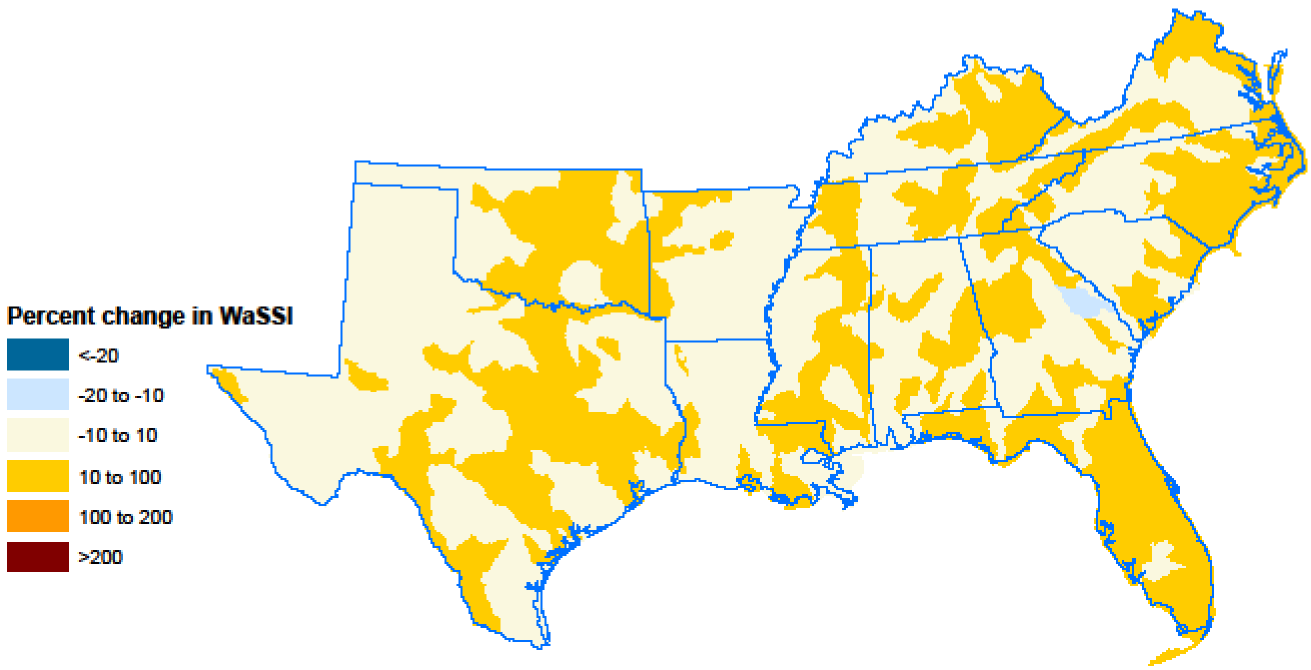

Figure 13-13–Projected average river flows among the 674 Natural Resources Conservation Service Watershed Boundary Dataset 8-digit Hydrologic Unit Code watersheds (HUCs) in the South under four future climate scenarios.

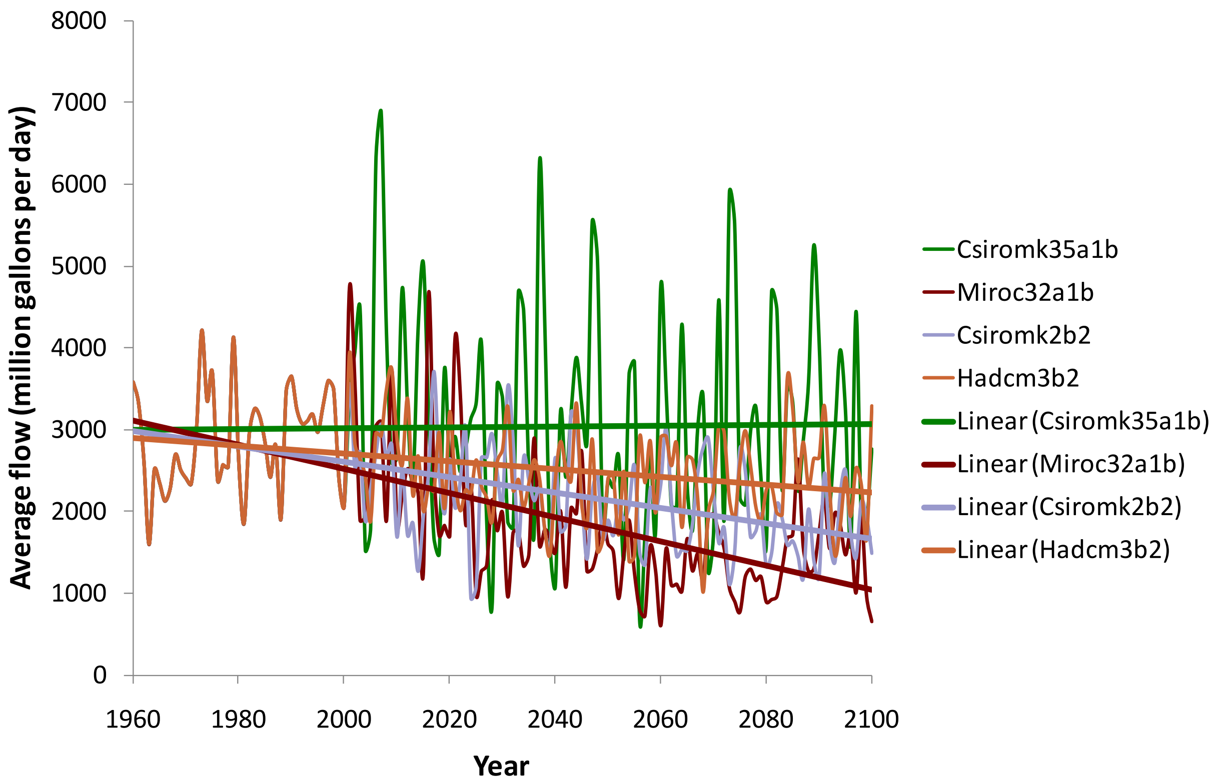

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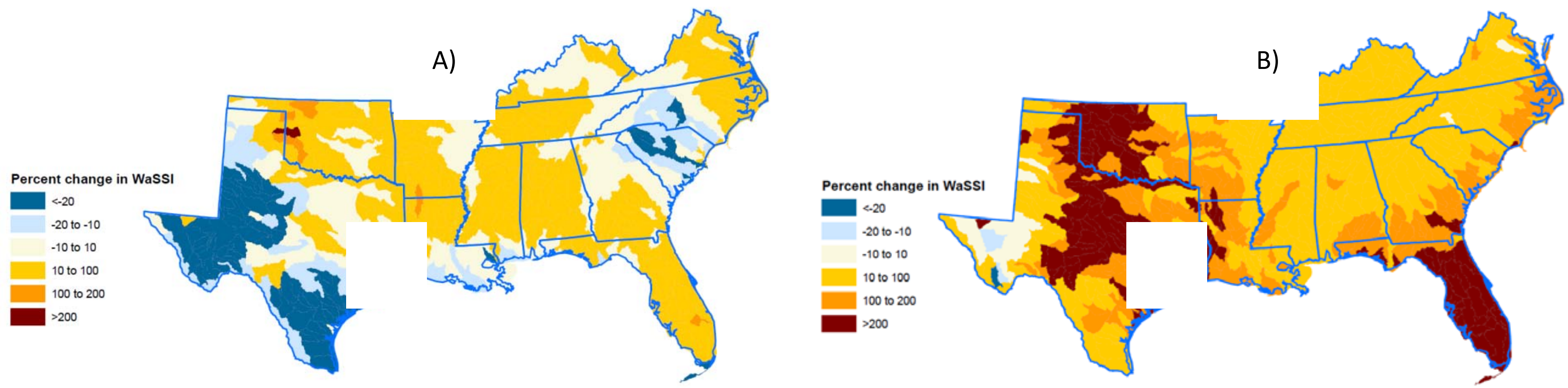
Figure 13-14–Percent change in water supply stress due to climate change (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) by 2050 under four climate scenarios: (A) csiromk35a1b, (B) miroc32a1b, (C) csiromk2b2, and (D) hadcm3b2 .



C)

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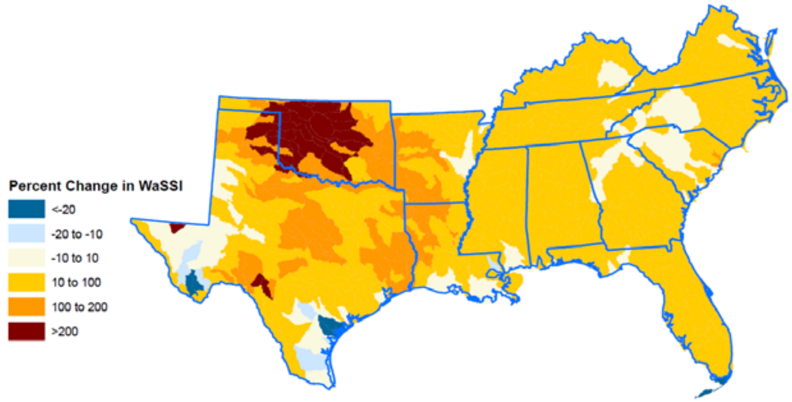
Percent Change in WaSSI
<-20
-20 to -10
-10 to 10
10 to 100
100 to 200
>200

D)

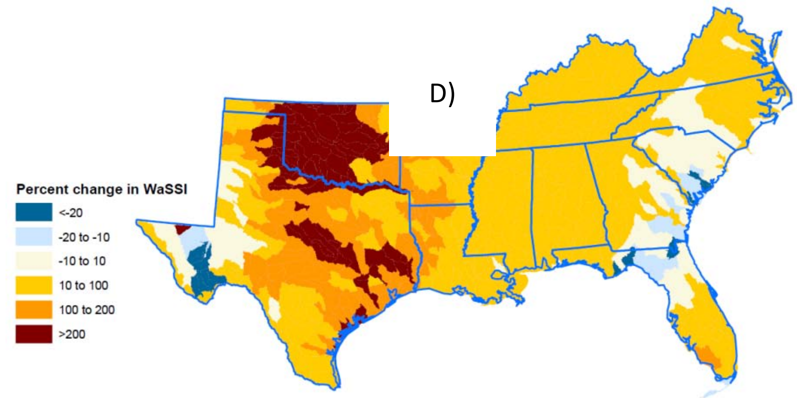
Percent change in WaSSI
<-20
-20 to -10
-10 to 10
10 to 100
100 to 200
>200

Figure 13-15–Percent change in water supply stress (defined by the Water Supply Stress Index or WaSSI and calculated by dividing water supply into water demand) for Natural Resources Conservation Service Watershed Boundary Dataset 8-digit Hydrologic Unit Code watersheds (HUCs) containing State capital cities across the South under four future climate change scenarios between the baseline period (1995 to 2005) and the future condition (2045 to 2055).

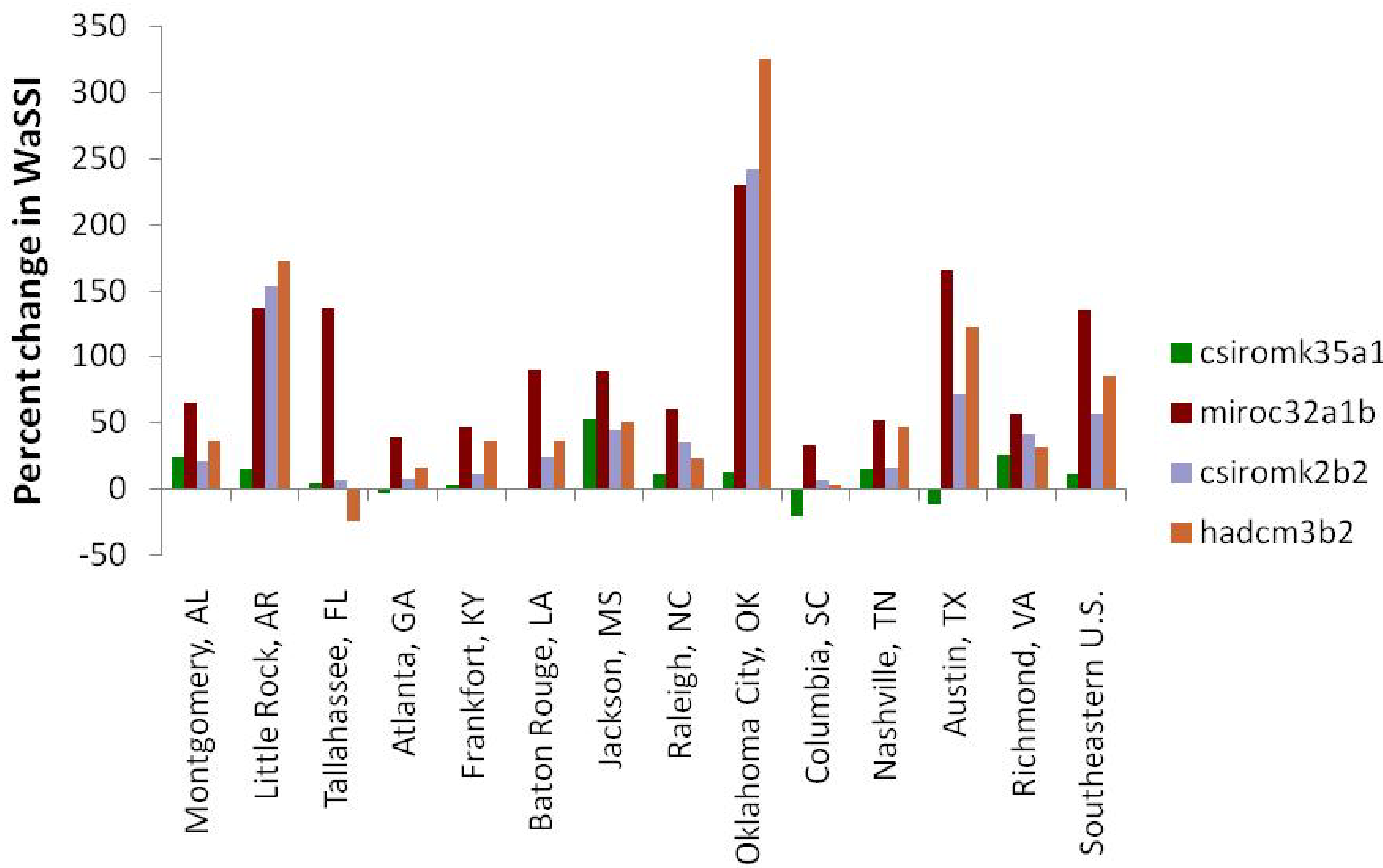

Figure 13-16—Land vulnerable to sea level rise along the Atlantic Ocean and Gulf of Mexico (Titus and Richmond 2001).

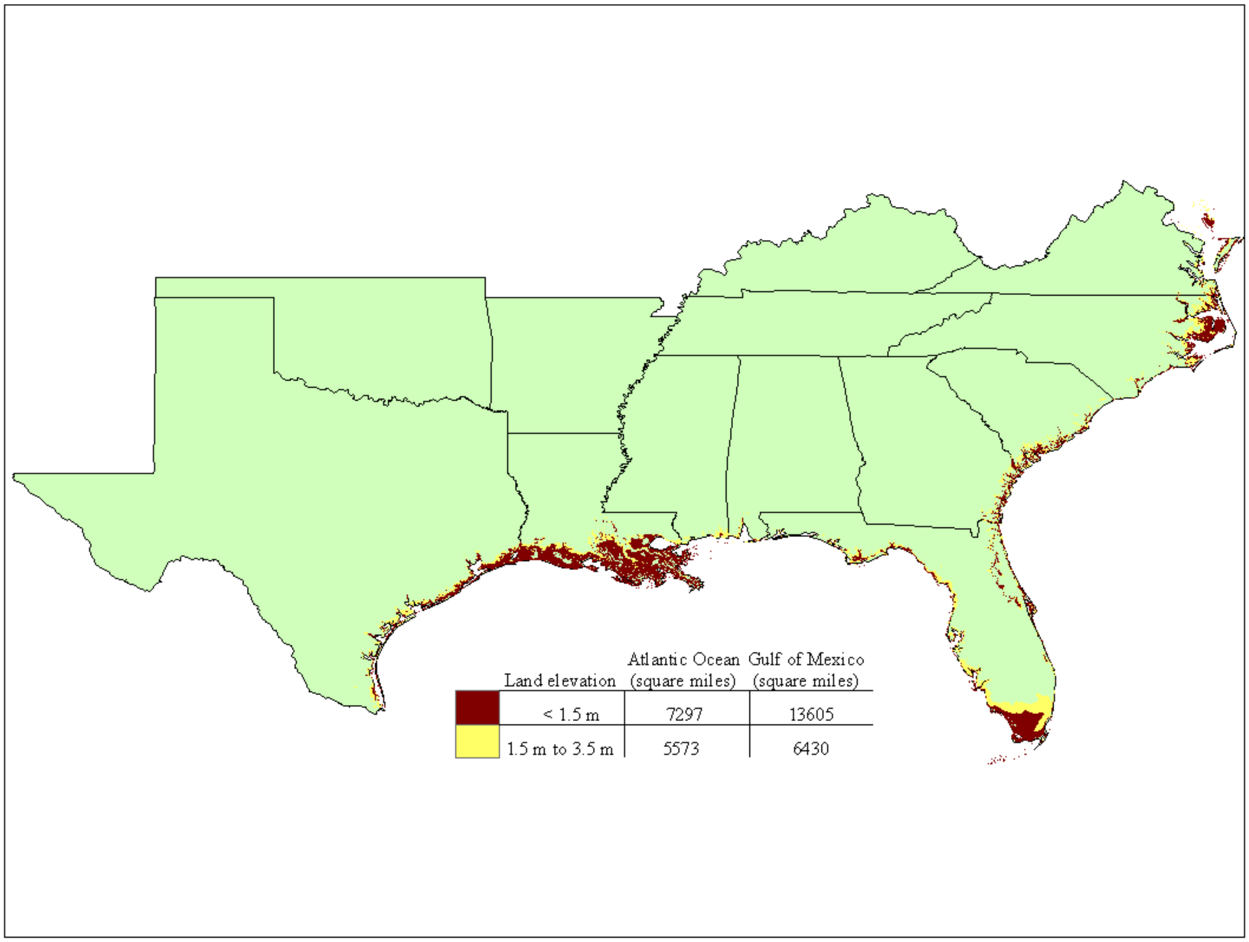

Figure 13-17—Coastal vulnerability to sea level rise along the Atlantic Ocean and Gulf of Mexico (Hammar-Klose and Thieler 2001).

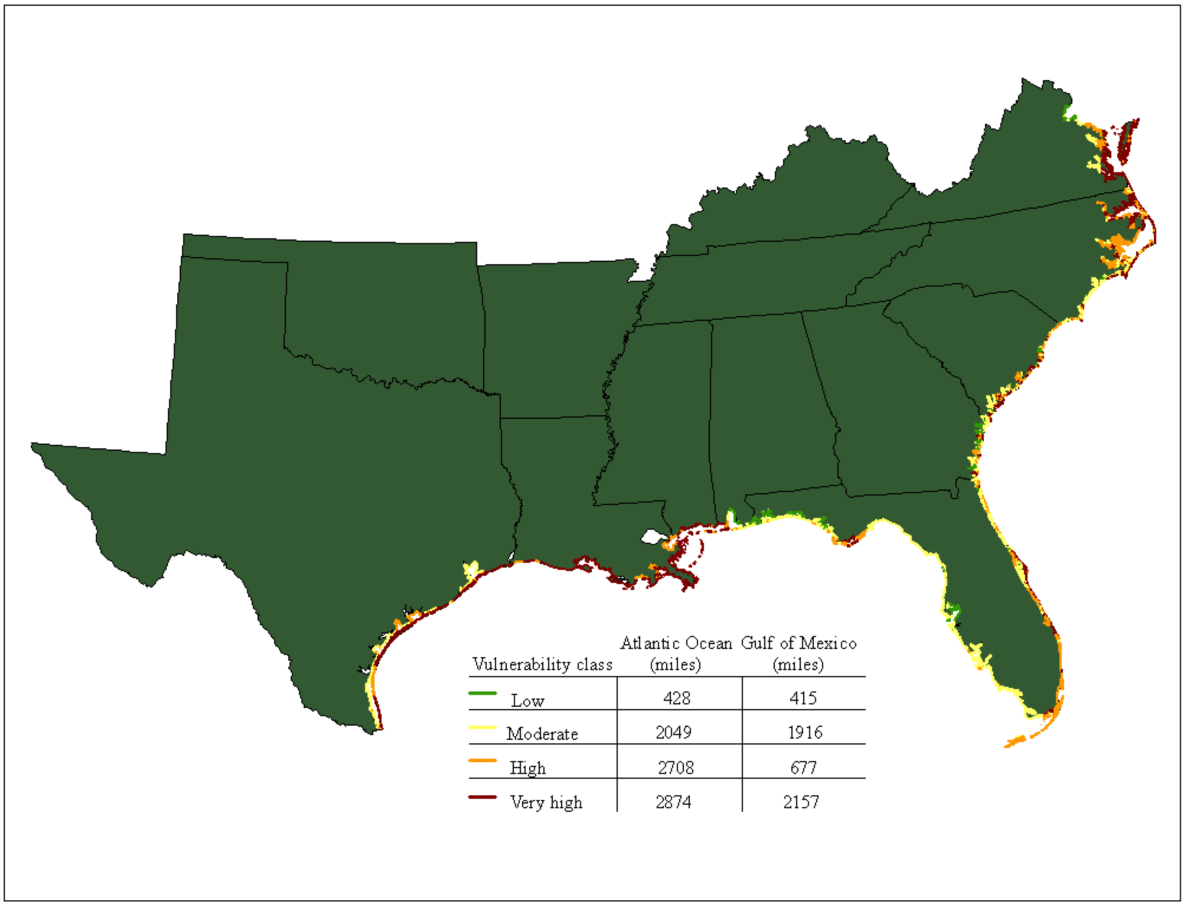

Figure 13-18—Vulnerability to sea level rise along the Gulf of Mexico: (A) western coastal areas and (B) eastern coastal areas (Hammar-Klose and Thieler 2001, Titus and Richmond 2001).

(A)

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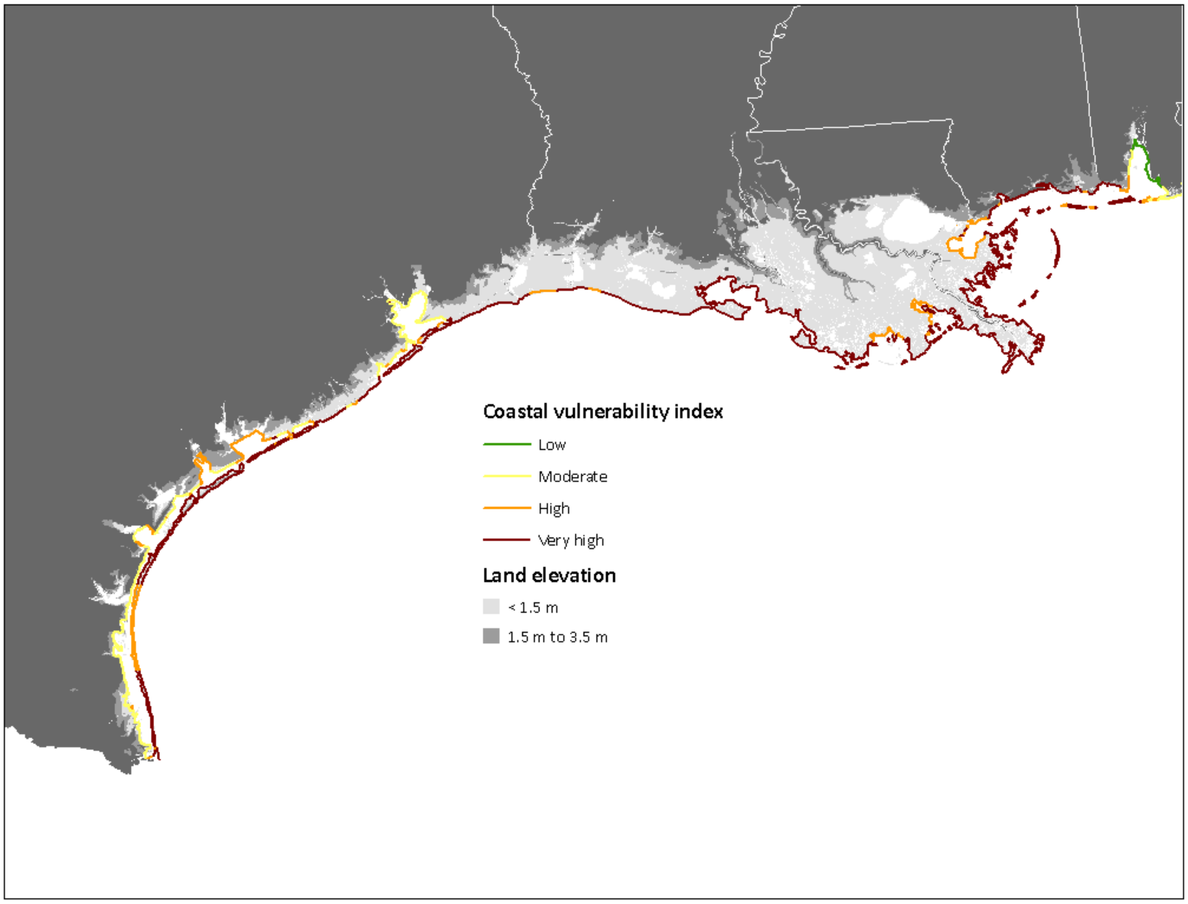


(B)

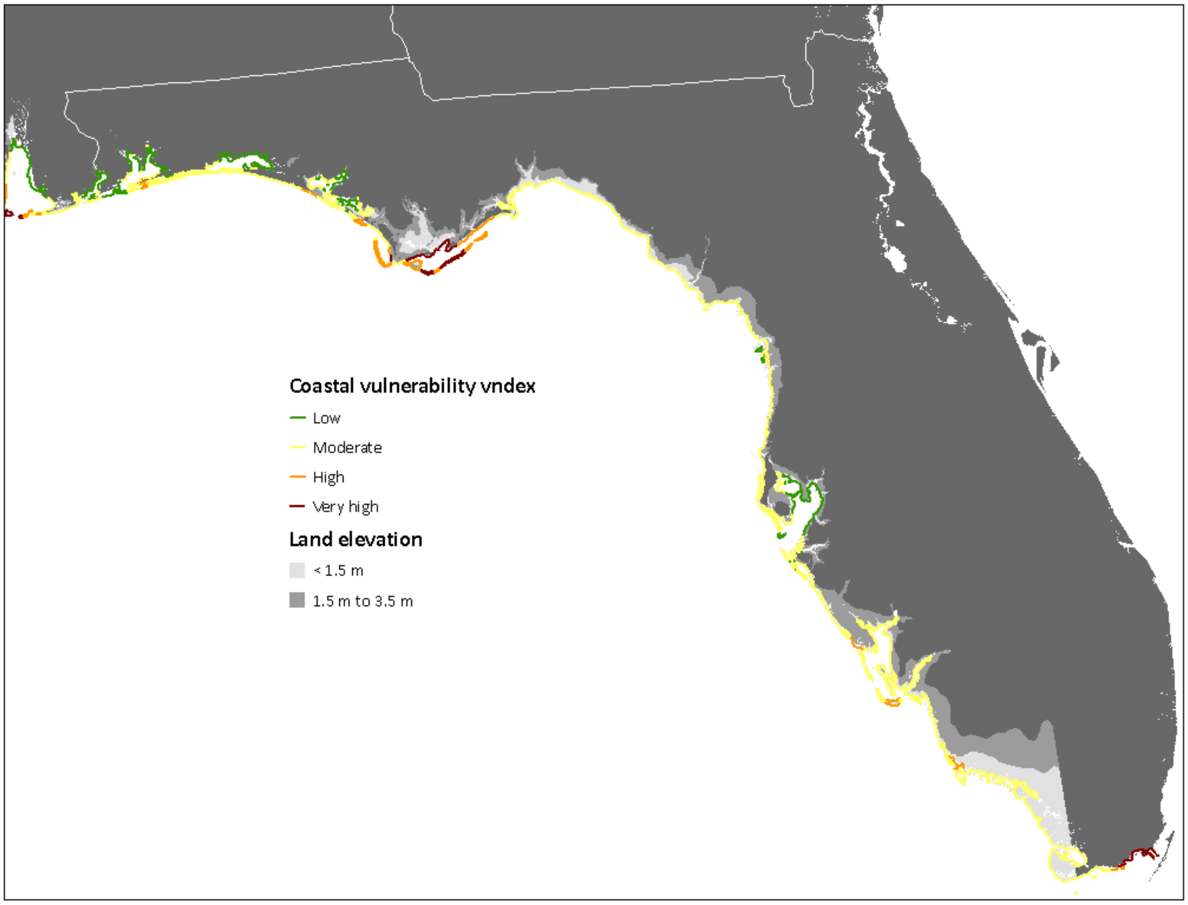
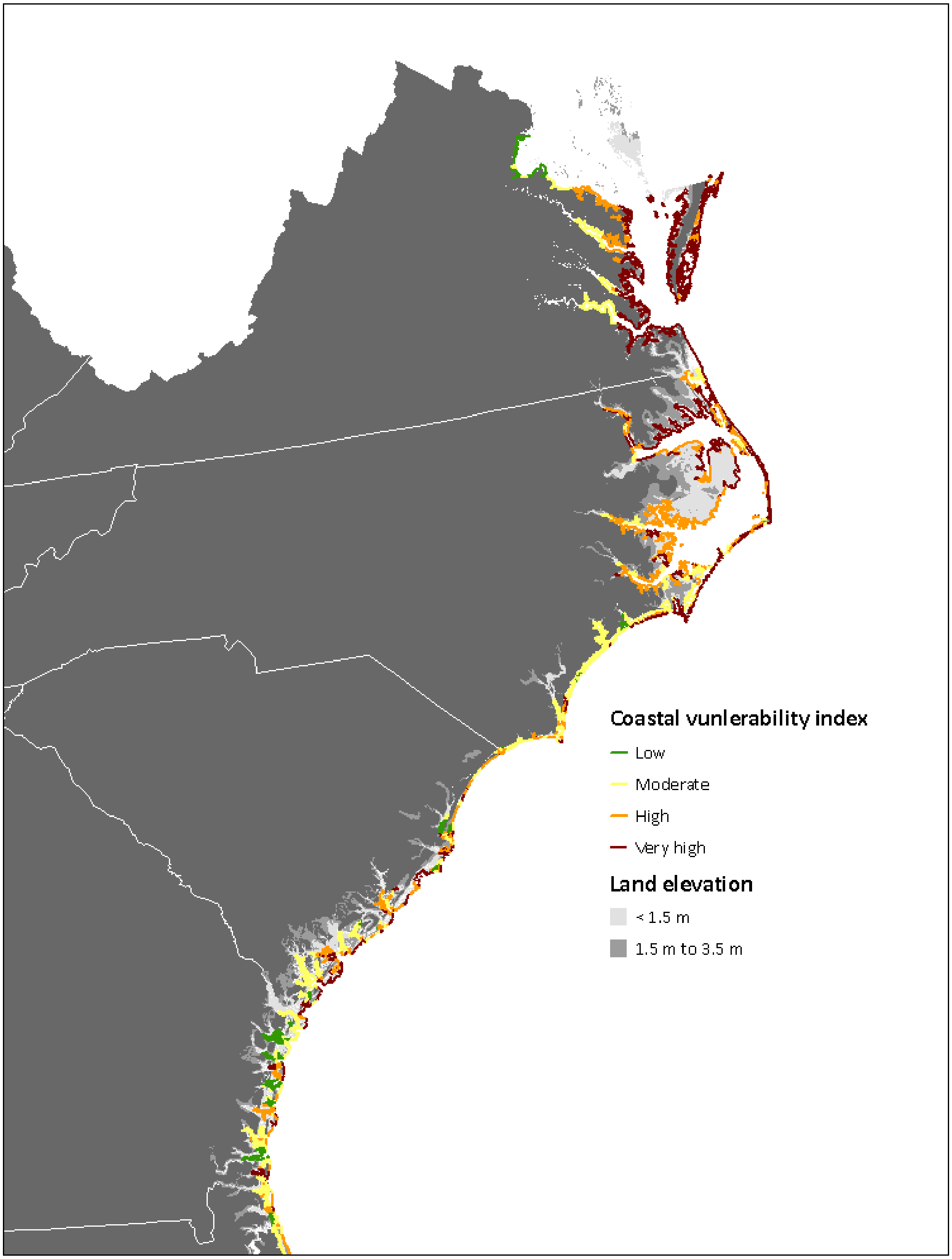

Coastal vulnerability vndex
Low
Moderate
High
Very high
Land elevation
< 1.5 m
1.5 m to 3.5 m

Figure 13-19—Vulnerability to sea level rise along the Atlantic Ocean: (A) northern coastal areas and (B) southern coastal areas (Hammar-Klose and Thieler 2001, Titus and Richmond 2001).

(A)


Coastal vunlerability index
Low
Moderate
High
Very high
Land elevation
< 1.5 m
1.5 m to 3.5 m

(B)