9 Managing Forest Water Quantity and Quality under Climate Change

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Water is a critical resource of the Southern United States and is intimately linked to other ecosystem and societal values. The South is known for its warm climate, rich water resources (Figure 9.1), and large acreage of forest lands that provide an ideal place for people to live. Indeed, water availability is central to sustaining an economy that relies on irrigation agriculture, forestry, recreation,

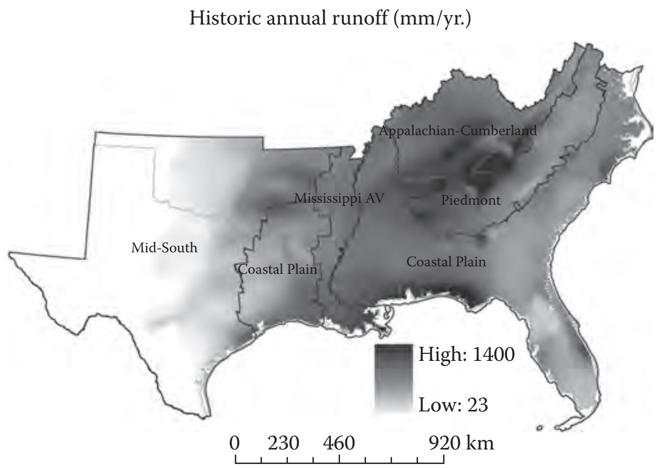


FIGURE 9.1 Mean annual runoff (1960–1980) distribution across the subregions of the South.

industry, power generation, transportation, and most importantly, the long-term future of natural ecosystems and human society (Hossian et al. 2011).

As in many other regions in the United States, the quality and quantity of water resources in the South are at risk from climate change, land conversion from forests to urban uses, increasing water demand from a growing population, and sea level rise (Lockaby et al. in press). A recent national climate change assessment (Karl et al. 2009) suggests that droughts, floods, and water quality problems are all likely to be amplified by climate change in most regions in the United States.

Understanding the potential impacts of climate change and associated stresses on water resources is the key to developing overall adaptation responses to minimize negative consequences at the local level. Today's forest managers face new challenges to maintain forest ecosystem services that are threatened by multiple stressors (Pacala and Sokolow 2004; Ryan et al. 2010). Adapting management practices to climate change by creating more resilient ecosystems is an essential step toward sustainability (Baron et al. 2009). But how much leverage can we expect using forest management in efforts to adapt to the more extreme hydrologic regimes caused by climate change (Ford et al. 2011b)? And what specific options and barriers will affect implementation of management strategies at the stand and landscape scales? Those are the kind of questions that scientists and managers must address if they expect to respond to climate change (National Research Council of the National Academies 2008).

The direct impacts of climate change on water resources will depend on how climate change alters the amount, form (snow vs. rain), and timing of precipitation and how this subsequently affects baseflow, stormflow, groundwater recharge, and flooding (Karl et al. 2009). Climate change poses profound threats to water quantity and quality across the South. The diverse climate, geology, and topography (Sun et al. 2004) and resulting physiographic differences create complex seasonal and spatial hydrologic patterns and water resource distributions (Figure 9.1). Approximately 8000 km of southern coastline are highly vulnerable to sea level rise, adding another level of complexity to water resource management. The large contrast of hydrometeorological and socioeconomic characteristics across the five broad subregions of the South (Coastal Plain, Piedmont, Appalachian-Cumberland Highland, Mississippi Alluvial Valley, and Mid-South) creates a wide range of water quantity and quality responses to climate change, adaptation, and mitigation management options (Table 9.1).

GOALS AND APPROACHES

The goals of this chapter are to provide land managers and policy makers a state-of-the-science summary of potential water resource responses to anticipated climate change, and to identify relevant

TABLE 9.1
Topographic and Hydroclimatic Characteristics of the Five Subregions of the South and Key Water Quantity and Quality Concerns Related to Climate Change

Ecoregion	Topography	Climate and Hydrology[a]	Key Concerns of Climate Change
Appalachian-Cumberland	Rugged topography with steep hillslopes and high elevations (300–2000 m)	High precipitation (1100–2000 mm/year), high runoff (400–1200 mm/year) and runoff ratio (>50%); low evapotranspiration caused by low air temperature	Increased flooding; droughts; erosion and sedimentation; and water temperature
Coastal Plain	Poorly to well drained flat lands with elevation <100 m above sea level	High precipitation (>1200 mm/year) and warm temperature, generally exceeding potential evapotranspiration leaving excess moisture, high evapotranspiration, variable runoff amount (300–800 mm/year), low runoff ratio (<30 percent most area) and slow-moving streams	Changes in wetland hydroperiod and water quality; hurricane impacts; inundation and saltwater intrusion due to sea level rise
Mid-South	Highly variable from flat coastal areas and Great Plains to rugged topography in the Ouachita-Ozark Mountains (0–2500 m)	A large climatic gradient; highly variable precipitation (400–1500 mm/year), runoff (50–550 mm/year) and runoff ratio (5–45%)	Increased flooding in the mountains; droughts in the plains; increased sedimentation due to higher rainfall intensity
Mississippi Alluvial Valley	Flat lands, low elevation (<50 m)	High precipitation, 1200–1500 mm/year, with a decreasing south–north gradient; low runoff amount (450–550 mm/year) and runoff ratio (<35%)	Change in wetland hydrology; inflows from upstreams; water quality from agricultural runoff and river transportation
Piedmont	Moderate slopes, elevation 100–300 m	High precipitation (1200–1400 mm/year), moderate runoff amount (200 to 800 mm/year), and ratio 20–60%	Increasing drought; water use demand; reactivation of sediment

[a] Runoff ratio is the ratio of mean annual runoff to mean annual precipitation, the balance of precipitation is lost to the atmosphere as evapotranspiration.

mitigation and adaptation practices to better manage forest watershed resources in the South. Our specific objectives are to (1) document and project the consequences of climate change and variability in altering the quantity, quality, and timing of water supplies at multiple scales; (2) present case studies to demonstrate how management can mitigate and adapt to climate change in different subregions; and (3) discuss the vulnerability of southern forest watersheds to water resource impacts resulting from climate change and other important stresses over the next 50 years.

This chapter provides an examination of historical and projected future climate changes and the impacts of rising sea levels on key hydrologic processes, water supply and demand related to human use, and water quality in southern forest watersheds. We examined the interactions of climate and water using long-term monitoring data collected at numerous sites (both U.S. Geological Survey and U.S. Forest Service stations) throughout the South. Statistical and process-based simulation models were used to project potential changes in water resources under multiple climate scenarios developed by combining four commonly used general circulation models (MIROC3.2, CSIROMK2, CSIROMK3.5, and HadCM3) with two emissions storylines (Intergovernmental Panel on Climate Change 2007): A1B representing low population/high economic growth and high-energy use, and

TABLE 9.2
Expected Response (Increase, Decrease, or Both) by Water Quantity and Quality Parameters to Climate Change and Other Key Stressors in the Southern United States

	Climate Change and Other Stressors			
Water Parameter	**Higher Air Temperature**	**Precipitation Change**	**Rising Sea Level**	**Land Use Changes (Population Growth, Urbanization)**
Water supply	Decrease	Both	Decrease	Decrease
Evapotranspiration	Increase	Both	Increase	Decrease
Peakflow, stormflow	Decrease	Both	Increase	Increase
Low flow	Decrease	Both	Increase	Both
Wetland hydroperiod	Decrease	Both	Increase	Decrease
Consumptive water use	Increase	Increase	Increase	Increase
Water temperature	Increase	Increase	Decease	Increase
Soil erosion, sedimentation	Both	Both	Increase	Increase
Chemical loading	Both	Both	Increase	Increase

B2 representing moderate growth and low-energy use. Most of the sections that follow utilize one or more of the following climate scenarios: CSIROMK3.5 A1B, CSIROMK2 A1B, HadCM3 B2, and MIROC3.2 A1B.

The chapter is organized in two main parts. First, we evaluate the vulnerability of water resources to future changes in climate and associated stresses at two different spatial scales: region-wide analyses and subregion case studies. The region-wide sections examine how key water resource processes are expected to respond to climate change across the entire South. We then propose adaptation methods based on our current understanding of how hydrologic processes are likely to respond to climate change over the next 50 years. Second, case studies provide more detailed investigations of particular processes within a given subregion, A response matrix (Table 9.2) that relates stressors and water quantity and quality parameters is also provided to summarize how key hydrologic parameters are expected to respond to future climate and development stressors.

WHAT WE ALREADY KNOW AND DO NOT KNOW

Climate change in the South—both observed and projected—is spatially complex. Across the region, mean air temperature increased 0.9°C from 1970 to 2008 (Karl et al. 2009). Although the amount of very heavy rainfall increased 15–20% from 1958 to 2007, the change in annual precipitation varied from a decrease of up to 15% in the Carolinas to an increase of 15% in areas of the Mid-South. Consequently, drought frequency has increased in this traditionally wet region (Karl et al. 2009). Mean annual temperatures in the region are expected to increase by an additional 2.5–3.5°C over the next 50 years (McNulty et al. in press), and water resources will likely become increasingly stressed (Lockaby et al. in press).

Climate change can impact hydrologic processes and water resources directly by altering precipitation, evapotranspiration, groundwater table, soil moisture, or streamflow; and indirectly by degrading water quality or reducing the water available for irrigation. Long-term streamflow data suggest that over the past century mean annual streamflow has increased in the Southern United States; and this increase is associated with increased precipitation (Intergovernmental Panel on Climate Change 2007; Karl and Knight 1998; Lins and Slack 1999). However, the general circulation models do not agree on the predicted change in direction (increases or decreases in total precipitation) of future precipitation events for the Eastern United States (Intergovernmental Panel on Climate Change 2007). A more consistently predicted aspect of future precipitation is that the frequency of the extremes will increase. Most general circulation models predict that as the climate warms, the frequency of extreme precipitation will increase across the globe (O'Gorman and Schneider 2009). Indeed, many regions of the United States have experienced an increased frequency of precipitation extremes over the last 50 years (Easterling et al. 2000a; Huntington 2006; Intergovernmental Panel on Climate Change 2007). However, unlike frequency, projections of the timing and spatial distribution of extreme precipitation are among the most uncertain aspects of future climate predictions (Allen and Ingram 2002; Karl et al. 1995). Despite these uncertainties, recent experience with droughts and low flows in many areas of the United States indicate that even small changes in drought severity and frequency have a major impact on agricultural productions and ecosystem services, including drinking water supplies (Easterling et al. 2000b; Luce and Holden 2009).

Although the properties that define water quantity and quality presented in Table 9.2 are intimately coupled, we discuss these two major response categories separately. For water quantity, we focus on key hydrologic fluxes including evapotranspiration, annual water yield, and low flows. For water quality, we examine water temperature and rainfall erosivity, the two parameters that are directly linked to climate change. We also quantify and map the locations with water supply stress to humans by linking water availability and human water use at a basin scale.

WATER QUANTITY

Evapotranspiration

Evapotranspiration is the combination of evaporation of water from plant and ground surfaces and transpiration. The proportion of rainfall that does not get consumed for evapotranspiration is available for streamflow and soil/groundwater recharge. Thus, evapotranspiration is a major regulator of streamflow, soil moisture, and groundwater recharge. In the South, a large proportion of annual precipitation (50–85%) is returned back to the atmosphere as evapotranspiration, leaving less than half to produce runoff (Lu et al. 2009). Most of the factors that control evapotranspiration—solar radiation, air temperature, precipitation, atmospheric carbon dioxide (CO_2) concentration, and vegetation characteristics—are expected to change under climate change (Sun et al. 2011b). Therefore, the effects of climate change on water resources and ecosystem function can be partially explained by alterations in evapotranspiration (Sun et al. 2011a).

Effects of CO_2 increase from climate change: The effects of increasing CO_2 concentration on evapotranspiration are complex, varying with scale, stand age, and other factors. At the plant's leaf surface, a variety of stomatal responses generally cause transpiration rates to decrease in response to long-term CO_2 increases (Beerling 1996; Franks and Beerling 2009; Prentice and Harrison 2009; Warren et al. 2011). However, at the stand level, the effects of CO_2 increase are highly dependent on the age of the stand. For example, in observational and modeling studies, evapotranspiration decreased by 11% in older stands, but increased in younger stands as a result of leaf area increases (Leuzinger and Korner 2007; Warren et al. 2011). Other modeling studies also suggest that the physiological effect of CO_2 on forest stand transpiration or evapotranspiration may vary between increases and decreases or may not change at all, depending on forest type and climate (Hanson et al. 2005; Ollinger et al. 2008; Tague et al. 2009). However, the effect of increasing CO_2 might be modest when the responses of leaf phenology and growth rate adjustment are compared to changes in other climatic variables such as precipitation, temperature, and humidity (Cech et al. 2003; Leuzinger and Korner 2010).

Forest evapotranspiration rates and the sensitivity of evapotranspiration to climate change are affected by tree species and forest type. For example, Ford et al. (2011a) showed that for a given tree diameter, yellow-poplar (*Liriodendron tulipifera*) transpiration rates were nearly twofold larger than hickory (*Carya* spp.) and fourfold larger than oaks (*Quercus* spp.). Yellow-poplar transpiration rates were also much more responsive to climatic variation compared to oaks and hickories (Ford et al.

2011a). In general, pine (*Pinus* spp.) forests are much more responsive to climatic variation than deciduous forests (Ford et al. 2011a; Stoy et al. 2006).

Changes in forest species composition in response to climate change may also affect evapotranspiration. Using climate change scenarios, Iverson et al. (2008) estimated that of 134 tree species examined, about 66 species would gain and 54 species would lose at least 10% of their suitable habitat, with most of the species habitat moving generally northeast. As species distributions change in response to climate change, especially if large areas of forests experience catastrophic mortality from droughts (Breshears et al. 2005), pest outbreaks, or other factors, evapotranspiration will also likely change.

Canopy and litter interception: Canopy and litter surface area can intercept precipitation and reduce the amount reaching the soil surface. Referred to as interception, this is an important component of forest evapotranspiration in the Eastern United States (Helvey 1967; Helvey and Patric 1965; Savenije 2004). Past work in western North Carolina (Helvey 1967), Georgia (Bryant et al. 2005), and coastal South Carolina (Amatya et al. 1996; Harder et al. 2007) found that interception losses varied from 12% to 22% of total precipitation, depending on forest type and elevation. Canopy and forest litter interception rates are affected by storm characteristics (wind speed, rainfall duration, intensity and drop size, and number of events), and vegetation and litter properties (canopy structure, leaf area index, and litter quality and amount); thus, climate change will likely alter canopy and litter interception losses (Crockford and Richardson 2000).

Evidence of climate change effects: Evidence of evapotranspiration response to climate change largely comes from observational studies that use proxies for evapotranspiration, such as estimates of evapotranspiration from the balance of precipitation and river discharge records, herbarium samples of leaves collected over time, agriculturally important climate indices, and remotely sensed forest greenup and senescence images. However, whether evapotranspiration is increasing or decreasing at continental and global scales is debatable, based on existing evidence. For example, Walter et al. (2004) concluded that evapotranspiration has been increasing across most of the United States, in six major basins, at a rate of 10.4 mm per decade. The increase of evapotranspiration was likely due to increase of irrigation, groundwater withdrawal, and increase precipitation (Walter et al. 2004), or expansion of forest areas in the southeast (Trimble et al. 1987). In contrast, Labat et al. (2004) have shown that global river discharge has been increasing at a rate of 4% for each 1°C increase in global temperature, suggesting a reduction in evapotranspiration. This increase in discharge has been directly attributed to the physiological effect of CO_2 decreasing evapotranspiration and not to the effect of changing land use (Gedney et al. 2006). A study of leaf samples collected from 1890 to 2010 showed a significant decline in stomatal density and maximum stomatal conductance (Lammertsma et al. 2011), supporting the notion of a downward trend in plant transpiration. Therefore, the combined effects of physical and chemical climate change on evapotranspiration are uncertain for certain ecosystems or regions.

Effects of growing-season length: The potential growing season for deciduous forests is increasing over time, which could increase annual evapotranspiration and thereby offset any possible decrease brought about by the effects of CO_2 on stomatal conductance. For example, Kunkel et al. (2004) show that the frost-free season across the United States has lengthened by about two weeks, but this phenomenon is spatially variable with more of an increase in the West than the East. The impacts of warmer winters and springs on deciduous forest greenup and senescence are difficult to generalize (Hänninen and Tanino 2011). For several boreal and temperate tree species, growth cessation in the autumn might come earlier with increasing temperatures. For others, spring bud burst might be delayed by warmer temperatures. For example, Zhang et al. (2007) show that North American lower-latitude forests have a delayed greenup over time, presumably because they do not receive the requisite chilling hours (Schwartz and Hanes 2010); in higher latitudes where chilling requirements are still being met, greenup is occurring sooner. This means that evapotranspiration in the lower latitudes should be delayed over time while evapotranspiration in the higher latitudes should be advancing over time.

Along with being affected by regional climate change, forest phenology can also be affected by local climate, specifically the heat-island effect and other influences from urban centers (Elmore et al. 2012). Lastly, important interactions between climate and abiotic conditions can also constrain evapotranspiration. For example, the potential increase in evapotranspiration resulting from a lengthened growing season can be constrained by reduced water availability and droughts that often arise late in the growing season (Jung et al. 2010; Zhao and Running 2010).

Low Flows

Low flows refer to streamflow during prolonged dry weather, a seasonal phenomenon and an integral component of the flow regime of any river (Smakhtin 2001). Low flows are normally derived from groundwater discharge and usually occur in the same season every year. Low flow is affected by climate, topography, geology, soils, and human activities (Smakhtin 2001). Effects of climate change on low flows have important consequences for water supply to reservoirs, transportation, power generation, aquatic habitats, and water quality (dissolved oxygen concentration, water temperature, salinity, and nutrient levels).

Previous studies suggest that the low flow characteristics have been changing in the South. For example, Lins and Slack (1999, 2005) reported significantly upward trends in annual minimum flows and 10th-percentile flows from 1940 to 1999 at most sites in the Appalachian-Cumberland Highland, Mississippi Alluvial Valley, and Mid-South; whereas many sites in the Coastal Plain and Piedmont exhibited significantly downward trends. McCabe and Wolock (2002) corroborated this result and found that the upward trends in streamflow were a result of a steep increase in precipitation beginning in the early 1970s.

Modeling study at watershed and regional scales: We used two different methods, the 7Q10 method (Telis 1992) and the minimum flows and levels method (Neubauer et al. 2008), to determine low flows in this study. The 7Q10 method, defined as the lowest mean flow that occurs for a consecutive 7-day period with a recurrence interval of 10 years, has been used for identifying extremely low flows (Reilly and Kroll 2003; Telis 1992). Conversely, the minimum flow and levels method is defined as the minimum water flows and/or levels required to prevent significant harm to water resources that are subject to water withdrawals (St. Johns River Water Management District 2010). This second method calculates how often and for how long the high, mean, and low water levels and/or flows need to occur to prevent significant harm.

For detecting historical low flow trends, we selected three forest-dominated headwater watersheds: Cache River at Forman, IL; Big Creek at Pollock, LA; and St. Francis River at Wappapello, MO—all located in the Mississippi Alluvial Valley. These watersheds have gauging stations maintained by the U.S. Geological Survey and have rather long (60–90 years) streamflow record periods.

To predict future low flow conditions, we applied an improved monthly scale Water Supply Stress Index model (Caldwell et al. 2012; Sun et al. 2008). The model uses a monthly water balance procedure that is sensitive to land cover and climate. It accounts for evapotranspiration, infiltration, soil storage, snow accumulation and melt, surface runoff, and baseflow processes within each basin, and conservatively routes discharge through the stream network from upstream to downstream watersheds. The model does not account for flow regulation by dams or for water diversion projects such as interbasin transfers. It has been evaluated against measured streamflow and evapotranspiration data at watershed to regional scales (Sun et al. 2011a). Sun et al. (2008) used this model to assess the impact of future climate change on monthly flows across some 2100 U.S. watersheds defined at the 8-digit Hydrologic Unit Code (HUC) scale. We focused on the 677 watersheds that encompass the South, some of which receive water from outside the region.

Climate data were derived using the CSIROMK3.5 A1B, CSIROMK2 A1B, HadCM3 B2, and MIROC3.2 A1B climate scenarios and downscaled to the 8-digit HUCs (Coulson et al. 2010a,b). Model results are presented as the mean low flow response under these four climate scenarios.

Using the 80% exceedance probability in their flow duration curves, we found that the low flow rates were 0.16 m^3/s at the Cache, 0.48 m^3/s at the Big Creek, and 5.01 m^3/s at the St. Francis gauging stations. The exceedance probabilities for the 60- and 90-day low flow durations at the three stations have increased during the study periods (Table 9.3). It appears that the low flows are occurring more frequently over time as the watersheds have become drier.

Water supply modeling shows that low monthly mean flows are projected to decrease 6.1% per decade across the South into the first half of the twenty-first century (Figure 9.2), with the largest decreases occurring in the Appalachian-Cumberland Highland, Mississippi Alluvial Valley, Mid-South, and western Coastal Plain (Table 9.4 and Figure 9.3). The large decrease in the Mississippi Alluvial Valley is partially the result of decreasing flows from streams outside of the southern region. Portions of the Coastal Plain and Mid-South were projected to have upward trends in low flows; however, these increases were not statistically significant (Figure 9.2). In all subregions, general low flows amounts were projected to decrease from 2010 to 2060 (Figure 9.3).

Total Water Yield

Water yield is the sum of surface runoff and groundwater discharge from a watershed. Changes in water yield from streams can greatly affect the health of aquatic ecosystems (Postel and Richter 2003) as well as public water supply capabilities (Postel and Carpenter 1997). To assess how climate change and increased water demand from population growth may affect future water supply stress in the South, we used an improved water supply stress model (Sun et al. 2008) to predict future water yields and water supply stresses.

Current trends: The effect of climate change on water yield has been analyzed using historical flow records at long-term gauging sites in watersheds with minimal human alteration. Lins and Slack (1999, 2005) reported significantly upward trends from 1940 to 1999 in all percentiles of streamflow—except maximum flows—at most sites in the Appalachian-Cumberland Highland, Mississippi Alluvial Valley, and Mid-South, but only at a few sites in the Coastal Plain and Piedmont. Groisman et al. (2003) and McCabe and Wolock (2002) corroborated this result and found that the upward trends in streamflow occurred as a result of a steep increase in precipitation beginning in the early 1970s. Conversely, Krakauer and Fung (2008) argued that increased evapotranspiration from climate change will ultimately lead to decreasing streamflows.

Future projections: Using the mean water yield response under the CSIROMK3.5 A1B, CSIROMK2 A1B, HadCM3 B2, and MIROC3.2 A1B climate scenarios, we found that annual water yield would decrease Southwide by approximately 10 mm per decade (3.7% of the 2001–2010 levels), or 50 mm (18% of the 2001–2010 levels) by 2060 (Figure 9.4 and Table 9.5). Although we found considerable seasonal variability in the projected water yield, the general trend was a statistically significant decrease ($p < 0.05$). Likewise, although we found considerable variability in the magnitude of water yield changes among the four climate scenarios, all four scenarios result in downward trends.

Mean water yield varied considerably across the South as well (Table 9.5 and Figure 9.5), with most of the Appalachian-Cumberland Highland and Mississippi Alluvial Valley and parts of the Mid-South and Coastal Plain, exhibiting statistically significant decreases of more than 2.5% per decade. Although portions of the Coastal Plain and Mid-South are projected to have upward trends in water yield, these increases were not statistically significant at the 0.05 level. The projected downward trends in water yield are not constant through the entire 2010–2060 period. In all subregions, water yield is projected to decrease from 2010 to 2025, level off from 2025 to 2045, and decrease again after 2045 (Figure 9.6).

Impacts on water stress: Decreases in water supply and increases in water demand as a result of population growth will likely combine to increase water supply stress to humans and ecosystems. We define water supply stress as the ratio of human-related water demand/withdrawal by all economic sectors to the total amount of water available (Sun et al. 2008). Water demand includes

TABLE 9.3
Exceedance Probability and Recurrence Interval for the 60- and 90-day Low Flows over Consecutive 10- or 20-year Periods at Three U.S. Geological Survey Stations Draining Forest Lands of the Mississippi Alluvial Valley

Site Name	Station Number	Low Flow (m^3/s)	Probability (%)	RI[a] (years)	Probability (%)	RI[a] (years)	Probability (%)	RI[a] (years)	Probability (%)	RI[a] (years)
Cache River at Forman, IL	3612000	0.16	20 years (1922–1942)		20 years (1942–1962)		20 years (1962–1982)		20 years (1982–2002)	
			60-day flow duration							
			0.54	1.9	0.57	1.75	0.62	1.62	0.67	1.5
			90-day flow duration							
			0.4	2.5	0.33	3	0.43	2.33	0.52	1.9
Big Creek at Pollock, LA	7373000	0.48	20 years (1942–1962)		20 years (1962–1982)		20 years (1982–2002)		10 years (2002–2011)	
			60-day flow duration							
			0.43	2.3	0.62	1.6	0.28	3.5	0.8	1.25
			90-day flow duration							
			0.38	2.6	0.57	1.75	0.29	3.4	0.7	1.4
St. Francis River at Wappapello, MO	7039500	5.01	20 years (1940–1960)		20 years (1960–1980)		20 years (1980–2000)		10 years (2000–2011)	
			60-day flow duration							
			0.61	1.62	0.62	1.61	0.65	1.58	0.58	1.7
			90-day flow duration							
			0.5	2	0.52	1.9	0.6	1.7	0.3	3.2

[a] Recurrence interval.

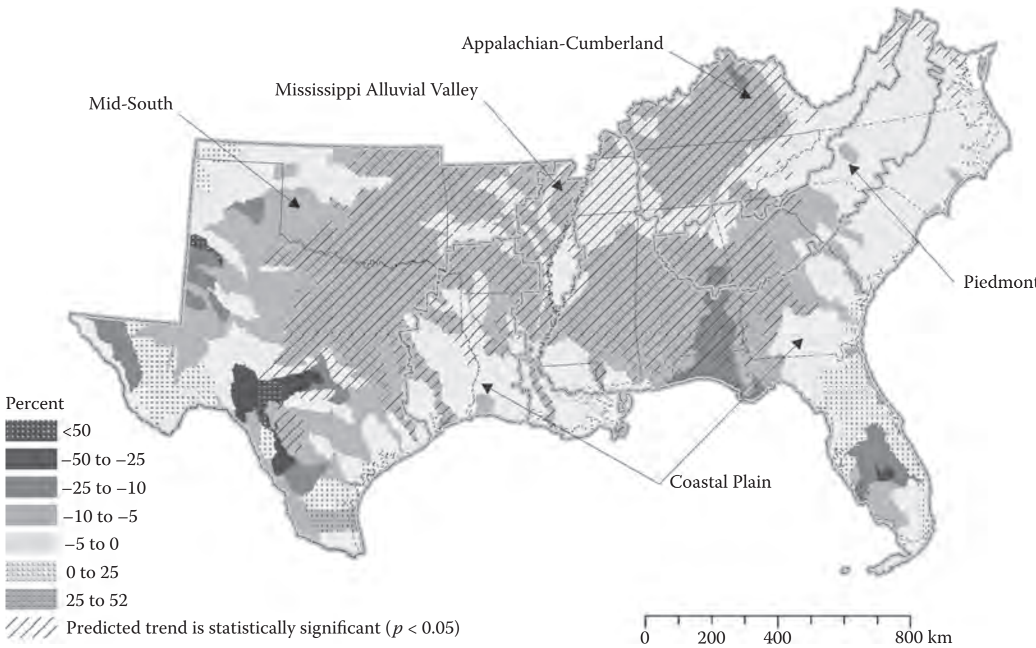


FIGURE 9.2 Predicted trend in annual minimum monthly streamflow for 2010–2060, normalized using the mean of the 2001–2010 annual minimum monthly flow.

water withdrawals by seven water use sectors: commercial, domestic, industrial, irrigation, livestock, mining, and thermoelectric. For future projections, we consider population growth impacts on water demand as well as land use and climate change impacts on water supply.

The impact of declining water yield and increasing population is projected to increase water supply stress over much of the South by 2060, particularly in developing watersheds (Figure 9.7). For example, the Upper Neuse River watershed, providing water supply for the Raleigh-Durham metropolitan area, is projected to experience a climate-induced, 14% decrease in water supply and a growth-induced 21% increase in water demand. This results in an increase in water supply stress from 0.30 in the first decade (2001–2010) to 0.44 in the last (2051–2060). The 0.40 value has been established as a general threshold at which a watershed begins to experience water supply stress (Alcamo et al. 2000; Vörösmarty et al. 2000), although stress may occur at lower or higher values depending on local water infrastructure and management protocols.

TABLE 9.4
Ensemble Mean Predicted Trends in Regional Annual Minimum Monthly Flows Compared to a Baseline Period (2001–2010)

	Baseline	Predicted Trend	
Region/Subregion	**(1 million m³/month)**	**(1 million m³/month/decade)**	**(percent of 2001 to 2010 mean/decade)**
All South	197	−12	−6.1
Appalachian-Cumberland	427	−20	−4.7
Coastal Plain	154	−9.7	−6.3
Mid-South	48	−3.0	−6.3
Mississippi Alluvial Valley	1261	−86	−6.8
Piedmont	41	−2.1	−5.1

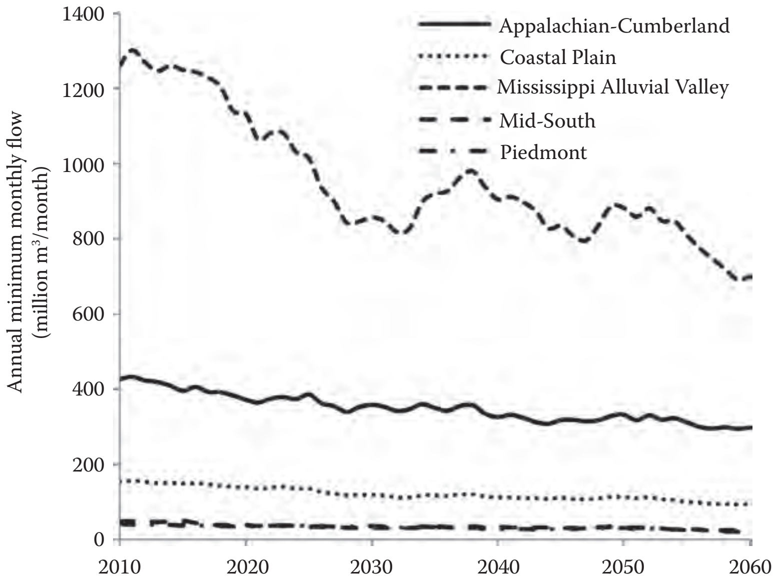


FIGURE 9.3 The annual minimum monthly streamflow by subregion in the Southern United States predicted for 2010–2060 using the mean value of four climate scenarios and displayed using a 10-year moving mean. The MIROC3.2-A1B, CSIROMK2-B2, CSIROMK3.5-A1B, and HadCM3-B2 climate scenarios for the Southern United States (McNulty et al. in press) each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high-energy use, and B2 representing lower population growth and energy use).

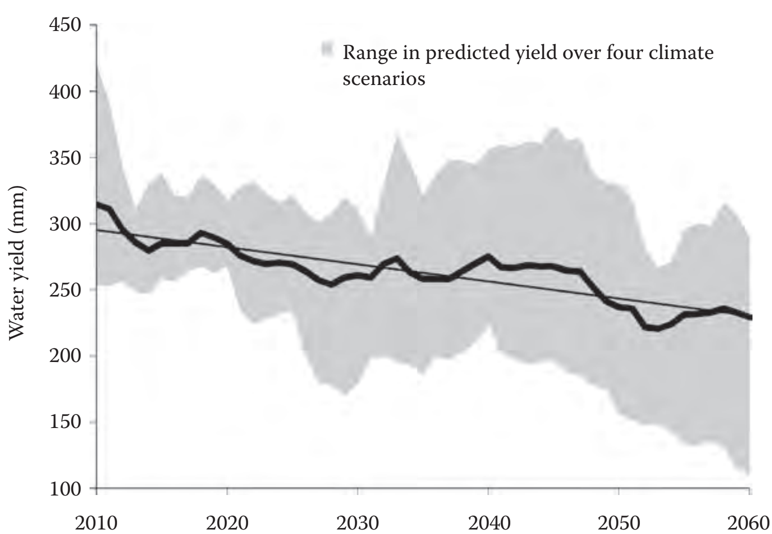


FIGURE 9.4 Predicted southwide annual water yield for 2010–2060. The shaded area represents the range in predicted water yield over the four climate scenarios; the heavy line is the 10-year moving mean; and the thin line is the trend.

TABLE 9.5
Mean of Predicted Trends in Regional Annual Water Yield Based on Four Climate Projections Compared to a Baseline Period (2001–2010)

Region/Subregion	Baseline (mm)	Predicted Trend[a] (mm/decade)	Predicted Trend[a] (% of 2001–2010 mean/decade)
All South	322	−10.2[b]	−3.2[b]
Appalachian-Cumberland	588	−16.2[b]	−2.8[b]
Coastal Plain	392	−11.5[b]	−2.9[b]
Mississippi Alluvial Valley	556	−25.1[b]	−4.5[b]
Mid-South	131	−5.4	−4.1
Piedmont	410	−8.7	−2.1

[a] The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 B2 climate scenarios for the Southern United States each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high-energy use, and B2 representing lower population growth and energy use).
[b] Significant at the 0.05 level.

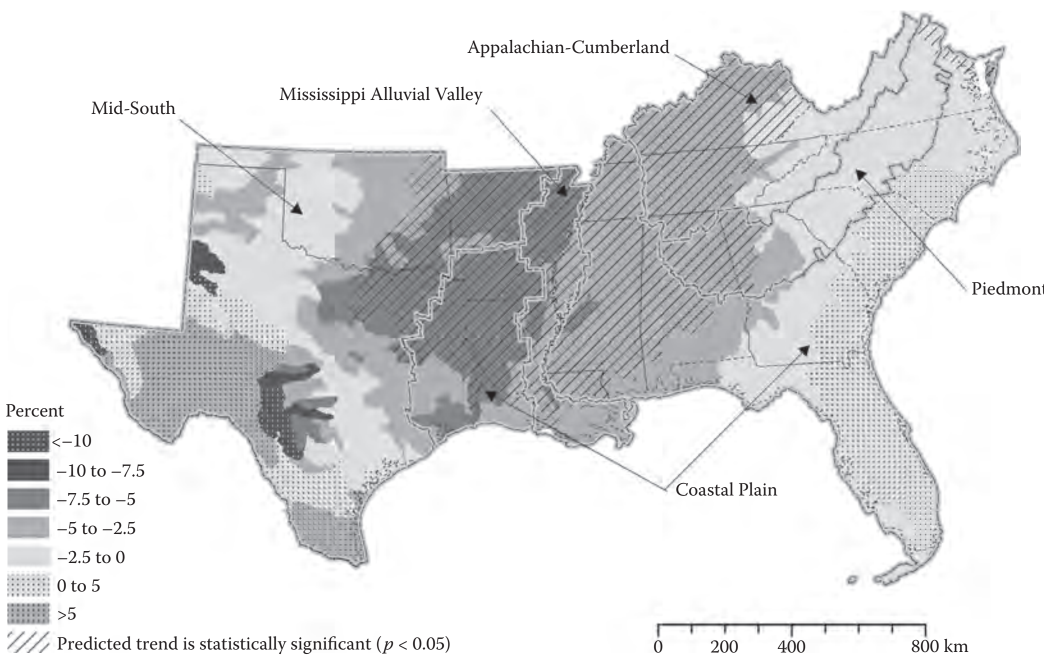


FIGURE 9.5 Predicted trend in mean annual water yield for 2010–2060, normalized using the mean of the 2001–2010 annual water yield.

WATER QUALITY

Climate change not only affects water quantity but also affects water quality (Cruise et al. 1999; Murdoch and Baron 2000; Whitehead et al. 2009). A warming climate may elevate water temperature sufficiently to harm aquatic life (Mohseni and Stefan 1999; Webb et al. 2008). Changes in precipitation amount or storm intensity can affect soil erosion potential by changing the runoff

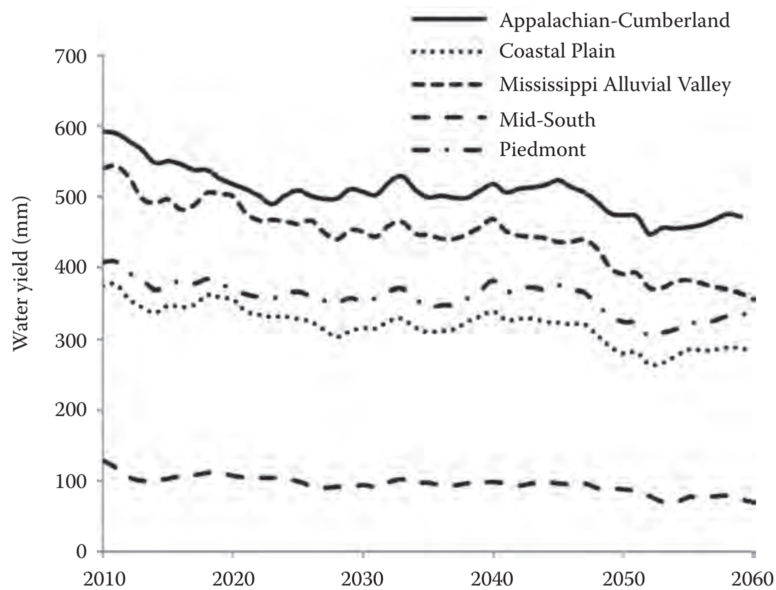


FIGURE 9.6 Predicted trend in annual water yield by subregion in the Southern United States, based on four climate scenarios and using a 10-year moving mean. The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 B2 climate scenarios for the Southern United States (McNulty et al. in press) each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high energy use, and B2 representing lower population growth and energy use).

amount, the kinetic energy of rainfall, or the vegetation cover that resists erosion. Increased erosion results in increased sediment delivery to streams and lakes. Moreover, water temperatures and sediment concentrations may increase in combination with decreased flow rates and velocities, thereby magnifying their individual impacts to aquatic life.

Water Temperature

Climate change is of particular concern for coldwater fish in the Southern Appalachian Mountains. For example, eastern brook trout (*Salvelinus fontinalis*) need dissolved oxygen levels in excess of 8 mg/L, striped bass (*Morone saxatilis*) prefer dissolved oxygen levels above 5 mg/L, and most warm-water fish need dissolved oxygen in excess of 2 mg/L. Increases in water temperature as a result of climate change would decrease instream dissolved oxygen concentrations. Streams in the Southern Appalachians provide crucial habitat for eastern brook trout and other coldwater species such as rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*). The lethal temperature limit for such species is approximately 25°C (Matthews and Berg 1997; Meisner 1990).

Several natural factors influence the extent to which changes in air temperature impact stream temperature; these include total streamflow, relative groundwater contribution to flow (Bogan et al. 2003; Matthews and Berg 1997; Poole and Berman 2001; Sullivan et al. 1990; Webb et al. 2008), and canopy cover over the stream. In addition, human-related factors that influence the air/water temperature relationship include runoff from impervious surfaces (Nelson and Palmer 2007), thermal discharges (Webb and Nobilis 2007), and reservoir releases (Webb and Walling 1993).

Several studies have assessed climate change impacts on U.S. stream temperatures using a variety of scales and resolutions, either by trend analysis of historical stream temperatures (Kaushal et al. 2010) or by developing air/water temperature models and projecting into the future (Eaton and Scheller 1996; Mohseni et al. 1998; O'Neal 2002, van Vliet et al. 2011). We adopted the latter

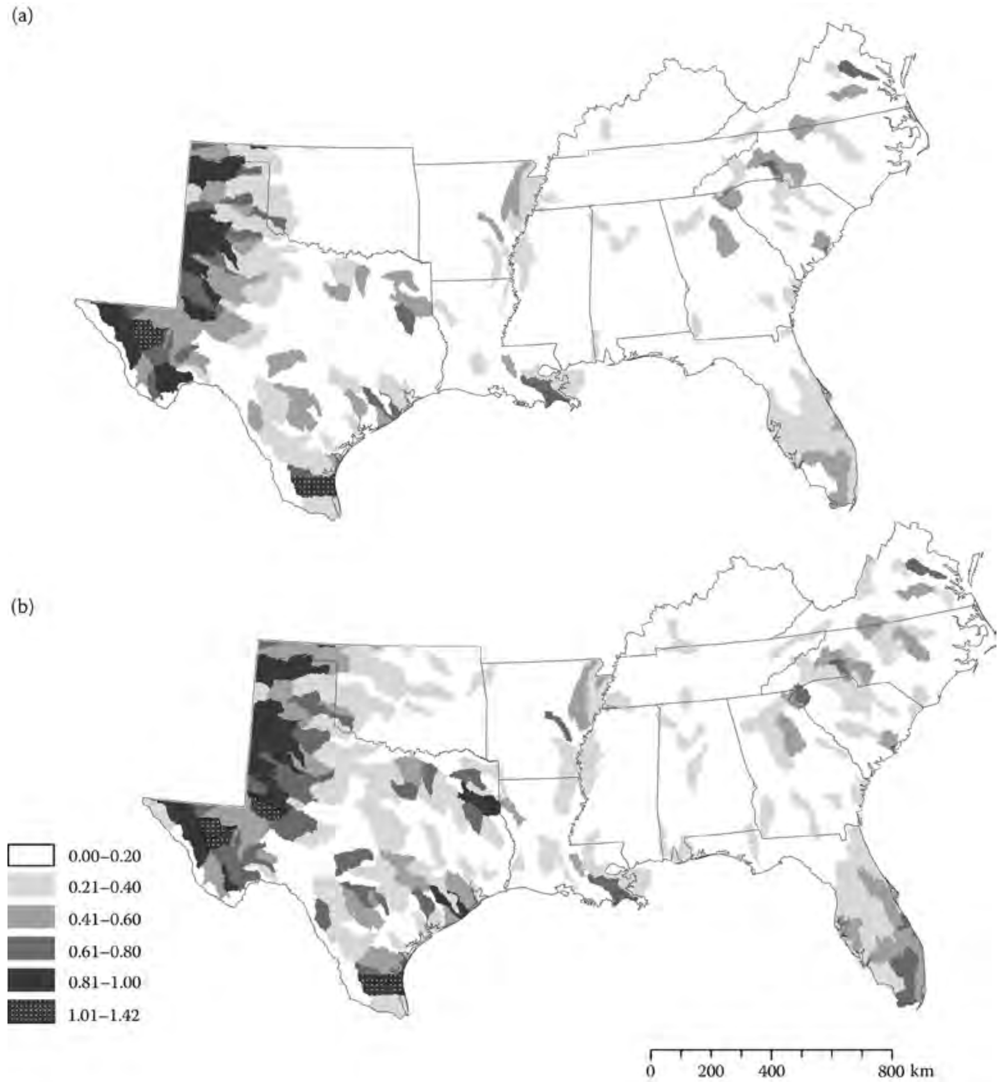


FIGURE 9.7 Mean annual Water Supply Stress Index (a ratio of water demand/water supply) based on four climate scenarios for (a) 2001 to 2010 (baseline), and (b) 2051 to 2060 (predicted). The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 B2 climate scenarios for the Southern United States (McNulty et al. in press) each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high energy use, and B2 representing lower population growth and energy use).

approach, developing monthly air/water temperature models using long-term historical data for 91 relatively undisturbed watersheds in the South, and then predicting how stream temperature changes under future climate scenarios.

Datasets: The stream temperature observations used in this study came from two sources. Seventy sites were selected from the Hydro-Climatic Data Network (Slack et al. 1993), which are all in watersheds with limited hydrologic alteration by humans. An additional 21 sites are located on smaller Forest Service experimental watersheds in the Appalachian-Cumberland Highland and Piedmont subregions of North Carolina. For both datasets, monthly mean stream temperature values were computed from mean daily values. The resulting database includes stream temperature observations collected during some portion of the 1960 to 2011 period in streams with drainage areas that range from 0.1 to 101,033 km^2 (Figure 9.8).

Air temperatures representing the recent time period came from 4-km by 4-km resolution monthly weather data, available from the PRISM Climate Group (Gibson et al. 2002). Data from the PRISM climate grid cells containing the stream temperature measurement sites were used to represent the air temperature at those sites. For future mean monthly air temperature, we used data from the CSIROMK2 B2, CSIROMK3.5 A1B, HadCM3 B2, and MIROC3.2 A1B climate scenarios (Coulson et al. 2010a,b).

Model development: S-curve monthly air/stream temperature models (Mohseni et al. 1998) were developed for each site. A monthly time step was selected because this is the resolution at which most future climate predictions are available. Models were developed using the most recent two years of stream temperature data at each site, leaving the remaining data to validate the models' accuracy in predicting stream temperature outside the periods used for model development.

Once the air/stream temperature models were generated, stream temperature values were computed for the historical time period (1960–2007) using the PRISM air temperature data, and the

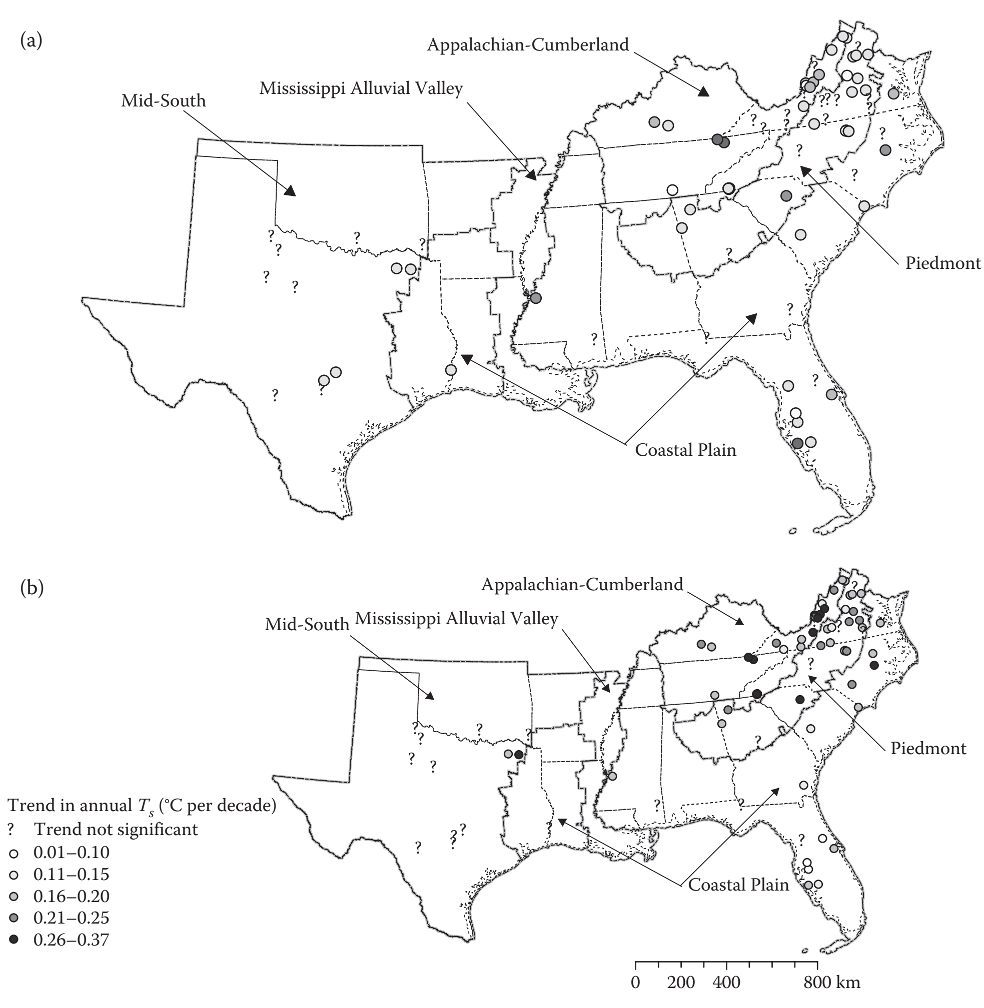


FIGURE 9.8 Estimated trend (1960–2007) in (a) mean annual and (b) annual maximum monthly stream water temperature across the Southern United States (°C/decade).

future (2011–2060) using each of the four climate scenarios to generate monthly time series of historical and future stream temperature at each site. For future predictions, the mean stream temperature over the four climate scenarios was computed to represent the central future condition, and the range in stream temperature among the future climate predictions was computed to represent the uncertainty in these predictions.

Stream temperature was regressed against time during both the historical and future time periods, with the slope of the regression representing the change in stream temperature over time. If the change for either period was significantly different from zero ($\alpha = 0.05$), then we reasoned that the stream temperature had changed in response to air temperature during that period.

Estimated historical trends: Sixty-two of the 91 sites showed significant changes, where mean annual stream temperature increased from 1960 to 2007 (Figure 9.8). The mean temperature increase across the 62 sites was 0.14°C per decade, with a range of 0.08–0.29°C per decade. For the sites with significant changes, the mean changes among subregions did not differ, although five of the largest increases were for sites in the Appalachian-Cumberland Highland.

More relevant to aquatic ecosystems than mean annual stream temperature are the extreme temperature conditions, such as the annual maximum monthly stream temperature. Seventy-one of the 91 sites showed significant changes in annual maximum monthly stream temperature from 1960 to 2007 (Figure 9.8). The mean increase in annual maximum monthly stream temperature for the 71 sites was 0.20°C per decade, with a range of 0.04–0.37°C per decade. The mean for sites in the Appalachian-Cumberland Highland (0.23°C per decade) was significantly different from that for sites in the Coastal Plain (0.15°C per decade) and Piedmont (0.15°C per decade). Significant changes in annual maximum monthly stream temperature were generally larger than the changes in mean annual stream temperature, suggesting that historical changes in climate have had more impact on the extremes in stream temperature than on mean stream temperature.

Predicted future trends: All 91 sites exhibited significant upward trends in mean annual stream temperature from 2011 to 2060 under all four of the future climate scenarios. The mean upward trend over all sites and climate scenarios was 0.26°C per decade, with means ranging from 0.21°C per decade under the HadCM3 B2 scenario (Figure 9.9) to 0.35°C per decade under the MIROC3.2 A1B scenario. The changes between subregions did not differ when compared using the mean of all climate scenarios.

The projected 2011–2060 change in annual maximum stream temperature increased at all 91 sites, and was significant at 27 sites under CSIROMK3.5 A1B, 74 sites under CSIROMK2 B2, 76 sites under HadCM3 B2, and all 91 sites under MIROC3.2 A1B (Figure 9.9). The mean upward trend among all climate scenarios for all sites with significant changes was 0.25°C per decade, ranging from 0.21°C per decade under CSIROMK2 B2 to 0.30°C per decade under MIROC3.2 A1B. Under HadCM3 B2, the mean change for sites in the Appalachian-Cumberland Highland (0.29°C per decade) and Piedmont (0.27°C per decade) was significantly different from the mean change in the Coastal Plain (0.18°C per decade).

Our modeling indicates that stream temperature values in the historical period have likely already increased in much of the South as a result of increasing air temperature, and suggests that they will continue to do so at an accelerated rate in the near future. In particular, sites in the Appalachian-Cumberland Highland and Piedmont have the highest upward trends in mean annual and especially annual maximum monthly stream temperature. Sensitivity of stream temperature to air temperature at a given site can be controlled or influenced by management practices, which will be discussed later in this chapter.

Soil Erosion and Sediment

Sediment is one of the primary threats to water quality in the South (West 2002), where surface erosion is the primary process by which sediment is created. Surface erosion is the detachment and removal of soil or mineral grains from the ground surface. Rainfall and surface runoff are

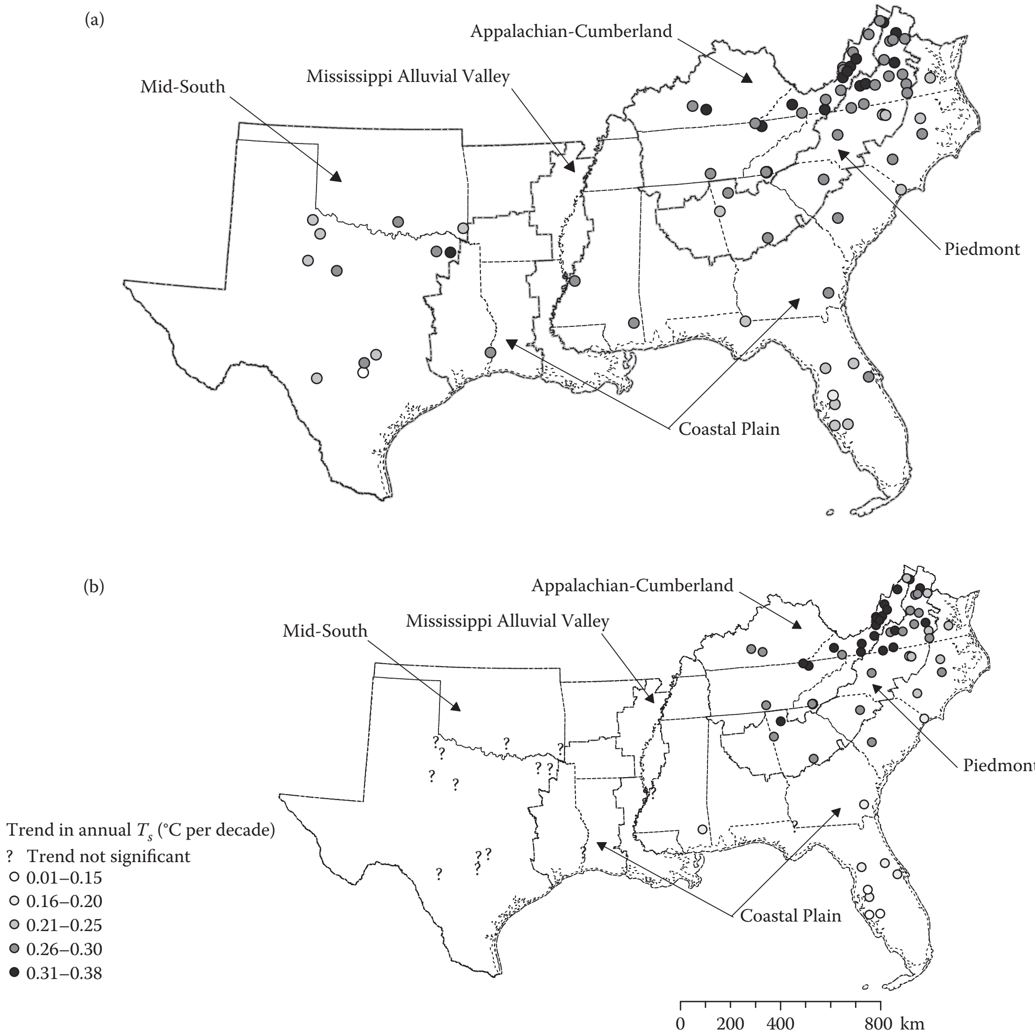


FIGURE 9.9 Predicted trend (2011–2060) in (a) mean annual and (b) annual maximum monthly stream water temperature across the Southern United States (°C/decade) under the HadCM3 B2 climate scenario. The HadCM3 B2 climate scenario for the Southern United States (McNulty et al. in press) combines the HadCM3 general circulation model with the B2 emissions storyline (representing lower population growth and energy use).

the primary forcing agents by which surface erosion occurs and sediment is transported to water bodies.

The Universal Soil Loss equation (Wischmeier and Smith 1978) and its successor, the Revised Universal Soil Loss equation (Renard et al. 1997), have long been used to estimate the amount of surface erosion associated with different environmental conditions and land use activities in the South and elsewhere. The Universal Soil Loss equation was originally developed for estimating surface erosion from cultivated lands, with the Revised Universal Soil Loss equation extending its applicability to rangelands (Renard et al. 1997). Dissmeyer and Foster (1984) showed how the Universal Soil Loss equation could be used in forest lands, and Dissmeyer and Stump (1978) provided specific data for doing so in the South.

The factors used within both equations represent the major parameters affecting surface erosion. Two of these factors are climate related: the rainfall–runoff erosivity factor (R) and the cover-management factor (C). The C-factor is indirectly affected by climate in that temperature and precipitation have long-term influences over the type and density of vegetation that can be sustained in a given area without irrigation or other human interventions. The R-factor provides an index of the intensity and amount of rainfall occurring at a given location over a long period of time, and as such is directly affected by climate.

The R-factor provides a useful surrogate for assessing potential changes in future surface erosion related to climate change. Past research has demonstrated that the amount of surface erosion increases with increased R-factor values (Pruski and Nearing 2002a,b; Renard et al. 1997). R-factor values can be estimated using monthly or annual precipitation data (Renard and Freimund 1994). Therefore, the change in R-factor values is used here to evaluate how future surface erosion may be affected by projected climate changes.

Past evaluations: Changes in R-factor values resulting from climate change have not been demonstrated in the South by directly comparing R-factors computed from recent precipitation data to those originally computed by Wischmeier and Smith (1965) from data for the mid-20th century. Possibly this did not happen because R-factor values were significantly revised by Renard et al. (1997) using additional climate stations and more recent climate data. The more recent R-factor map (Renard et al. 1997) is generally similar to the earlier map in the South, with some marked differences along the Gulf of Mexico in Louisiana and Florida and within the Appalachian-Cumberland Highland. However, it is not clear whether these differences are the result of actual climate change or the inclusion of new stations within these areas.

Predicted precipitation based on general circulation models and emissions storylines has been used to assess how future climate change may affect R-factor values within the conterminous 48 states. In general, the R-factor value changes determined by Phillips et al. (1993) showed little consistency in the South among the four climate scenarios they considered. The one exception was in the Appalachian-Cumberland Highland where all scenarios indicated increased R-factor values. Given their methods, Phillips et al. (1993) warned against overemphasizing the details of their results for individual regions such as the South.

Results from Nearing (2001) also showed little geographic agreement in R-factor predictions for the South under the two climate scenarios considered. This analysis was also at the national scale, but used more recent general circulation models, more sophisticated emissions storylines, finer scale climate projections, and statistical models for predicting R-factor values.

Methods for predicting future trends: Our examination used a somewhat more conservative emissions storyline than past research, and a finer scale climate projection (HadCM3 B2) than past studies. We focused our analysis on the South and used similar-sized drainage basins to display the results. R-factor values are computed using the statistical models of Renard and Freimund (1994) based on mean annual precipitation. Renard and Freimund (1994) also derived models for predicting the R-factor based on the modified Fournier index (Arnoldus 1977), which is calculated using both mean monthly and mean annual precipitation. However, the models based on mean annual precipitation alone predicted R-factor values that more closely matched the values computed by Renard et al. (1997) for the South. Like Nearing (2001), the mean annual precipitation values we used to compute R-factor values were derived from continuous 20-year periods, which in our case were 2011 to 2030, 2031 to 2050, and 2051 to 2070.

To evaluate how much R-factor values would change in the future, we defined 1961 to 1990 as the baseline or "historical" period. R-factor values for this period were computed using precipitation data generated by rescaling 4-km × 4-km PRISM data (Daly et al. 1994) to the 677 southern watersheds whose boundaries are defined by the U.S. Geological Survey 8-digit HUC naming system. The mean watershed area was 3660 km^2. HUC boundaries often extend beyond state boundaries along the interior perimeter of the South to include entire drainage basins. Future precipitation projections were derived from a county-level dataset (Coulson et al. 2010a,b) that was overlaid onto the HUC layer and used to compute area-weighted values for each of the 677 watersheds analyzed.

Determining threshold of concern: Because recommended limits for soil loss caused by surface erosion have not been established for most forest lands in the South, the R-factor change required to exceed a given soil loss limit cannot be determined. Instead, the "threshold of concern" approach (Sassaman 1981) is used here to define the R-factor change limits beyond which undesirable increases in erosion might occur. Although our concern is surface erosion, we used the change in R-factor values to identify when and where mitigation or adaptation actions to future climate change may be most likely.

Soils that have similar physical characteristics exhibit similar surface erosion responses. "Major land resource areas" are U.S. geographic areas composed of similar soils that have been mapped by the U.S. Department of Agriculture (2006). Soils within these areas have developed over millennia in response to a range of rainfall erosivity that is characteristic of the climate in which the soils occur. It seems reasonable that soils developed under higher rainfall erosivity conditions should be able to tolerate a higher relative change in R-factor values than soils developed under lower erosivity conditions. For this analysis we assumed that the threshold for R-factor change is ±10% of the maximum R-factor value identified by Dissmeyer and Stump (1978) for each major land resource area in the South (Figure 9.10).

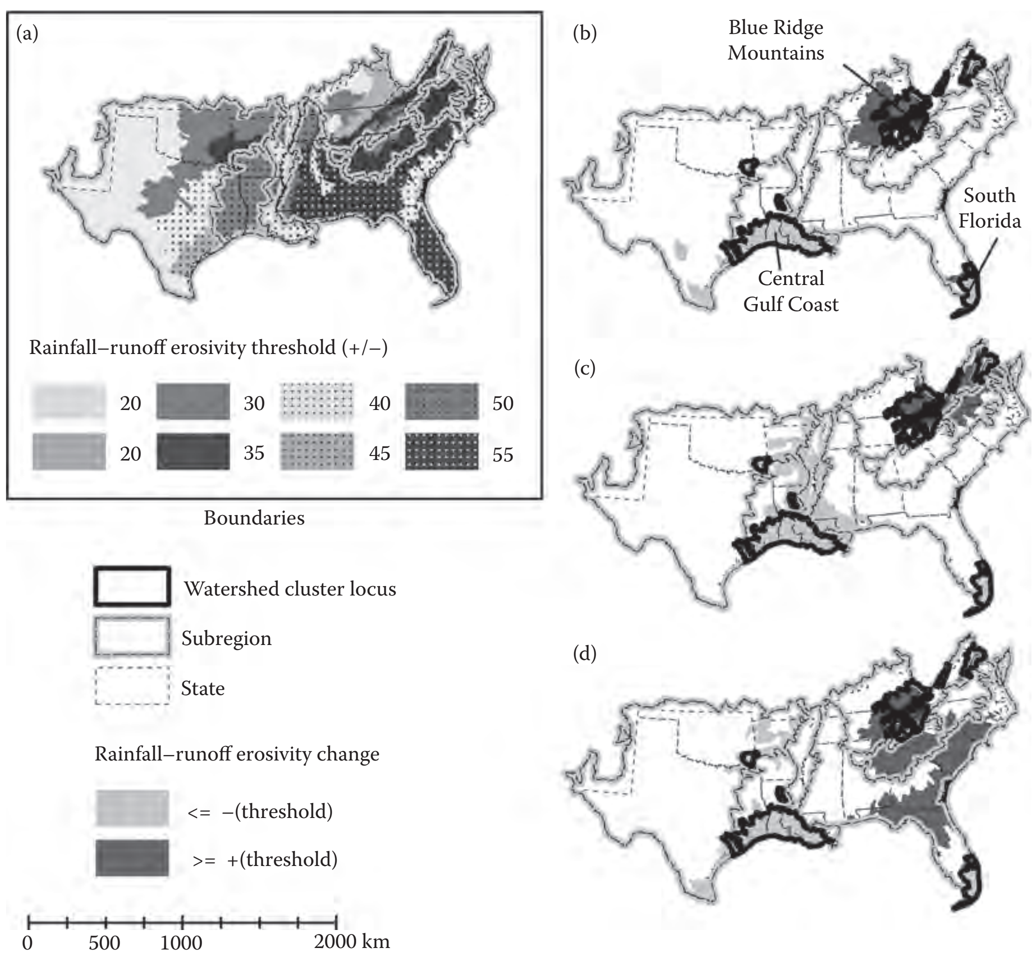


FIGURE 9.10 Changes in future rainfall/runoff erosivity factor for Southern U.S. watersheds based on the HadCM3 B2 climate change scenario, showing defined change thresholds for rainfall–runoff erosivity (a) and watersheds predicted to exceed the thresholds during the (b) 2011–2030, (c) 2031–2050, and (d) 2051–2070 periods. The HadCM3 B2 climate scenario for the Southern United States (McNulty et al. in press) combines the HadCM3 general circulation model with the B2 emissions storyline (representing lower population growth and energy use).

Thresholds for both positive and negative R-factor changes are used. Using a positive change threshold seems obvious because future surface erosion should increase if R-factor values increase. However, a decrease in R-factor values may also result in increased erosion. Using simulation modeling, Pruski and Nearing (2002b) found that erosion could increase when R-factor values either increased or decreased. This occurred under certain conditions because the reduced rainfall associated with decreased R-factor values also caused biomass to decrease and thereby reduced the erosion resisting characteristics of the vegetation.

Projected changes in R-factor: The R-factor changes from the baseline period (1961–1990) for each of the three future periods are discussed below. Hereinafter, only those watersheds that exceed either the positive or negative thresholds are considered.

The future change in R-factor values shows several important trends. Watersheds where the R-factor change is projected to exceed either the positive or negative threshold tend to cluster together (Figure 9.10). These clusters are geographically distinct with each having a locus exhibiting changes in all three future time periods, and a zone of adjoining watersheds where change is apparent in two of the three time periods. For discussion, these three clusters are referred to as the Central Gulf Coast, Blue Ridge Mountains, and South Florida. Each of the clusters shows a consistent but different change behavior. The Central Gulf Coast watersheds all exhibit notable decreases in R-factors, the Blue Ridge Mountain watersheds exhibit increases, and in South Florida both response types occur. The clusters differ in their expansion over time. The Blue Ridge Mountains cluster appears to steadily increase in geographic extent over time, while the extent of the Central Gulf Coast cluster appears to fluctuate somewhat over the study period. The South Florida cluster remains stable with little size change over time.

The geographic and temporal trends in R-factor changes are caused by the combined effect of future precipitation changes (Figure 9.11) and the thresholds defined for the soil areas (see Figure 9.10). Clearly, the clusters noted above correspond closely to those areas showing relatively large precipitation changes. The fact that both the Central Gulf Coast and South Florida clusters occur in areas where the thresholds are among the highest in the South suggests that these areas may be particularly susceptible to surface erosion in the future.

Central Gulf Coast cluster: Historically, the watersheds within or adjoining the Central Gulf Coast cluster all have relatively high R-factor values as demonstrated by the correspondingly high thresholds (30–55) defined for them (Figure 9.10). The primary locus of the Central Gulf Coast cluster is in southeastern Texas and southern Louisiana, where 27 contiguous watersheds exhibit notable R-factor decreases in all three time periods. An additional three watersheds (one in northern Louisiana and two in southeastern Oklahoma) are disconnected from the cluster, but also exhibit notable changes in all three periods. For the period ending in 2030, 46 watersheds have notable R-factor decreases, with all but a few occurring along the Gulf coast or slightly inland. By 2050, the affected area has extended up the Mississippi River and eastward along the Gulf of Mexico to include 102 watersheds. Almost all of Louisiana and most of Arkansas are involved in this period, but 8 watersheds along the coast and in interior southern Texas that previously exhibited notable decreases no longer do so. By 2070, the area affected contracts to only 56 watersheds with the Louisiana and Texas coastal areas continuing to be involved, as are a number of watersheds in Arkansas and eastern Oklahoma.

Over the entire 2011–2070 study period, the change in R-factor for the Central Gulf Coast cluster is most consistent in the Coastal Plain and Mid-South watersheds (Figure 9.12). Mean decreases in R-factors are consistently around 70 for Coastal Plain watersheds and just below 50 for watersheds in the Mid-South. The watersheds in the Mississippi Alluvial Valley are more variable with mean decreases between those of the Coastal Plain and Mid-South watersheds, but with a few actually showing decreases larger than those in Coastal Plain watersheds.

Blue Ridge Mountain cluster: The Blue Ridge Mountain cluster has two groupings of watersheds that exhibit consistent increases in R-factor for all three time periods. The larger locus of 15 watersheds occupies the eastern half of the Appalachian-Cumberland Highland where Virginia,

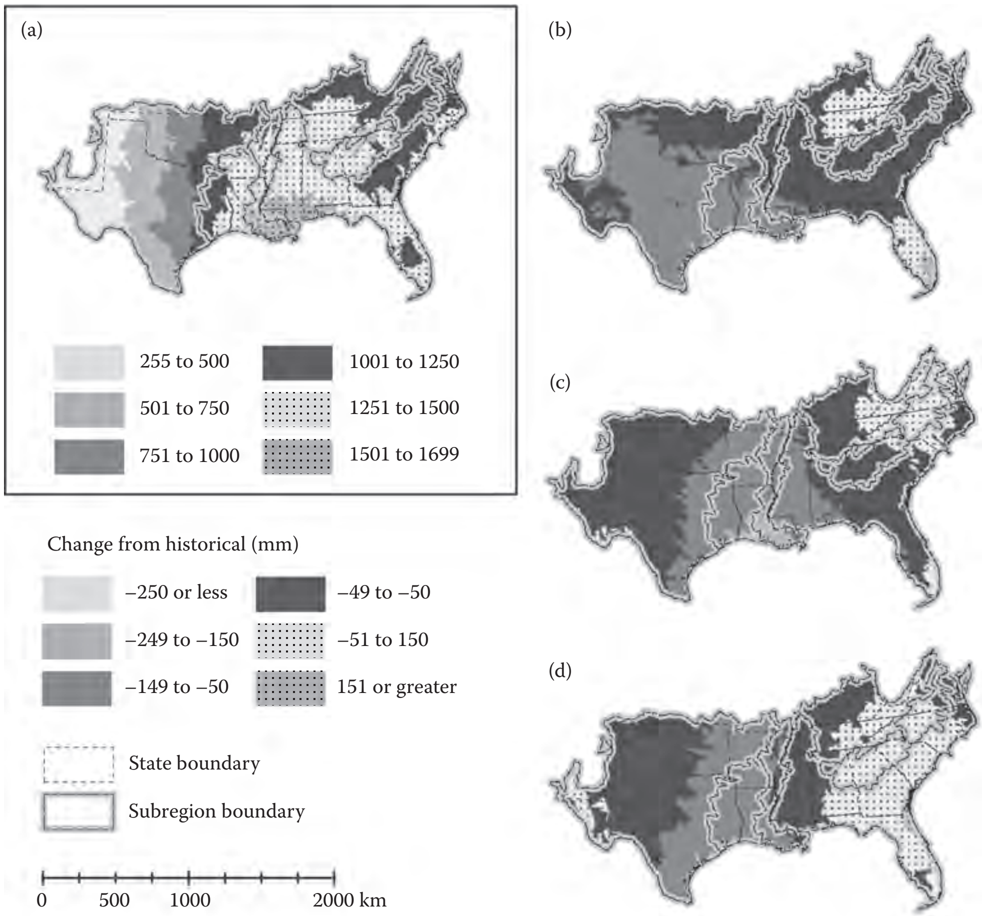


FIGURE 9.11 Future change in mean annual precipitation for Southern U.S. watersheds as shown by (a) the mean for the historical period (based on PRISM modeling), and predicted increase (+) or decrease (–) from the historical mean for the (b) 2011–2030, (c) 2031–2050, and (d) 2051–2070 periods based on the HadCM3 B2 climate change scenario. The HadCM3 B2 climate scenario for the Southern United States (McNulty et al. in press) combines the HadCM3 general circulation model with the B2 emissions storyline (representing lower population growth and energy use).

Kentucky, Tennessee, and North Carolina come together. The smaller locus consists of five watersheds that occur mostly within the northern Piedmont in Virginia (see Figure 9.10). Historically, watersheds in this core area have lower relative R-factor values (200–350), but projected increases in annual precipitation suggest a marked increase in future rainfall erosivity.

The number of affected watersheds increases steadily over time. By 2030, 51 watersheds show notable R-factor increases. By 2050, the area affected shifts somewhat eastward into the Piedmont and the total number increases to 63. By 2070, the affected area has spread southeast into the Coastal Plain and the number of affected watersheds doubles (126).

R-factor changes in affected watersheds in the Appalachian-Cumberland Highland and Piedmont are fairly consistent over time, with means between 35 and 45 (Figure 9.12). In contrast, R-factor changes are more variable over time for affected watersheds in the Coastal Plain, reflecting their shifting location within the Coastal Plain. In 2030, the single watershed that exceeds the threshold occurs on the coast of the Florida panhandle (although not discernible in Figure 9.10) where

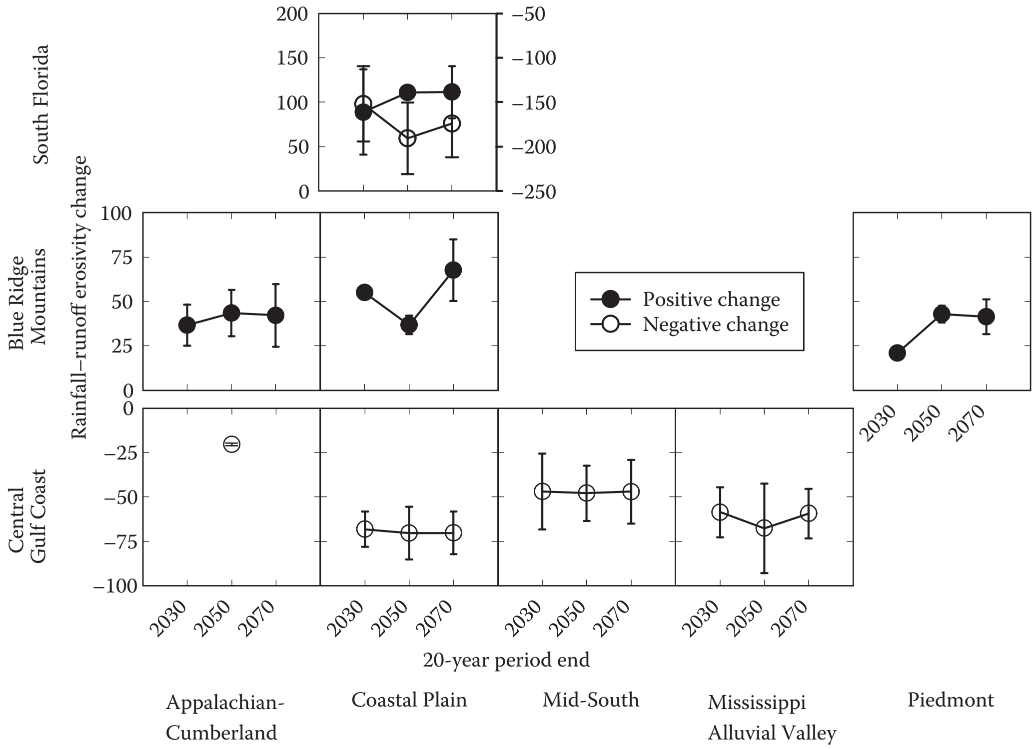


FIGURE 9.12 The mean change in rainfall–runoff erosivity factor for Southern U.S. watersheds by "geographic change cluster" (Central Gulf Coast, Blue Ridge Mountains, and South Florida) and subregion for three 20-year study periods.

historical R-factors and thresholds are relatively high. By 2050, the affected Coastal Plain watersheds are all located along the Chesapeake Bay, where historical R-factors are lower, and the mean R-factor change is now 37 (Figure 9.12). By 2070 the mean change has increased to 68 because the affected watersheds have shifted southward and R-factor changes are larger. The effect of R-factor increases on surface erosion within this cluster may be amplified by the area's steeper terrain, particularly in the Appalachian-Cumberland Highland.

South Florida cluster: The South Florida change cluster is unusual in that its watersheds exhibit both notable increases and decreases over a relatively small geographic area. The locus of the cluster consists of three watersheds on the eastern side of the southern Florida peninsula and illustrates the extreme contrasts projected to occur in the future (Figure 9.10). Two of the watersheds exhibit consistent R-factor decreases ranging from −122 to −219, whereas the adjoining watershed to the south (encompassing the Florida Keys) exhibits increases of 111–177. If not attributable to model error, these remarkable differences indicate tremendous climatic variability over short distances.

All three time periods reflect this marked geographic variability. By 2030, seven watersheds show notable changes with two decreasing and five increasing. By 2050, only the watersheds within the cluster locus are affected; by 2070 one additional watershed joins the increase total.

Throughout the study period, R-factor changes in the South are largest in South Florida. Watersheds exhibiting notable increases have mean values between 89 and 111 (Figure 9.12). Those showing notable decreases have means of −152 to −190.

Other considerations: Past research has noted that the response of surface erosion to R-factor changes may not be a one-to-one ratio. Erosion simulation studies using agricultural conditions and the Revised Universal Soil Loss equation showed that soil loss did not respond in equal proportion to changes in the R-factor; rather, the percent change in soil loss is somewhat less than the change in R-factor value (Pruski and Nearing 2002a; Renard and Ferreira 1993). Pruski and Nearing (2002a) attribute this to increased biomass production under wetter conditions that thereby increases the ability of vegetation to resist erosion. Renard and Ferreira (1993) found that temperature changes actually affected soil loss amounts more than rainfall because of the effect that temperature has on residue decomposition rates.

The way rainfall occurs can also affect erosion response. Pruski and Nearing (2002b) found that varying the number of days with rain, the rainfall intensity, and the season when rainfall occurred all affected erosion response, and sometimes resulted in a percent change in soil loss exceeding that of the R-factor. It should also be noted that the steeper slopes that often occur in forest lands could compound the effect of R-factor changes.

In summary, our modeling results show that climate change has the potential to produce marked changes in R-factor values in the South, that R-factor changes would be largely restricted to three geographic clusters (the Central Gulf Coast, Blue Ridge Mountains, and South Florida), and that R-factor changes are likely to persist out to 2070. Land managers working within areas covered by these clusters may well need to consider mitigation and adaptation measures in planning future land disturbing activities.

Salinity Intrusion

Saltwater intrusion into freshwater aquifers and drainage basins can threaten the biodiversity of freshwater tidal marshes and contamination of municipal, industrial, and agricultural water supplies (Bear et al. 1999). The balance between the hydrologic flow conditions within a coastal drainage basin and fluctuation in sea level governs the magnitude, duration, and frequency of salinity intrusion into coastal rivers. Future climate change is likely to alter these hydrologic balances on the East Coast of the United States by increasing air temperatures, changing regional precipitation regimes, and causing sea levels to rise.

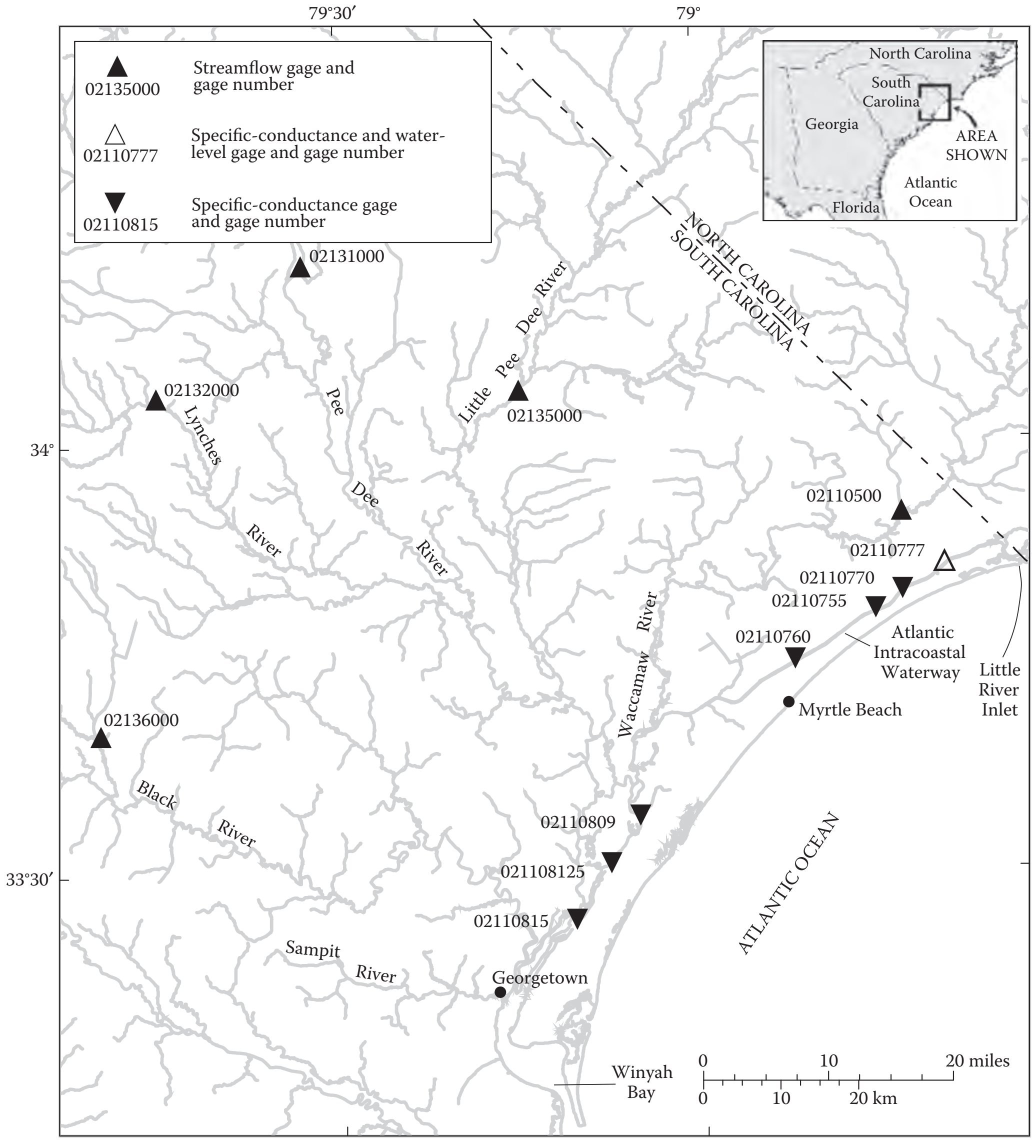
We examined saltwater intrusion at two municipal intakes—on the Atlantic Intracoastal Waterway and the Waccamaw River along the South Carolina Grand Strand near Myrtle Beach (Figure 9.13)—to illustrate how future sea level rise and a reduction in streamflows can potentially affect salinity intrusion, threatening municipal water supplies and the biodiversity of freshwater tidal marshes (Furlow et al. 2002).

Modeling methods and databases: An updated salinity intrusion model of the Pee Dee River Basin (Conrads and Roehl 2007) was used to evaluate the potential effects of climate change on salinity intrusion. The model was developed using data-mining techniques, including empirical modeling using multilayer perceptron (Rosenblatt 1958) artificial neural network (Jensen 1994) models.

The U.S. Geological Survey maintains a real-time stream-gauging network of recorders for streamflow (>50 years), water-level, and specific conductance (<25 years) in the Pee Dee and Waccamaw River Basins (Figure 9.13). During the past 25 years of data collection, the estuarine system has experienced various extreme conditions including large 24-h rainfalls, the passing of tropical systems and major offshore hurricanes, and record droughts.

The artificial neural network models that were developed used a subset of the Pee Dee/Waccamaw network. Model outputs are the specific conductance at seven coastal gauging stations using data from five upland streamflow recorders, tidal water levels at the northern end of the Intracoastal Waterway, and wind speed and direction from a coastal meteorological station. The models were validated using historical measurements of specific conductance at selected coastal stream gauging stations.

To simulate the effects of sea level rise, the mean coastal water levels were incremented by 0.5-foot segments to simulate sea level rise up to 3 feet. To simulate the effect of reduced streamflow to the coast, daily historical streamflows were reduced by increments of 5% up to 25%. Daily specific



Base from National Hydrography Dataset;
UTM projection, zone 17, datum NAD83

FIGURE 9.13 Location of study monitoring stations within the Pee Dee and Waccamaw River Basins and Atlantic Intracoastal Waterway in South Carolina.

conductance values were simulated for each incremental rise in sea level and each incremental reduction in streamflow from July 1995 to August 2009.

Findings: Results for the Pawleys Island stream gauge (station 021108125), just downstream from a municipal freshwater intake, were selected for this analysis (Figure 9.13). The model satisfactorily simulates the specific conductance in the 2000-μS/cm range and accurately simulates the high intrusion events in the fall of 2002 and 2008 that exceeded 10,000 μS/cm.

Operations at municipal water treatment plants become more difficult if specific conductance values for source water exceed 1000 to 2000 μS/cm. Figure 9.14 shows the number of days that specific conductance values exceeded thresholds of 1000, 2000, and 3000 μS/cm for the 14-year

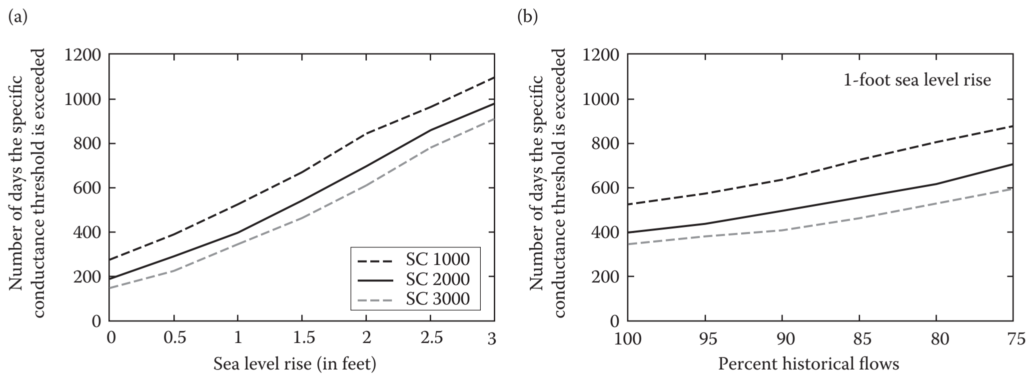


FIGURE 9.14 Number of days that specific conductance (SC) thresholds would be exceeded at Pawleys Island Station (gauge 021108125) in South Carolina for (a) sea level rise up to to 3 feet, and (b) with decreased historical flows concurrent with a 1-foot sea level rise.

simulation period. For example, daily specific conductance >2000 μS/cm historically occurred for almost 200 days (Figure 9.14), during which the municipal intake was unavailable. A one-foot sea level rise would double the number of days the municipal intake is unavailable to 400 days and a two-foot rise increases the unavailability to nearly two years (700 days).

Changes in precipitation patterns that are caused by changes in the climate have the potential to decrease streamflow to the coast. Salinity intrusion occurs during low streamflow periods; a decrease in streamflow combined with a sea level rise could increase the duration of salinity intrusion. For a specific conductance threshold of 2000 μS/cm, a one-foot sea level rise combined with a 10% decrease in historical streamflow would increase the days the municipal intake is unavailable by 25%, or an additional 100 days (Figure 9.14). A 25% reduction of low streamflow levels increases the incidences of intake unavailability to >700 days.

MANAGEMENT OPTIONS FOR MITIGATION AND ADAPTATION

Our analyses clearly show that climate change and its impacts on water quantity and quality have occurred in several subregions of the South. Although climate scenarios for the next several decades do not agree about the magnitude or direction of the expected changes for some variables (especially precipitation), they all point toward a climatic regime that the region has not experienced before (Milly et al. 2008). This would require a reevaluation of forest water resource management methods, even for those practices that have been successful in the past (Galloway 2011), because they were developed for previous climate characteristics. Existing historical records from individual forest monitoring sites may be too short to be useful in accurately forecasting the impacts of climate change, but the accumulated data from many sites provide a foundation for developing mitigation and adaptation methods in forest environments of the future. Computer simulation models provide valuable tools for identifying the potential risks and consequences of climate change and for helping land managers design response actions that minimize adverse impacts.

Various risks to natural ecosystems (Carlisle et al. 2010) and society can result from climate change impacts on water resources (Table 9.6). Innovative adaptation options are needed to reduce or adapt to the severe consequences of climate changes, such as water supply shortages, habitat loss, and increased wildfires. Best management practices (BMPs) that have been found to be most effective in reducing nonpoint source pollution will likely need to be enhanced and revisited so that they best reflect future hydrologic and management conditions. Integrated watershed management that enhances ecosystem resilience to climate disturbances and maintains ecosystem services—including climate moderation and mitigation—would improve adaptation efforts. Below, we discuss

TABLE 9.6
Potential Adaptation Options to Managing Hydrologic Impact and Risks from Climate Change

Hydrologic Impacts	Risks to Ecosystems and Society	Adaptation Options
Water supply stress increase	Water shortage; drying up of drinking wells; consequences to aquatic ecosystems, socioeconomics, and businesses	Maintain watershed health, thinning forests; reduce groundwater and surface water use for irrigation of croplands and lawns; enhance water conservation
Evapotranspiration increase	Hydrologic droughts; wildfires; insect, disease outbreaks	Use native tree species; reduce tree stocking
Increase of peakflow, stormflow volume, floods	Flooding; increased soil erosion and sedimentation	Reduce impervious areas; increase stormwater retention ponds; increase evapotranspiration by increasing forest coverage
Low flow decrease	Water quality degradation; fish habitat loss; reduced transportation capacity	Reduce off stream water withdrawal; adjust water outflow from reservoirs
Wetland hydroperiod change	Wildlife habitat loss; carbon dioxide emission	Plug ditches; adjust water outflow from reservoirs
Stream water temperature increase	Water quality degradation; loss of cold fish habitat	Maintain riparian buffers and shading
Soil erosion, sedimentation increase	Water quality degradation; siltation of reservoirs; increase of cost of water treatment	Enhance best management practices (BMPs) for forest roads; redesign riparian buffers; minimize direct discharge of runoff from roads to streams
Chemical loading increase	Water quality degradation; higher cost of water treatment	Maintain streamflow quantity; forest best management practices (BMPs)

management options and offer some recommendations to address specific water resource concerns related to climate change.

Gunn et al. (2009) recommend a toolbox approach with the three strategies of resistance, resilience, and response. Resistance strategy is a set of short-term approaches to address immediate threats and focuses on minimizing the impacts of disturbance that are exacerbated by climate change. Resilience strategy is both a short- and long-term approach to address the capacity of a forest stand or community to recover from a disturbance and return to a reference state. Response—the most costly strategy to facilitate the movement of species over time—requires acceptance of a level of uncertainty that will be uncomfortable for many forest managers and relies on close collaboration with forest ecologists, silviculturists, and other specialists.

Sea Level Rise and Hurricanes

Close proximity to the ocean makes some Coastal Plain ecosystems extremely vulnerable to hurricanes and sea level rise, both of which are linked to climate change and variability. Louisiana, Mississippi, Alabama, and Florida are projected to see significant increases in salinity levels associated with saltwater intrusion, as well as a degraded quality of the inflows. Forests in low-lying areas such as mangroves in Florida and cypress (David et al. 2010) are particularly vulnerable to sea level rise and increased air temperature. Other concerns are more wildfires and more areas where pine species are apt to be favored. More frequent, high-intensity summer rainfall may lead to downstream flooding in certain areas. Higher water temperatures and changes in freshwater delivery will likely alter estuarine stratification, residence time, and eutrophication. Occurrence of many stressors—such as pollution, harvesting, habitat destruction, invasive species, land and resource use, and extreme natural events—if concurrent would intensify hydrologic responses.

A first step might be to redesign forest road cross-drainage structures using projected high-intensity storms. Coastal areas are expected to see more land development and urbanization with continued population growth. Handling increased stormwater runoff from intense rainfall on flat lands and poorly drained soils requires special designs of urban BMPs. Protection of natural wetlands would maintain their hydrologic functions and buffer disturbances at the landscape level. Resilience and vulnerability evaluation of artificially drained managed forests will be more accurate if they incorporate projected future hydrology.

Although increases of sea level rise and decreases in streamflows show substantial effects that would have operational consequences for municipal water-treatment plants in coastal areas, the climate scenarios that were used for our projections would help water resource managers plan for mitigation efforts to minimize the effect of increased salinity of source water. Mitigation efforts may include timing of withdrawals during outgoing tides, increased storage of untreated runoff, timing larger releases of regulated flows so that the saltwater–freshwater interface moves downstream, and blending higher conductance surface water with lower conductance water from an alternative source such as groundwater.

Extreme Precipitation Events and Low Flows

Our results suggest that forest management activities that fall short of changing forest composition or stand structure after a clear-cut will not substantially alter streamflow responses to extreme precipitation events. Although often statistically significant, differences among annual streamflow responses were small under mean climate conditions versus observed or projected extremes in annual drought or high annual precipitation, and would result in changes that range from 7% increases to 5% decreases. These results suggest a limited capacity to create watershed conditions more resilient to extreme annual precipitation than native hardwood forests managed using traditional forestry practices that rely on natural regeneration. In contrast, management activities that converted deciduous hardwoods to pine monocultures substantially lowered streamflow in times of extreme annual precipitation; such actions may reduce flood risk but also exacerbate drought. This argues for careful consideration of the tradeoff between managing forests for opposite extremes during contingency land use planning.

Landowners and policy makers both look to forests as a means to offset climate change effects (Pacala and Sokolow 2004) and to forest managers to create ecosystems that are more resilient to extremes and changing climate (Baron et al. 2009). We have shown that in Appalachian-Cumberland Highland watersheds, changing forest cover through species conversions affects streamflow and thus downstream water supply in ways other than what would be expected from unmanaged forests; however, forest cover change also affects many other ecosystem services, including carbon sequestration (Liao et al. 2010). Forests cannot be managed solely for water resources without affecting carbon sequestration, and vice versa (Jackson et al. 2005). Whether increasing forest cover or converting deciduous to pine forest cover will mitigate climate change is uncertain (Jackson et al. 2008), but our study shows the potential of forest management to mitigate against the hydrologic impacts of extreme annual precipitation associated with climate change.

In light of predicted decreases in low flows, water resource conservation and hydrological system protection will become vitally important for keeping the minimum streamflow required to prevent significant harm to water resources. Management practices that help to reduce water use, reclaim wastewater, and enhance infiltration and groundwater discharge to streams may become important tools in mitigating climate change impacts on low flows.

Water Quality

Climate change has the potential to degrade water quality. Because our research indicates that climate change will likely increase stream temperatures, the most effective mitigation options would be those that focus on decreasing stream temperature through management, particularly in the summer months (Swift 1973, 1982; Wooldridge and Stern 1979). Maintaining or increasing shading from solar radiation through riparian buffer retention, conservation, or restoration has been shown to decrease stream temperatures (Burton and Likens 1973; Kaushal et al. 2010; Peterson and Kwak 1999; Swift 1973, 1982; Swift and Messer 1971).

Wide BMP usage has helped control nonpoint source pollution, including sediment loading to water bodies. To mitigate climate change impacts, BMP adjustments may be necessary to address anticipated increases in storm flow, soil erosion, and sediment. If climate change alters rainfall patterns as predicted, the size of large runoff events (such as the 20-year storm) will likely increase; to put it another way, the current 20-year storm will occur more frequently. Storm intensity in general is predicted to increase, requiring larger capacity in forest road drainage structures and BMP improvements to reduce the effects of increased storm water discharge. Possibilities for BMP innovations to handle future climate conditions in forested watersheds are:

Broad-based dips: Current standards are based on road grade. Compensating for increased runoff will require adjustments to the relationship between road grade and distance from one dip to the next; for a given road grade, the distance from one broad-based dip to the next would need to be shorter with increased rainfall amount/intensity. The alternative would be to modify design criteria of the dip itself to allow increased volumes of storm flow.

Culvert and other cross-drainage structure size: Increasing the size of culverts, causeways for forest roads and drop structures, and gully check dams for a given drainage area may be required. An alternative would be to install hooded inlets that increase culvert capacity. In situations where road elevation is limiting, hooded inlets would decrease the need for additional fill at channel crossings.

Riparian buffers: Increasing the use of riparian buffers would be most helpful. In addition, increased buffer widths may be needed to improve the capacity of undisturbed stream side zones to absorb potential overland flow resulting from more frequent, extreme storm events. The Effective Functional Width is a tool for illustrating how increased overland flow during intense storms can decrease the capacity of a riparian buffer of a given width to protect water quality. Figure 9.15 depicts

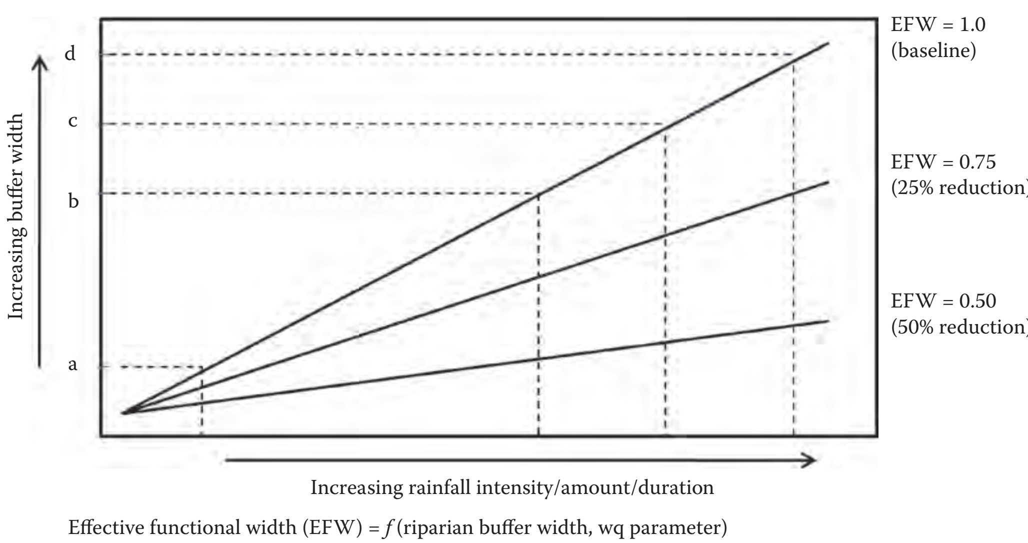


FIGURE 9.15 Theoretical model for maintaining the effectiveness of a stream buffer for reducing surface water quality degradation; EFW is the effective buffer width.

the decision space for buffer width determination. Under undisturbed conditions a given parameter has a value of 1.0, indicating an inherent capacity of the riparian zone to absorb materials being transported downslope as a result of prevailing levels of erosion and decomposition. After disturbance (such as a timber harvest), the condition of the parameter deteriorates by some factor depending on its sensitivity to the effects of the land-disturbing activity. In order to maintain an acceptable level or condition of a given parameter, riparian buffer width adjustments would be needed. If a 25% or even 50% reduction in the condition of the parameter is tolerable, and is itself the most limiting or most sensitive parameter in that system, then narrower buffers may be assigned. Further, buffer width is a function of some measure of rainfall intensity so that for a given maximum expected intensity/amount/duration, buffer width may be adjusted to maintain an acceptable level of parameter condition.

Storm water management: Often, the management of forested watersheds is intended to regulate water yield, a paradigm that may need reorienting toward management of peak flows to face the challenges of climate change. Road design and construction practices may be needed that more effectively disconnect the road system from the stream network; this would slow flow routing during extreme events and increase belowground storage. Storm water management that maximizes opportunities for infiltration would reduce the impact of extreme storm events on peak flows. Further, predicted increases in peak storm water discharge would place increased demand on water treatment facilities, often resulting in discharge of untreated effluent. Improvements such as water gardens, porous parking lots, sediment basins, and sustainable urban drainage systems that incorporate urban forestry practices would encourage infiltration, reduce needed treatment facility capacity, and ultimately reduce the risk of degraded surface water quality.

Other human influences not related to forest management may help mitigate the impact of climate change on stream water quantity and quality. For example, deep-water releases from reservoirs can have a cooling effect similar to groundwater, and municipal wastewater from a deep underground pipe can cool streams in the summer and increase baseflow (Bogan et al. 2003). Wastewater discharge as a means to reduce stream temperatures should be used with caution, as it may also have a warming effect in winter months. Re-using treated wastewater helps to reduce higher-temperature effluent volume entering streams (Kinouchi et al. 2007), and decreasing water withdrawals from streams through water conservation may help maintain a more stable temperature (Webb and Nobilis 1995). Increasing shaded stormwater retention wetlands, increasing urban tree canopy over any runoff surface, and reducing impervious surface area (Peterson and Kwak 1999) can all help decrease stream temperatures, particularly in urban areas or areas where the riparian vegetation has been degraded.

Moderating Climate and Mitigating Impacts through Forest Restoration and Afforestation

Natural and managed forests provide the best water quality among land uses (Lockaby et al. in press). In addition, forests can also modulate regional climate by controlling energy and water transfers between the atmosphere and forested land-surface (Chen et al. 2012; Liu 2011; Liu et al. 2008; Sun et al. 2010). Past agriculture-to-forest land conversion in the South may have led to significant changes in land-surface energy and water processes, including an increase in solar radiation absorbed by the land surface caused by the lower light reflection of forests (Liu 2011). An increase in absorbed radiation provides more energy for heat transfer from the land surface to the atmosphere through sensible and latent heat fluxes and increasing evapotranspiration, which ultimately reduce runoff and water yield. These changes in land-surface processes can further change atmospheric dynamics and hydrologic processes, including precipitation and soil moisture. Some simulation studies have indicated that precipitation is likely to increase with large-scale afforestation (Liu 2011).

Forest restoration and afforestation are expected to play an important role in mitigating the impacts of climate change. A plan to plant over 7 million ha of new trees by 2020 to replace pasture and farming lands in the South, as well as in the Great Lake states and the Corn Belt states (Watson 2009) is even larger than the one carried out by the Civilian Conservation Corps during the Great

Depression, which planted 3 billion trees from 1933 to 1942. Although its primary purpose would be to mitigate the greenhouse effect by increasing the capacity of forests to remove atmospheric CO_2, this effort would also be useful in mitigating the impacts of the greenhouse effect on forest water conditions.

Mitigating the hydrologic impacts of climate change through forest restoration and afforestation is a complex issue. Achieving the mitigation goal requires consideration of many factors when making restoration or afforestation plans. First, approaches for changing forest hydrologic conditions in an afforested area can be different or even opposite of those for surrounding landscapes. Differences can even exist within the afforested area, depending on local atmospheric patterns and physiography. Second, vegetation types are expected to change in the future as a result of climate change. For example, some deciduous forests in the region's upper latitudes are projected to change to savanna communities, and some coastal mixed forests to conifer woodlands (Neilson et al. 2005). Ideally, future afforestation would be done in the places where forests are projected to be most degraded by climate change. Finally, one mitigation tool could have mixed outcomes. For example, afforestation may increase soil water storage by increasing precipitation and decreasing runoff, but may also reduce it by increasing evapotranspiration.

SUMMARY OF REGION-WIDE RESPONSES

Climate changes (increases in rainfall variability and air temperature) are happening across the South and are predicted to continue in the near future.

- Climate warming would result in increased water loss through evapotranspiration through increased evaporative potential and plant species shift, and thus, under a similar precipitation regime, climate warming could cause decreased total streamflow, low flow rates, and regional water supply.
- Decreased precipitation in the western Coastal Plain and Mid-South would have serious impacts on water availability and aquatic habitat.
- Water supply stress is projected to increase significantly by 2050, the result of hydrologic alterations caused by climate change and increased water use by key economic sectors such as domestic water supply, irrigation agriculture, and power plants; water stress will likely be more severe in the summer season.
- Runoff and soil erosion are projected to increase in some areas, the result of changes in rainfall that either increase rainfall erosivity or decrease vegetative cover protection.
- Water temperature is projected to increase with air temperature rise, resulting in possible impacts on coldwater fish habitat in the Appalachian-Cumberland Highland.
- Salinity intrusion in coastal fresh water systems is likely to increase in response to sea level rise and decreased fresh-water inputs from uplands—both consequences of climate change.
- Forest management has the potential to mitigate or reduce damage from hydrologic extremes and degraded water quality caused by climate change, but practices must be applied judiciously to avoid unintended consequences.
- Existing BMPs will need adjustments and enhancement to increase watershed resilience to the likely adverse impacts of climate change on water quantity and quality.

CASE STUDIES

The South has diverse physiographic and socioeconomic characteristics, and thus water resource distribution, water use patterns, and response to climate change vary across the region. This section uses four case studies to provide a closer look at some of the water resource issues facing the Coastal Plain, Mississippi Alluvial Valley, Mid-South, and Appalachian-Cumberland Highland.

Booming coastal development and climate change are two major environmental concerns in the Coastal Plain (Williams et al. 2012). Much of the South's lower Coastal Plain (along the Gulf of Mexico and the southeastern coast of the Atlantic Ocean) consists of natural and managed forests, depressional wetlands, pine flatwoods, riparian buffers, and bottomland hardwoods on brackish waters (Lockaby et al. in press). Research from Santee Experimental Forest in South Carolina and two sites in North Carolina is used to examine how climate change is affecting lowland forests in the Coastal Plain.

The rich soils and relatively flat landscape of the Mississippi Alluvial Valley have long supported an economy dominated by agriculture and forestry. However, in recent years concerns are growing over future water supplies as water demands by agriculture increase. Using data from the Yazoo River basin in Mississippi, we examine how climate change may affect the amount and timing of water supplies in this subregion.

The Appalachian-Cumberland Highland has the highest concentrations of interior forest in the South (Wear and Gries 2002) and the most rugged terrain. Most climate scenarios disagree on whether precipitation amounts will increase or decrease for this subregion, but agree that an increase in frequency of precipitation extremes will occur. Research from the Coweeta Hydrologic Laboratory in North Carolina is used to investigate how typical forest management systems might affect peak streamflow amounts under climate change scenarios in forests of the Appalachian-Cumberland Highland.

Some of the largest and most continuous forest areas in the Mid-South occur in the Ouachita Mountains. An important concern of forest managers here and elsewhere in the Mid-South is how climate change may affect surface erosion and sediment delivery to streams and water bodies. Using simulation modeling, we examine how sediment risks would change in the Ouachita National Forest of Arkansas and Oklahoma under future climate conditions.

Case Study 1: Coastal Plain

The forested landscapes of the lower Coastal Plain are characterized by poorly drained soils with near-surface water tables and low topographic relief. When managed properly, these lands are highly productive for agriculture and commercial forestry (Amatya and Skaggs 2001). Because of their close proximity to sensitive estuarine ecosystems, these landscapes are vulnerable to large freshwater runoff from upstream, tropical storms, tidal surges, and saltwater intrusion from rising sea levels.

Flow generation within forested wetland watersheds may reflect a mechanism wherein saturation of the upper soil layer and forest floor produces surface runoff and rapid lateral transfers within this highly conductive layer (La Torre Torres et al. 2011; Sun et al. 2000b). Rapid rise of shallow water tables during rainfall events is common in the Coastal Plain (Amatya et al. 1996; Sun et al. 2000b; Williams 1978; Young and Klawitter 1968), and unlike upland watersheds, usually dominates hydrologic responses. Total flow depends largely on the dynamics of the water table (hydroperiod), which are driven by rainfall and evapotranspiration (Amatya et al. 1996; Amatya and Skaggs 2011; La Torre Torres et al. 2011; Lu et al. 2009; Pyzoha et al. 2007). Because of high evapotranspiration in the lower Coastal Plain, even third-order streams can lose all surface flow in normal dry periods (Amatya and Radecki-Pawlik 2007; Amatya et al. 2009; Dai et al. 2010a; Williams et al. 2012). In contrast, flooding problems during the winter wet periods are not uncommon within the Coastal Plain (Amatya et al. 2006a; Sheridan 2002; Young and Klawitter 1968). Summer droughts in the Coastal Plain are also on the rise (Karl et al. 2009).

Several general studies have been conducted to examine impacts of potential climate change and variability on stream and drainage outflows, evapotranspiration, and water table depths on the coasts of the lower Coastal Plain (Amatya et al. 2006b; Cruise et al. 1999; Dai et al. 2010b, 2011a,b; Hatch et al. 1999; Lu et al. 2009; Sun et al. 2000b), with findings similar to those noted in earlier sections. We use data from three research areas to determine if climate change impacts have already occurred in the lower Coastal Plain or are likely to occur in the future. They show that (1) the lower

Coastal Plain has seen an increase in air temperature with minimum temperatures increasing more than the maximum; however, despite changes in temperature, changes in annual streamflows and water table depths in one watershed were not observed; (2) increased frequency of large storms is likely to impact forest plant communities and increase flood occurrence; and (3) increased spring and summer droughts will likely make forest vegetation vulnerable to stresses caused by high evapotranspiration demands.

Santee Experimental Forest: The U.S. Forest Service established the Santee Experimental Forest within the Francis Marion National Forest (about 55 km northwest of Charleston, South Carolina) to conduct scientific research on forests and water in a Coastal Plain setting (Amatya and Trettin 2007). This site is characteristic of the subtropical area of the Atlantic coast with long and hot summers followed by short, warm, and humid winters. The mean annual temperature (1946–2007) is 18.5°C, and the mean annual precipitation is 1370 mm. In the past 60 years, extreme temperatures have exceeded 40°C in summers and fallen below –14°C in winters.

Dai et al. (2011b) reported that global warming caused a rapid (0.19°C per decade) increase in surface air temperature during the last 63 years at the Santee (Figure 9.16). The increase in mean annual temperature was slow or within normal variability from 1946 to 1969 but accelerated after 1969. The increase rate of the mean daily minimum temperature (0.26°C per decade) was twice as much as that of the mean daily maximum temperature (0.13°C per decade). The annual minimum temperature increased at a rate of 0.38°C per decade in the last 63 years ($p < 0.05$), accompanied by a large fluctuation. However, there was a downward trend in the annual maximum of about 0.19°C per decade ($p < 0.05$). These results indicate that the current warming trend has had a larger effect on lower temperatures than on higher ones in this area (Dai et al. 2011b).

The temperature increase rate at the Santee (0.19°C per decade) in the last 63-year period is substantially higher than the global mean increase rate of 0.07°C per decade during the twentieth century (Intergovernmental Panel on Climate Change 2001), about 0.06°C per decade higher than the global rate for land in the same time period (Hansen et al. 2010), and approximately the same as the global mean of 0.2°C per decade since 1976 (Hansen et al. 2006; Trenberth et al. 2007). The Santee results indicate that warming in this area became apparent six years prior to it becoming discernible globally (Hansen et al. 2006).

The year-to-year variability of precipitation in the Santee was large over the last 63 years. Annual precipitation ranged from 835 to 2026 mm, with a mean of 1370 mm (Dai et al. 2011b). Although the variations in annual precipitation exhibited an upward trend from 1946 to 2007, the increase

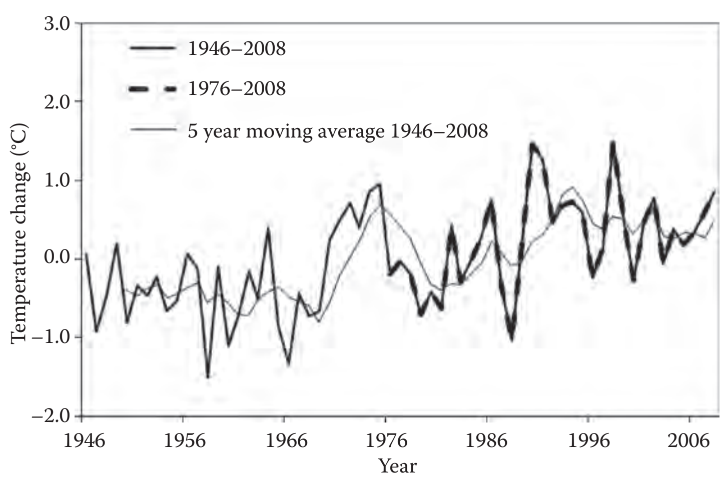


FIGURE 9.16 Temperature change from 1946 to 2008 at the Santee Experimental Forest on the South Carolina coast.

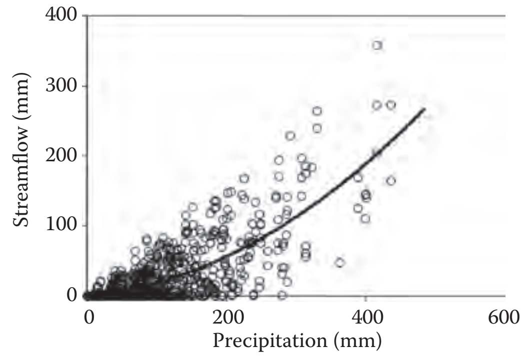


FIGURE 9.17 Relationship of monthly streamflow to precipitation from 1950 to 2000 at the Santee Experimental Forest on the South Carolina coast.

rate was not statically significant ($p > 0.1$). This upward trend was probably a result of a three-year drought period in the 1950s or from a multi-decade precipitation fluctuation. The mean number of large storm event (>50 mm) increased from 4.4 times per year from 1946 to 1981 to 5.7 times per year from 1982 to 2008. However, there was no significant upward trend in the last 63 years as a whole. This suggests that an increase in storm frequency may still be developing in this area.

Streamflow within the low-gradient forest watersheds largely depends on precipitation and evapotranspiration. As a result, first-order streams can be without flow during summer with high evapotranspiration demand and may flood during wet winters or summers that see large events like tropical storms. A scatter plot of monthly streamflows from all first- and second-order watersheds with monthly rainfall at Santee shows a significant ($p < 0.01$) relationship (Figure 9.17) showing the strong dependence of streamflow on precipitation. If storm intensity increases because of climate change, the frequency of high streamflows that produce flooding is likely to increase as well.

Vegetation damage from Hurricane Hugo caused streamflow impacts that continued for several years following the disastrous 1989 storm (Figure 9.18). The observed streamflow within the 14-year period after the hurricane is above the trend line, although mean annual precipitation was about 45 mm lower than the mean in the observation period (1965 to 2007). The high streamflow

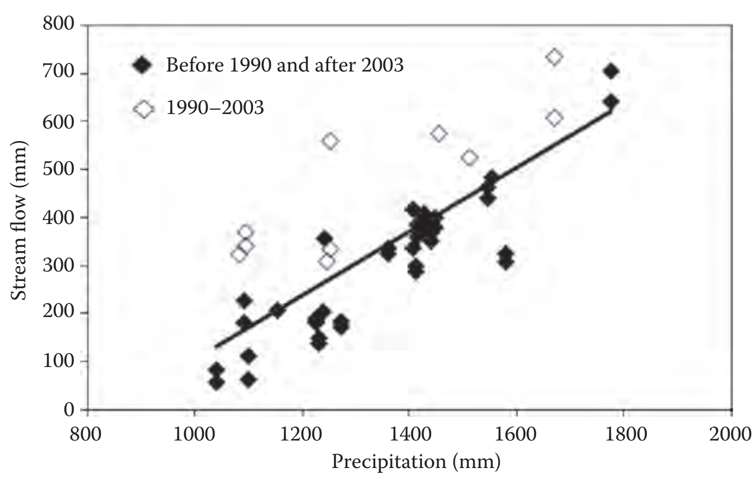


FIGURE 9.18 Impact of annual precipitation on annual streamflow on watershed 77 and watershed 80 from 1990 and 2003 at the Santee Experimental Forest on the South Carolina coast. The blank diamonds represent the relationship between precipitation and streamflow between 1990 and 2003 after Hurricane Hugo in 1989.

rate in the years following the hurricane was caused by destruction of over 80% of the canopy at this site (Hook et al. 1991), which undoubtedly caused a substantial decrease in the evapotranspiration demands of plants. These results are consistent with those reported by Shelby et al. (2005) for sites in the North Carolina Coastal Plain. However, the flow recovered about 10 years after the hurricane as the vegetation regenerated on the North Carolina watersheds (Williams et al. 2012).

A significant decrease ($p < 0.02$) in water table depth occurred from 1964 to 1993 because of annual precipitation increases (Figure 9.19). High water table level in this area in summers and autumns is mostly related to high precipitation (Amatya and Skaggs 2011; Amatya et al. 2009; Sun et al. 2000a).

The water table level was high throughout most of the winter–spring (December–February) seasons (Dai et al. 2011b) consistent with other studies by Amatya and Skaggs (2011). However, precipitation in these seasons was much lower than that in summers. The high water table level in winters or early springs is primarily caused by a low demand in evapotranspiration, indicating that evapotranspiration is one of the key factors that influence groundwater depth in these first-order watersheds (Amatya and Skaggs 2011; Amatya et al. 2009). These results indicate that if evapotranspiration increases because of climate change, water table depths may also increase in the Santee watersheds.

Williams et al. (2012) examined how Hurricane Hugo affected streamflow using historical streamflow data from a control (watershed 80, a first-order, forested watershed within the Santee, not harvested) and a treatment (watershed 77, salvage harvest after damage) watershed. They found that the watershed 80 had lower flow than watershed 77 during the pre-Hugo period, a relationship that reversed in 1993. The reversal continued for nearly 10 years, after which the streamflow of watershed 77 returned to near reference level. One explanation may be that increased evapotranspiration by the regenerating vegetation caused the lower streamflow (Amatya et al. 2006a; Williams et al. 2012).

To predict likely hydrologic response to future climate change, we applied a physically based distributed wetland hydrological model, MIKE SHE (Danish Hydraulic Institute 2005; Lu et al. 2009), to watershed 80 (Dai et al. 2010a). The MIKE SHE model links the hydrology with forest vegetation through the leaf-area index, rooting depth, and canopy storage capacity. Modeling results (Figure 9.20) suggest that the annual mean streamflow would increase or decrease by 2.4% with a 1% increase or decrease in precipitation, and decrease with an increase in air temperature. A quadratic polynomial relationship (Dai et al. 2011a) described the relationship between changes in groundwater table depth and precipitation (Figure 9.19). Both the mean annual water table depth

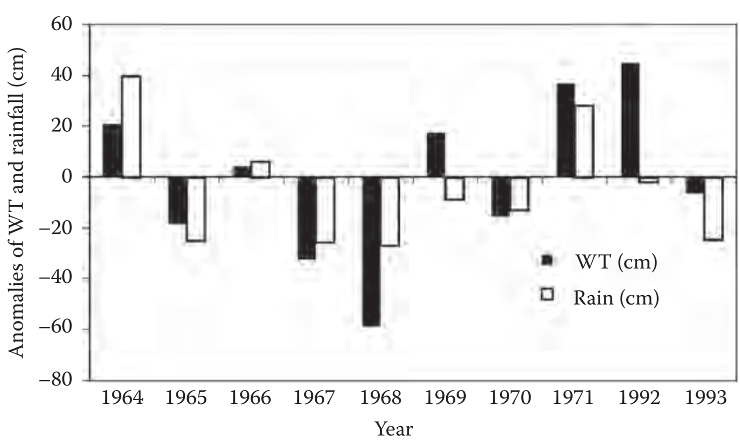


FIGURE 9.19 Impact of annual precipitation on annual mean water table on watershed 77 at the Santee Experimental Forest on the South Carolina coast (Dai et al. 2011b). WT is annual mean water table depth; rain is annual precipitation.

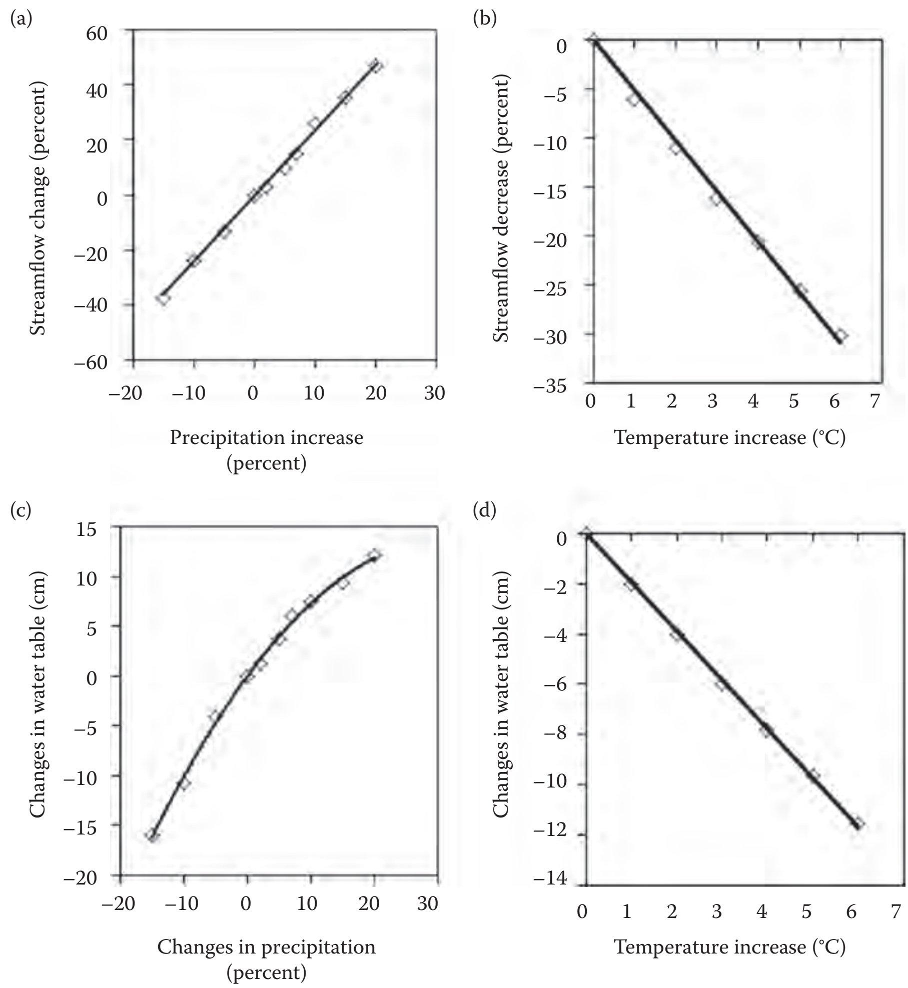


FIGURE 9.20 Simulated sensitivity of hydrologic responses to potential climate change at watershed 80 at the Santee Experimental Forest on the South Carolina coast: (a) streamflow response to precipitation, (b) water table response to precipitation, (c) streamflow response to air temperature, and (d) water table response to air temperature.

and mean annual streamflow decreased with an increase in temperature within the range of 0–6°C (Figure 9.20).

Carteret site: The Carteret site is on a typical Coastal Plain landscape in Carteret County in North Carolina and is owned by the Weyerhaeuser Company. The research site, established in late 1987, consists of three artificially drained experimental watersheds (D1, D2, and D3), each about 25 ha, covered with loblolly pine plantations. Details of all the hydrometeorological measurements and forest hydrologic studies at the site have been documented elsewhere (Amatya and Skaggs 2011; Amatya et al. 1996, 2000, 2003, 2006a, 2006b; Tian et al. 2011).

Annual rainfall ranged from 852 mm in 2001 to 2308 mm in 2003, with the 21-year mean of 1517 mm, which was 8% higher than the 50-year (1951–2000) long-term mean of 1390 mm observed at the nearby Morehead City weather station. Similar to the Santee, no statistically significant change ($\alpha = 0.05$) was apparent in the 21-year rainfall series at the Carteret site. January showed the largest decrease followed by October and December. Hurricanes occurred in 7 of the 21 years, mostly in September. Consequently, the mean rainfall of 552 mm from July to

September was significantly higher than other seasons, accounting for a third of the mean annual rainfall.

Although the 0.44°C rise in observed mean annual temperature during the entire 21-year period was not statistically significant ($\alpha = 0.05$), the 0.85°C rise from 2000 to 2008 was highly significant (Figure 9.21) and is consistent with other recent studies (Chapman and Davis 2010). The mean annual increase in net radiation (in evaporation equivalent energy units) of 430 mm from 1988 to 2008 was also significant. This radiation increase produced an estimated 270 mm increase in the annual Penman–Monteith potential evapotranspiration for a grass reference during the same period (Figure 9.21). The annual runoff coefficient, defined as the percent of rainfall occurring as streamflow, varied from 5% in 2001 to as large as 56% in 2003, the wettest year. The mean annual runoff coefficient of 32% was somewhat higher than the published data for similar conditions. Annual streamflow did not change during the 21-year period. Similarly, the 21-year annual mean water table

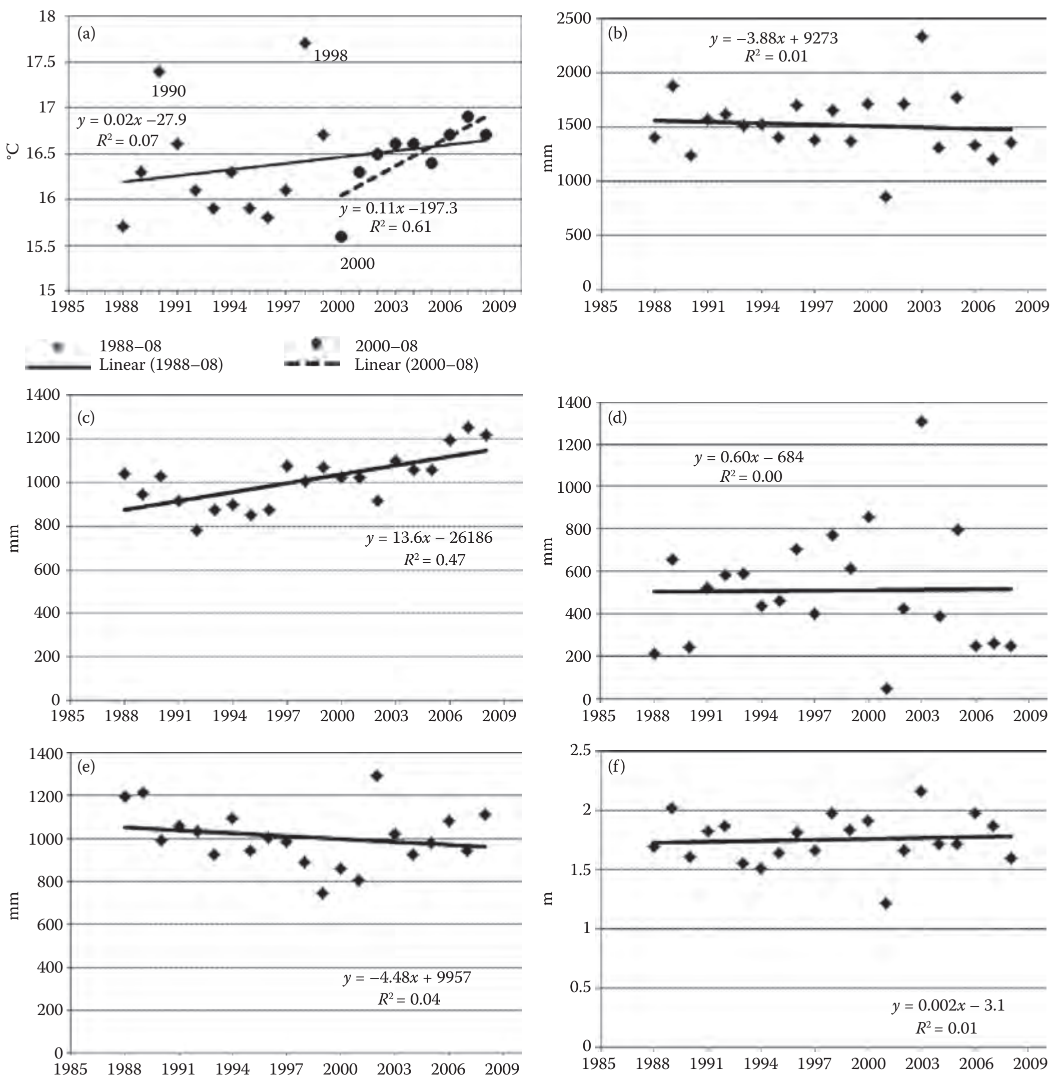


FIGURE 9.21 Annual trends in 21-year hydrologic variables at the Carteret study site in North Carolina: (a) temperature, (b) potential evapotranspiration, (c) evapotranspiration, (d) rainfall, (e) outflow, and (f) water table elevation. (Adapted from Dai, Z. et al. 2011a. *Atmosphere* 2: 330–357, doi:10.3390/atmos2030330.)

depth (96 cm) did not change despite the pine trees growing from 14 to 35 years stand age during that period and increases in temperature and potential evapotranspiration (Figure 9.21). Field data suggest the site is not water-limited except in extreme dry years (Amatya and Skaggs 2011; Sun et al. 2002). The combination of precipitation and air temperature rise appears to dictate future hydrologic change for this study site.

Parker Tract site: The Parker Track study site near Plymouth, North Carolina was established in late 1995 by Weyerhaeuser Company and North Carolina State University (Amatya et al. 2003). The landscape has a flat topography and poorly drained mineral or organic soils. Vegetation ranges from second-growth mixed hardwood and pine (*Pinus* spp.) forest to loblolly pine (*P. taeda*) plantations of various stand ages. Intensive hydrometeorological measurements at this site were conducted from 1996 to 2004 to support a variety of watershed studies (Amatya et al. 2003, 2004, 2006a, 2006b; Appelboom et al. 2008; Fernandez et al. 2006, 2007). Since 2005 studies have been shifted to understand how the coupling of management and climate disturbance affects forest ecosystem carbon and water balances (Noormets et al. 2010; Sun et al. 2010).

A simulation study was conducted to evaluate the potential effects of climate change on the hydrology of a 2950-ha managed pine forest at the Parker Tract site (Amatya et al. 2006b). The DRAINWAT model (Amatya et al. 1997) was first validated with a 5-year (1996–2000) data series from the study site (Amatya et al. 2004) and then run for the 1951-to-2000 period using historical weather data from Plymouth to determine the long-term hydrology. Separate simulations were conducted with 2001-to-2025 climate datasets projected by two General Circulation Models: CGC1 and the HadCM2 (Kittel et al. 1997). Simulation results (Figure 9.22) indicated that the CGC1 model yielded a significantly ($p < 0.0001$) lower mean total outflow (167 mm) than historical climate data or the HadCM2 model (380 mm). The decrease in outflow resulted from the drier conditions predicted by the CGC1 model relative to those predicted using the historical data. The 5% increase in rainfall using the HadCM2 model appeared to have no effect on runoff but may increase the evapotranspiration of this pine forest. The drier climate scenario (CGC1) did increase forest evapotranspiration compared to the historical data and was similar to predictions by the wetter (HadCM2) climate scenario (Figure 9.22). The water table depths based on the CGC1 model were deeper compared to those based on the HadCM2 model. However, the deeper annual water table depths predicted using both climate scenarios did not reduce the water table depth to the extent that evapotranspiration was limited by soil water depletion in the root zone. We concluded that both climate scenarios had very little or no impact on evapotranspiration, indicating the temperature increase has less of an effect on the soil moisture than does increased rainfall.

Case Study 2: Mississippi Alluvial Valley

Located within the Mississippi Alluvial Valley, the Yazoo is the largest river basin in Mississippi, with a total drainage area of 34,600 km^2. The goal of this study was to estimate the potential impact of future climate change on hydrologic characteristics in the Yazoo River Basin using an integrated hydrologic simulation model. This study developed a site-specific model for the Yazoo based on watershed, meteorological, and hydrological conditions. The calibrated and validated model was applied to predict the potential impact of future climate changes upon streamflow under multiple climate change scenarios.

Study watershed and data acquisition: The Yazoo basin is separated equally into two distinct topographic areas: the Bluff Hills and the Mississippi Alluvial Delta (Guedon and Thomas 2004; Mississippi Department of Environmental Quality 2008; Shields et al. 2008). The Bluff Hills area is a dissected, hilly, upland area where streams originate from a mixture of pine (*Pinus* spp.), pine-hardwood, and oak-hickory (*Quercus* spp., *Carya* spp.) forests, and pastures. The residual and alluvial soils in this area are originally derived from loess and are highly erodible. This area supports a variety of land uses with forestry and small-scale agriculture predominating. The Delta area is a flat lowland with a highly productive agricultural economy (Mississippi Department of Environmental

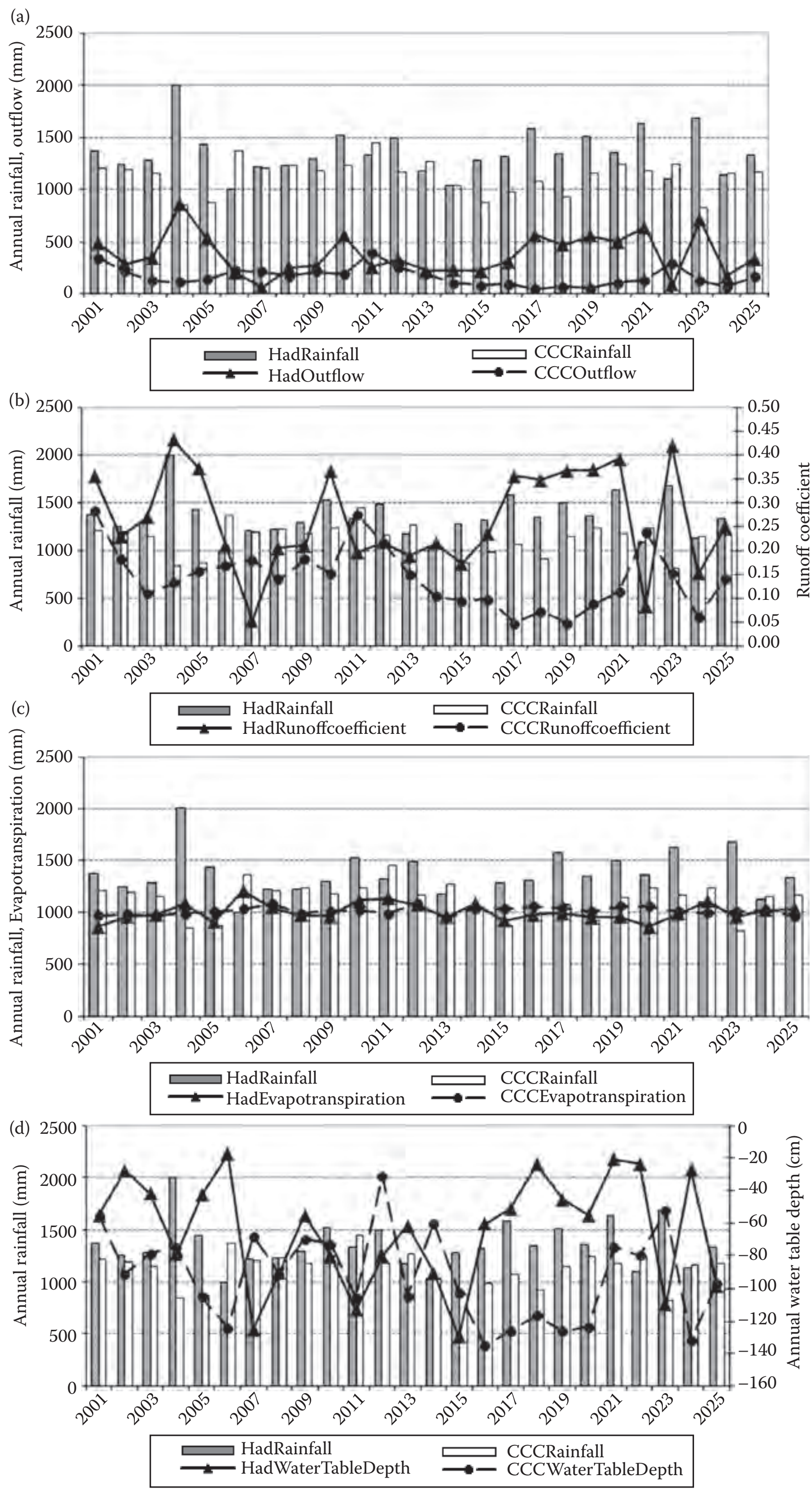


FIGURE 9.22 Annual rainfall at the Parker Tract site predicted by two general circulation models (HadCM2 and CGCI) for (a) drainage outflows, (b) runoff coefficients, (c) evapotranspiration, and (d) mean water table depths, as predicted by DRAINWAT using a 25-year climate scenario data series.

Quality 2008). The fine-textured, alluvial soils in this area are derived from past deposition by the ancestral Mississippi and Ohio Rivers (Guedon and Thomas 2004). Overall, the Yazoo area considered in this study is comprised of 31.8% (196 km^2) cropland, 4.4% (27.3 km^2) wetland, 0.3% (1.8 km^2) barren, 1.8% (11.5 km^2) urban, and 61.7% (381.2 km^2) forest uses. Environmental, climatic, and hydrologic data for the Yazoo (watershed 08030208) were obtained from a variety of public sources. Past climate data were obtained from weather stations or streamflow gauging stations within the Yazoo. These data were adjusted to represent the entire Yazoo basin. Potential evapotranspiration data were computed based on air temperature. These past climate data were used only for model calibration and validation purposes.

Four future climate scenarios were used in this study: HadCM B2, CSIROMK2 B2, CSIROMK3.5 A1B, and MIROC3.2 A1B. The data used are monthly air temperature and precipitation for a period from 2000 to 2050, which were computed for each U.S. Geological Survey 8-digit Hydrologic Unit Code (HUC) watershed within the Yazoo. These four climate scenarios were used to assess the impact of future climate changes on water discharge, evaporative loss, and water yield.

Model development: Two modeling systems are used in this study: the Better Assessment of Science by Integrating Point and Nonpoint Sources (BASINS) model (U.S. Environmental Protection Agency 2010) and the Hydrological Simulation Program-FORTRAN (HSPF) model (Bicknell et al. 2001). The BASINS system integrates a geographic information system, standardized environmental databases, and state-of-the-art modeling tools into one convenient package (U.S. Environmental Protection Agency 2010). The hydrological model is a comprehensive model for simulating many hydrologic processes within watersheds of almost any size and complexity (Bicknell et al. 2001).

Model calibration was done using a 5-year period from January 1, 2000 to December 31, 2004. To reduce uncertainty in the calibration process, only the six hydrologic parameters most sensitive to the hydrological model predictions (Donogian et al. 1984) were adjusted. The differences between the observed and predicted annual water outflow volumes for the calibration and validation period were within an acceptable range (Bicknell et al. 2001).

Effects of climate change on streamflow: Comparison of mean annual water-yield for the past 10 years (2001–2011) versus 40 years in the future (2011–2050) under the four climate scenarios indicates a continuing decrease (Table 9.7). The percent change in mean annual water yield varied from 29.47% for the CSIROMK3.5 A1B scenario to 18.51% for the MIROC3.2 A1B scenario, with all four climate scenarios indicating continuing decreases out to 2050. The same downward trends were observed for maximum annual water yields. The decreases in mean and maximum annual water yields were primarily the result of the projected precipitation decrease. Mixed results were found for the mean annual evaporative loss, with the CSIROMK2 B2 scenario indicating a long-term increase and the other three scenarios indicating a long-term decrease. Further research is thus necessary to better determine how evaporative losses will respond in the future.

Figure 9.23 shows the minimum, mean, and maximum values for monthly discharge rate and water yield (measured at the outlet of the watershed) under the four climate scenarios during the 40-year simulation period (2011–2050). In general, the MIROC3.2 A1B scenario produced the highest monthly minimum, mean, and maximum discharge rate and flow volume in most years because it forecasted the highest annual precipitation (Table 9.7). Overall, the mean and maximum annual discharge rates were projected to decrease in the Yazoo because of a decrease in precipitation during the next 40 years. Projected precipitation changes had profound impacts on flow discharge rate and annual water yield in this Mississippi Alluvial Valley watershed.

Case Study 3: Appalachian-Cumberland Highland

Extreme events often present greater challenges to water resource managers than those presented by average conditions. In this case study, we asked whether climate impacts within the Appalachian-Cumberland Highland may be either mitigated or exacerbated by forest management practices that alter land cover. We used a retrospective analysis of long-term climate and streamflow data

TABLE 9.7
Comparison of the Sum and Mean Values for Precipitation, Evaporative Loss, and Water Outflow between the Past 10 Years and Future 40 Years in the Yazoo River Basin

	Precipitation			Evaporative Loss			Water Yield		
Climate Scenario[a]	2001–2010 (mm/year)	2011–2050 (mm/year)	Change (%)	2001–2010 (mm/year)	2011–2050 (mm/year)	Change (%)	2001–2050 (mm/year)	2011–2050 (mm/year)	Change (%)
					Mean Annual				
HadCM3 B2	1477	1341	−10.1	1102	1002	−10.0	237	192	−23.7
MIROC3.2 A1B	1397	1269	−10.1	1186	1032	−14.9	154	134	−15.1
CSIROMK3.5 A1B	1646	1462	−12.6	1483	1199	−23.7	127	98	−29.6
CSIROMK2 B2	1387	1388	0.1	1143	1243	8.0	124	117	−6.3
					Maximum Annual				
HadCM3 B2	1831	1958	6.5	1419	1317	−7.8	348	384	9.3
MIROC3.2 A1B	1605	2517	36.2	1380	2073	33.4	188	326	42.2
CSIROMK3.5 A1B	2504	2174	−15.2	1896	1748	−8.5	352	385	8.6
CSIROMK2 B2	1615	2095	22.9	1295	1892	31.5	111	139	20.2

[a] The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 B2 climate scenarios for the Southern United States each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high-energy use, and B2 representing lower population growth and energy use).

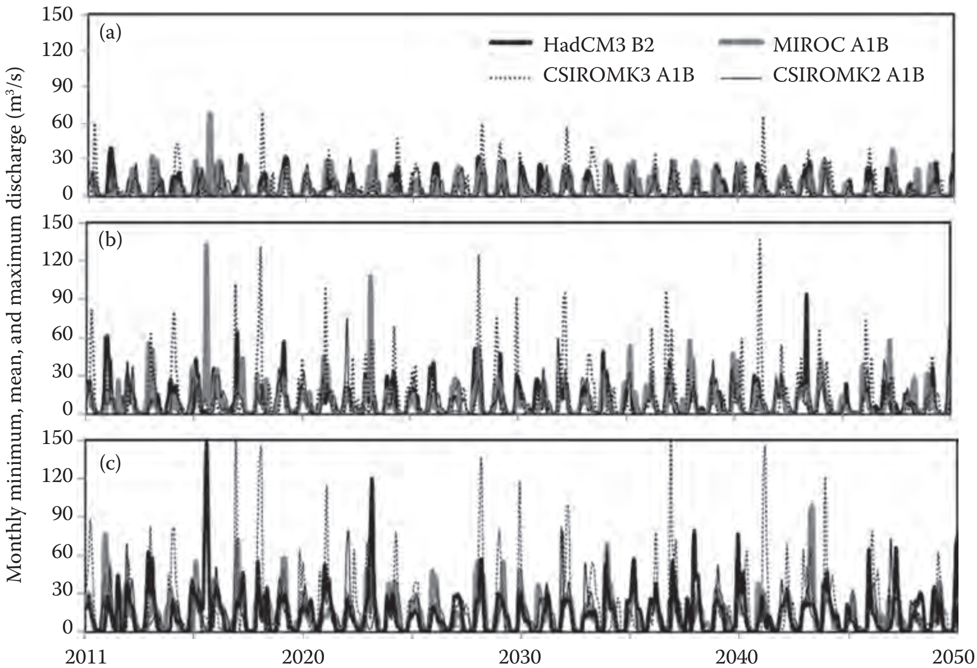


FIGURE 9.23 Simulated (a) monthly minimum, (b) mean, and (c) maximum discharge for the Yazoo River Basin from 2011 to 2050, based on four climate change scenarios. The MIROC3.2 A1B, CSIROMK2 B2, CSIROMK3.5 A1B, and HadCM3 B2 climate scenarios for the Southern United States (McNulty et al. in press) each combine a general circulation model (MIROC3.2, CSIROMK2, CSIROMK3.5, HadCM3) with an emissions storyline (A1B storyline representing moderate population growth and high energy use, and B2 representing lower population growth and energy use).

collected at the Coweeta Hydrologic Laboratory to determine whether streamflow from managed watersheds responds differently from unmanaged reference watersheds when subjected to variation in air temperature and extremes in annual precipitation.

The Coweeta Hydrologic Laboratory, located in western North Carolina, is a 2185-ha basin wherein climate and hydrologic monitoring and forest watershed experimentation began in the early 1930s. The basin has a marine, humid temperate climate with frequent, small, low-intensity rainfall events and annual precipitation averaging 1800 to 2300 mm, depending on elevation. Trend analysis suggests that air temperature at Coweeta has increased significantly since the early 1980s (Laseter et al. 2012). For this study, extreme precipitation years, both wet and dry, were identified according to Guttman (1999).

Forest management treatments: We used the long-term streamflow records from six watersheds, each with a different management and land-use history. Two watersheds (watershed 1 and watershed 17) were species conversion experiments from deciduous forests to evergreen, eastern white pine (*Pinus strobus*) plantations at 1.8-m × 1.8-m spacing. Two watersheds at high and low elevations (watershed 37 and watershed 7, respectively) were clear-cut using a range of techniques. Another watershed (watershed 13) was subjected to successive clear-cutting separated by a 23-year interval; the result was a multispecies coppice stand, in which the vegetation recovered via stump sprouting from existing, well-established root stock (Leopold et al. 1985). The remaining treatment watershed (watershed 6) underwent old-field succession following conversion of a mixed hardwood forest to a grass cover. All these management activities are still widely practiced on both publicly and privately managed forests in the Eastern United States, and thus represent prevalent land management strategies.

Modeling the interactions of management and precipitation on annual streamflow: The effect of management treatments on annual water yield was determined using the paired-watershed approach. Our goal was to predict the streamflow response to management knowing streamflow from an unmanaged watershed, a model of watershed response, and a model of the interaction of the watershed response and precipitation. Our approach was conceptually similar to the classic paired watershed regression approach detailed fully in Ford et al. (2011b). We found that interactions between watershed vegetation recovery and temperature were never significant and therefore air temperature is not included in the final model.

We used statistically downscaled mean annual precipitation from two climate scenarios, MIROC3.2 A1B and CSIROMK3.5 A1B, representing low and high forecasts of mean annual precipitation, respectively, to project streamflow from 2010 to 2050. Along with forecasted streamflow from the reference watershed, we modeled watershed treatment responses from the end of the observed series (2010) out to the year 2050. We assumed that forest communities on the treated catchments were comparable to the reference watershed in 2010, and that the original treatments were applied to the same catchments in 2011 (meaning that forest age was reset to 0 in 2011). We assumed that species composition and structure (such as stocking and leaf area index) recovered over the ~40-year posttreatment period comparable (both spatially and temporally) to what was observed in the actual treated watersheds.

Watershed recovery, management, and climate effects on streamflow: In all scenarios, land management altered the expected level of streamflow (Figure 9.24). Initial increases in streamflow compared to what would be expected had the catchment remained unmanaged ranged from 21.4 cm/year (19%) to 38.1 cm/year (70%) larger. Increases in streamflow immediately after treatment persisted generally only in the species conversions stands (3.5–5 years), and were associated with controlling competition so the new forest could establish. In general, streamflow recovered to pretreatment levels in 6–7 years. The rate of watershed recovery depended on management and

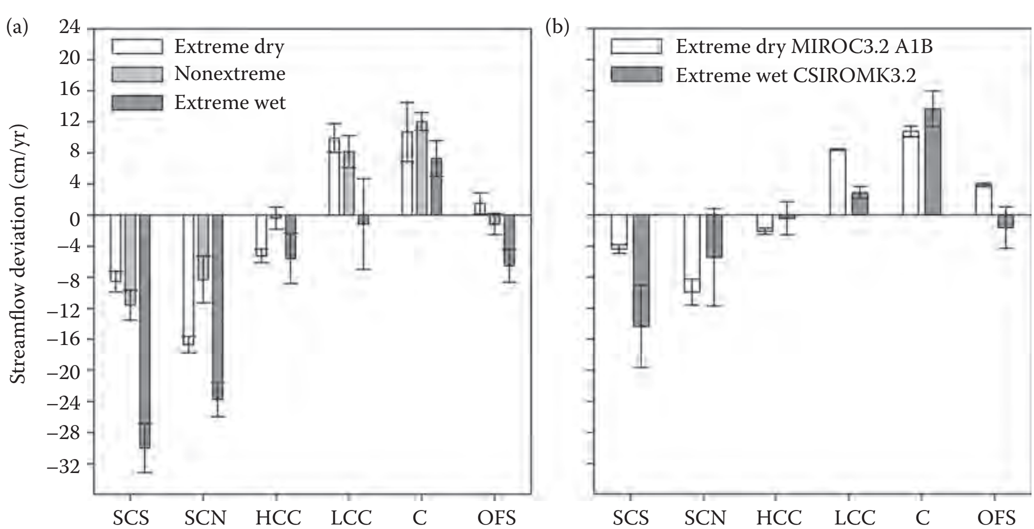


FIGURE 9.24 At the Coweeta Hydrologic Laboratory in the Appalachian-Cumberland Highland, (a) observed streamflow responses to six management treatments during extreme wet years, extreme dry years, and nonextreme years from the beginning of recordkeeping to 2010 as compared to (b) mean of extreme wet and dry years modeled repeating entire management cycle in all watersheds—reset stand age to 0 in 2011, forecast out to 2050—for both CSIROMK3.5A1B and MIROC3.2 A1B climate change scenarios. The MIROC3.2 A1B and CSIROMK3.5 A1B climate scenarios for the Southern United States (McNulty et al. in press) each combine a general circulation model (MIROC3.2 and CSIROMK3.5) with the A1B emissions storyline (representing moderate population growth and high-energy use). SCS, SCN = south- and north-facing species conversion; HCC, LCC = high- and low-elevation clearcut; C = copice; OFS = old field conversion.

location. Watershed recovery was faster for several low-elevation treatments compared to the high-elevation treatment, and also faster for south-facing stands compared to north-facing stands. The second cutting in the coppice stand experienced the fastest watershed recovery rate among all management treatments compared to the first cutting, which was among the slowest to recover.

At any forest age, the physical and biological components controlling hydrologic processes in the treatment watersheds responded differently than the reference watersheds to variation in precipitation (not temperature). The only exception to this was the coppice-managed stand. In all other watersheds, as precipitation increased, the treatment effect increased. This effect was most pronounced in the old-field succession, the south-facing species conversion, and the low-elevation clear-cut. In addition, these three land uses differed significantly from one another in the magnitude of the effect on streamflow deficit. The largest land use and climate interaction was seen in the old-field succession, followed by the two species conversions and the high-elevation clear-cut, then by the low-elevation clear-cut. In general, the drier the year, the more managed watersheds responded like the reference watersheds. Likewise, the wetter the year, the larger the differences were between control and treated watersheds.

Management simulated under climate change scenarios: Forecasted temperature and precipitation out to year 2050 were significantly different from the mean of observed conditions, depending on the climate scenario used. The MIROC3.2 A1B forecasted warmer and drier conditions for the Southern Appalachian Mountains, with drought frequency increasing appreciably: in a 10-year period, 6 years would be defined as drought years, compared to 1 year in 10 for the observed record. Extreme wet years were completely eliminated. Conversely, CSIROMK3.5 A1B forecasted wetter conditions for the Southern Appalachians, with frequency of extreme wet years increasing to 3 years in 10, compared to the 1 year in 10 for the observed record; droughts were completely eliminated.

We simulated watershed responses to future climate scenarios by assuming that the same set of management treatments were repeated in 2011 and we modeled responses to 2050 using forecasted climate. The most notable differences were observed for the south-facing species conversion and the coppice treatments (Figure 9.24). Streamflow responses for both management treatments were different from the observed record, primarily because the simulation period was relatively short (~40 years); thus the proportionally longer time period that the vegetation was in a younger, less-mature state than the reference watershed.

Potential for mitigation or exacerbation by forest management practices: Management affected the resulting vegetation structure and function, and the vegetation response to climate was different from the reference watersheds. Whether the effects of extreme wet or dry precipitation years exacerbated or mitigated the streamflow response depended on the management treatments. For example, managing a catchment with a species conversion treatment reduced annual streamflow during both extreme wet and dry event years, which may potentially exacerbate low flows and drought, but it also may mitigate high flows and flood risk. This conversion could decrease the apparent frequency of observed extreme wet event years on average by a factor of three (Figure 9.25). For example, in 2010 the annual streamflow generated from 2500 mm of precipitation would have been 130 cm without management; whereas with management, it was only 102 cm. The precipitation amount required to produce 102 cm on the watershed had it not been treated is only 2391 mm. The probabilities associated with 2500 mm compared to 2391 mm differ by a factor of three (0.0006 vs 0.002). Thus, the apparent frequency of extreme precipitation events decreased by threefold. This management treatment moves the observed streamflow distribution toward lower flows and could help mitigate high flows under a wetter future climate.

Pine forests in this temperate area intercept precipitation year-round with their evergreen canopies, as well as transpire year-round in the area's relatively mild winter temperatures. Both processes divert precipitation from streamflow in ways that differ from hardwood stands. Higher evapotranspiration from pine forests means that soils have a larger capacity to store excess water during wet years. This may be a good option under a future climate with increased precipitation, but a poor choice for a climate projected to be drier because the higher evapotranspiration also means that less soil water is available during drought conditions (Farley et al. 2005).

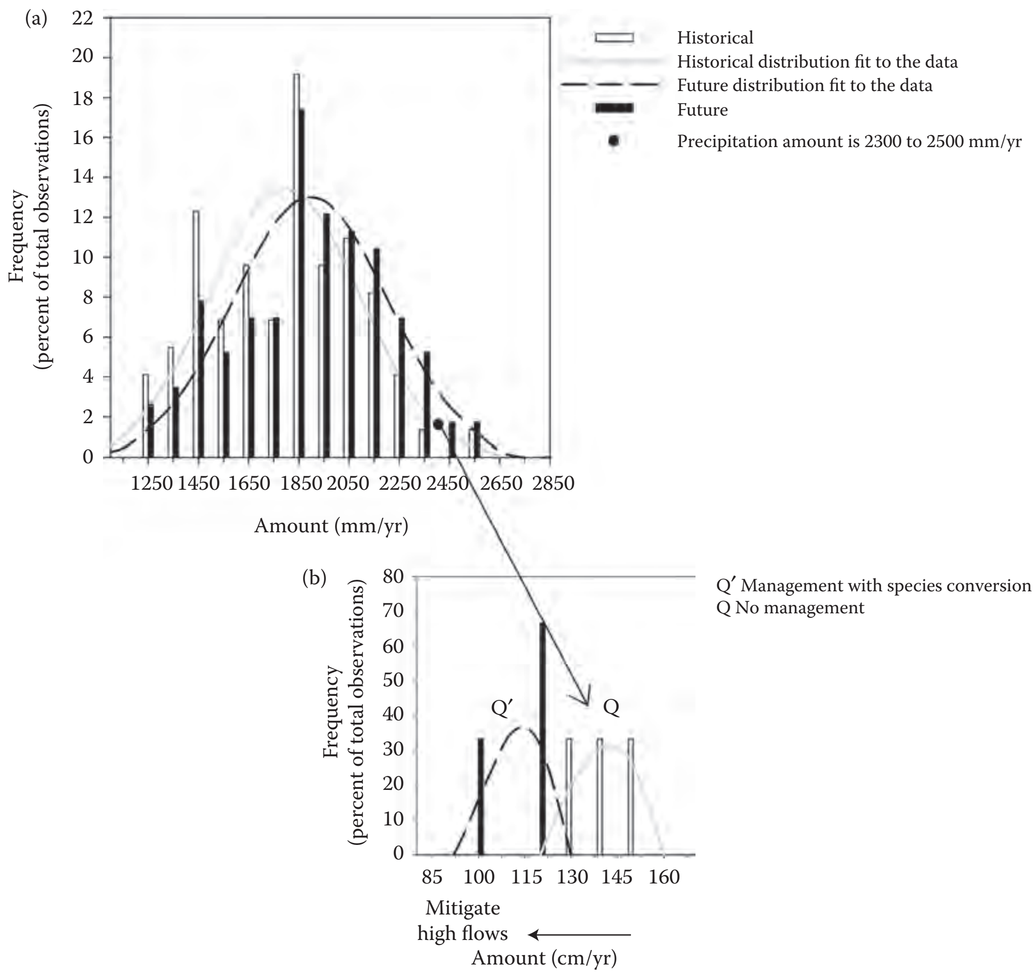


FIGURE 9.25 At the Coweeta Hydrological Laboratory in the Appalachian-Cumberland Highland, (a) frequency distribution of future annual precipitation—based on the CSIROMK3.5 A1B climate change scenario—and historical annual precipitation, and (b) observed streamflow frequency distribution resulting from management with a species conversion or no management for the given precipitation amount. The CSIROMK3.5 A1B climate scenario for the Southern United States (McNulty et al. in press) combines the CSIROMK3.5 general circulation model with the A1B emissions storyline (representing moderate population growth and high energy use).

Impacts on stream temperature in a headwater watershed: Watershed 32 (0.4-km^2 drainage area) at Coweeta is a control watershed with mixed hardwood stands that have remained undisturbed since 1927. This site had a significantly upward trend in estimated mean annual stream temperature of 0.12°C per decade from 1960 to 2007 (Figure 9.26). The mean change in stream temperature was predicted to increase to 0.19°C per decade from 2011 to 2060 under the HadCM3 B2 scenario. There is some variability in the estimated change of mean annual stream temperature among climate scenarios. For example, the change ranged from 0.16°C per decade under the CSIROMK3.5 A1B and CSIROMK2 B2 scenarios to 0.27°C per decade under the MIROC3.2 A1B scenario. Despite the variability in the change in future stream temperature, the trend was increasing and significant ($p < 0.05$) regardless of the climate scenario selected. The conservative HadCM3 B2 scenario predicted upward trends in annual monthly minimum (0.23°C per decade) and maximum stream temperature (0.24°C per decade) at this site. Thus, the

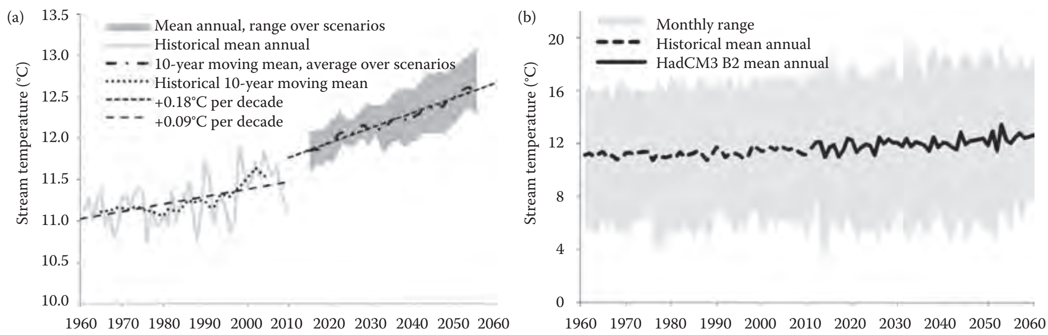


FIGURE 9.26 For watershed 32 at the Coweeta Hydrologic Laboratory in the Appalachian-Cumberland Highland, (a) 10-year moving mean of mean annual stream temperature and (b) mean annual stream temperature and annual range in monthly stream temperature under historic and HadCM3 B2 future climate conditions. The HadCM3 B2 climate scenario for the Southern United States (McNulty et al. in press) combines the HadCM3 general circulation model with the B2 emissions storyline (representing lower population growth and energy use).

mean annual maximum stream temperature was predicted to increase from 11.9°C to 12.5°C by the decade ending in 2060.

Case Study 4: Mid-South

Located in the northeast corner of the Mid-South, the Ouachita National Forest covers over 6800 km^2 of the Ouachita Mountains in western-central Arkansas and eastern Oklahoma (Figure 9.27). Surface erosion and sediment delivery to streams and water bodies is a major ongoing concern of

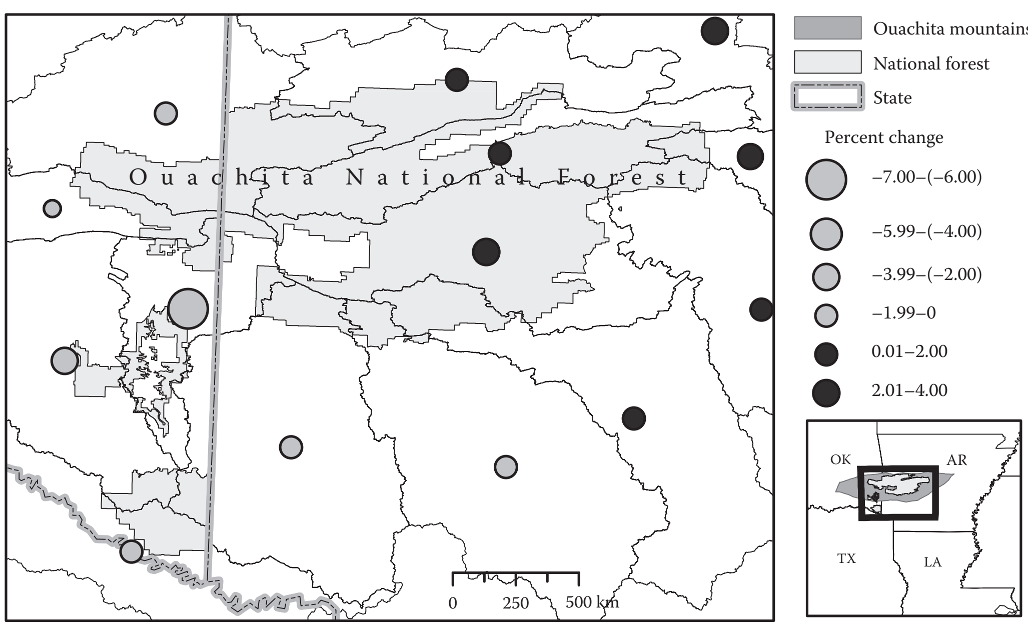


FIGURE 9.27 Percent change in mean annual precipitation from historical for the 2041–2060 period in watersheds draining the Ouachita National Forest based on an ensemble general circulation model with the A1B emissions storyline (representing moderate population growth and high-energy use). (Adapted from Mote, T.L. and J.M. Shepherd. 2011. Monthly ensemble downscaled climate model data for the Southeast U.S. http://shrmc.ggy.uga.edu/downscale.php. Accessed: November 07, 2011.)

the Forest (U.S. Department of Agriculture, Forest Service 2005) and of forest managers throughout the subregion. Within the forest boundaries, the predominant sediment-generating process is surface erosion related to timber harvesting, off-highway vehicle trails, and forest roads. A limited amount of private agricultural land occurs within or adjacent to the Ouachita, and these areas can also be important sediment sources. Concern about sediment impacts to stream habitats and water quality prompted the development of the Aquatic Cumulative Effects (ACE) model (Clingenpeel and Crump 2005) to assess sediment risks within the Ouachita.

Anticipated changes in the climate affecting the Ouachita suggest that future surface erosion risks may change. Predicted temperature increases can affect vegetation in ways that reduce ground cover and root binding of the soil (Pruski and Nearing 2002b). Changes in rainfall amount and intensity can directly affect erosion drivers like runoff amount and raindrop impact energy, as well as indirectly affect the amount and types of vegetation that can be sustained. Rainfall changes are the primary concern as snowfall represents only a very small portion of annual precipitation and its effect on sediment production is negligible.

To assess the overall sediment risk of climate change on the Ouachita, we conducted a series of simulations to determine how plausible changes in rainfall amount, rainfall intensity, and vegetation cover affect sediment risk under a future climate. Future precipitation and temperatures were projected using an ensemble general circulation model and the A1B emissions storyline representing low population/high economic growth and high-energy use (Intergovernmental Panel on Climate Change 2001) and hereafter referred to as the "ensemble" scenario. Based on future temperature and rainfall, different levels for each of the factors that affect sediment production were determined. The ACE model (Clingenpeel and Crump 2005) was used to assess how changes in these factors might affect sediment risks across the Ouachita. Finally, the effect of increasing road maintenance on future sediment risks was also evaluated.

Study methods: The period from 2041 to 2060 was selected as the future period for which sediment risks would be simulated. The climate characteristics computed were annual precipitation, monthly precipitation, and mean monthly temperature for each year within the 20-year period. These characteristics were computed for the watersheds that drain the majority of the Ouachita lands using the U.S. Geological Survey 8-digit Hydrologic Unit Code (HUC) scale.

The ACE model (Clingenpeel and Crump 2005) computed the estimated sediment yield within a12-digit HUC (hereafter called a "subwatershed"). Sediment yield from road areas was estimated using the WEPP model (Elliot 2004); for all other areas it was estimated using a version of the Universal Soil Loss equation (Wischmeier and Smith 1965) with model factor values developed for forestry and related management practices (Dissmeyer and Stump 1978). For the 2041 to 2060 period, the distribution of different land-use practices was assumed to be the same as that which currently occurs. The combined sediment yield from road and unroaded areas was used to determine the sediment risk class (high, moderate, or low) for each subwatershed. A review of the ACE model is given in Marion and Clingenpeel (2012). For this analysis, we computed the sediment risk for all 190 subwatersheds within the Ouachita.

The levels of three factors were varied to explore how Ouachita sediment risks might respond to climate change: the R-factor, the C-factor, and the number of wet days per month. The R-factor is a measure of precipitation erosivity (Wischmeier and Smith 1965) and is used in the ACE model for estimating sediment losses in unroaded areas. Three different R-factor values were considered. The first is the R-factor predicted using the mean of the projected annual precipitations for the 2041–2060 period (Renard and Freimund 1994). The other two R-factor values were ±15% of the predicted R-factor for each watershed. R-factors were computed for each watershed and then used to compute mean values for each Ouachita subsection (see Clingenpeel and Crump 2005 for an explanation of how subsections are used in the ACE model). The subsection means were then assigned to all subwatersheds within each subsection.

The C-factor is a measure of the relative effect of the ground cover on sediment loss. The higher the C-factor value, the more sediment is lost through erosion. Two C-factor levels were considered:

no change from the current C-factor value, and a 15% increase. The 15% increase is based on Pruski and Nearing (2002b), who found that under certain soil and climate conditions, a decrease in precipitation could result in an increase in sediment loss caused by a decrease in biomass. The simulations of Pruski and Nearing (2002b) only considered agricultural conditions, but the locations that produced this response are those that are geographically closest to the Ouachita. Lacking other guidance for how C-factors might change under forest conditions, we assumed that the same response is possible on the Ouachita. C-factors were assigned to subwatersheds by subsection.

The number of wet days per month is a variable in the WEPP-Road (Elliot 2004) routine used by the ACE model to estimate sediment yield from road areas. This variable was used here to simulate increases in rainfall intensity by decreasing the number of wet days per month. Two factor levels were analyzed: a 20% decrease and at 50% decrease. A "no change" level was not considered because precipitation intensity has already been shown to be increasing in the Mid-South (White 2011).

Three additional factors were used to capture other climate change effects on road sediment losses in the 2041–2060 period: monthly maximum temperature, monthly minimum temperature, and mean monthly precipitation. These factors are all used by the WEPP-Road routine. They were taken directly from the climate scenarios and held constant during all simulations. The minimum and maximum from each 20-year mean monthly temperature series were used to estimate the respective minimum and maximum monthly temperatures, and the mean of each series was used to estimate the mean monthly precipitation.

Lastly, we also examined how increased maintenance on roads might reduce the sediment risks. All factor combinations were computed assuming the current maintenance level will be used from 2041 to 2060, and then again assuming an increased maintenance level.

Findings: The ensemble scenario indicates that by 2060 mean annual air temperature will increase 2.6–3.4°C relative to the current period in the Ouachita watersheds. Mean annual precipitation in the watersheds is projected to decrease somewhat (0.9–7.0%) on the western side of the Ouachita (Figure 9.27); but increase somewhat (0.4–2.6%) on the eastern side. The change in R-factor values between the current period and the 2041-to-2060 period follows the precipitation change pattern, with R-factor values decreasing 1.8–13.7% on the western side and increasing 0.6–5.3% on the eastern side.

The combined effects of the different changes in total rainfall amount, rainfall intensity, and vegetation cover on the sediment risk ratings are shown in Figure 9.28. The predicted changes in R-factors have little effect on the risk classifications of the Ouachita subwatersheds unless a concomitant change in C-factors occurs. Using the predicted R-factors, the numbers of subwatersheds in each risk class do not change appreciably from their current number. If R-factors are 15% less (low R condition), noticeable improvement occurs, with 25 subwatersheds moving from a higher to a lower risk class. The sediment risk worsens when R-factors are 15% larger than the predicted values (high R condition), with 15 subwatersheds shifting up to a higher risk class. These results indicate that future conditions would have to become wetter than is projected by the ensemble model before erosivity changes alone would worsen the current sediment risk.

However, if C-factors change concurrently with the R-factors, the effect on sediment risk is magnified. A 15% increase in the C-factor (reduced vegetation cover) causes the number of high-risk subwatersheds to increase well above the current number under both the predicted and high R levels, and to be equivalent to the current number under the low R level. Curiously, the number of low risk subwatersheds at the low R level actually increases above that under current conditions when the C-factor is increased 15%. This result may be important because if the future climate is drier than predicted by the ensemble climate scenario, the chance of C-factors increasing would be relatively higher than under the wetter conditions represented by the predicted and high R levels.

Surprisingly, increasing the rainfall intensity by decreasing the number of wet days from −20% to −50% had almost no effect on the sediment risk classifications. This suggests that the possible increase in road sediment generated by increasing the rainfall intensity is not sufficient by itself to

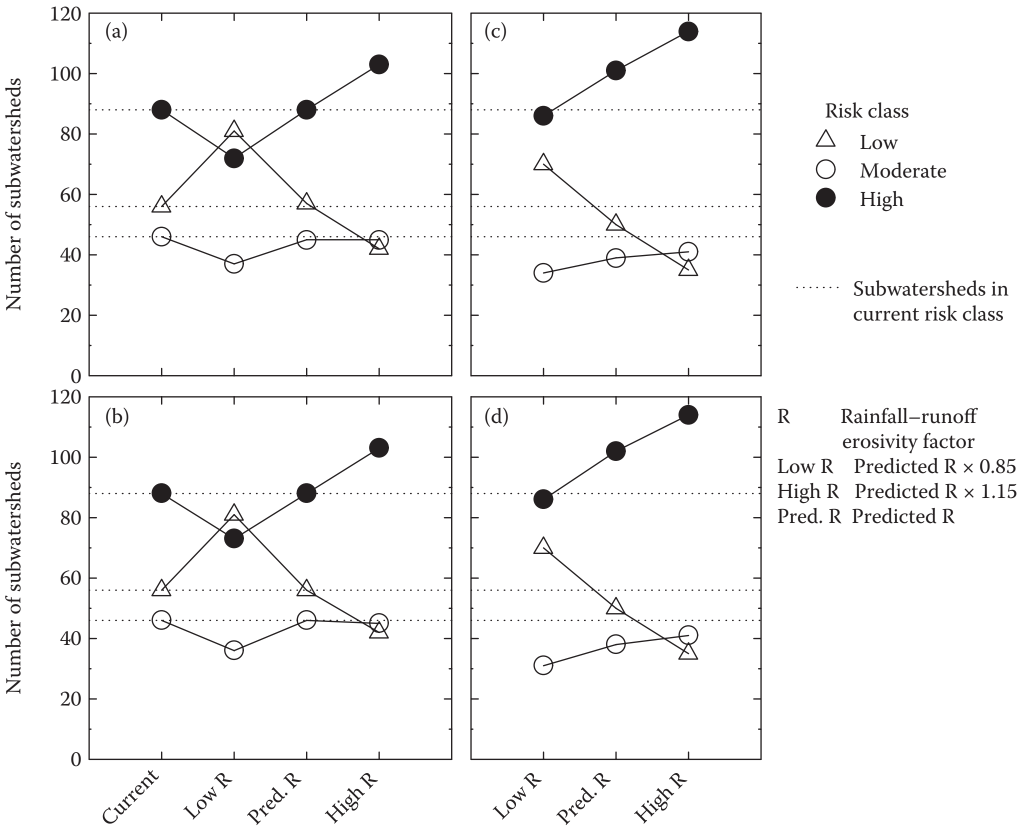


FIGURE 9.28 Effects of current and predicted (2041–2060) changes in total rainfall amount, rainfall intensity, and vegetation cover factor on the sediment risk rankings for Ouachita National Forest subwatersheds: (a) 20% decrease in wet days with no change in vegetation cover factor, (b) 50% decrease in wet days with no change in vegetation cover factor, (c) 20% decrease in wet days with 15% increase in vegetation cover factor, and (d) 50% decrease in wet days with 15% increase in vegetation cover factor. The rainfall–runoff erosivity and vegetation cover factors (Wischmeier and Smith 1965), and the number of wet days per month (Elliot 2004) are used within the Aquatic Cumulative Effects model (Clingenpeel and Crump 2005) to predict sediment risk ranking. Rainfall–runoff erosivity and number of wet days are based on precipitation predicted for the 2041–2060 period using an ensemble general circulation model and the A1B emissions storyline (representing moderate population growth and high-energy use) (Mote and Shepherd 2011).

alter the effects caused in unroaded areas by changes in the R- and C-factors. However, increasing rainfall intensity does have an effect when road maintenance is considered.

Increasing road maintenance could reduce possible increases in sediment risk resulting from the climate changes discussed so far. The effect of increasing road maintenance above current levels on sediment risk classification is shown in Figure 9.29. Increasing road maintenance always decreases road erosion in the ACE model; therefore, it is not surprising that the number of improved subwatersheds (those moving to a lower risk class) increased for all factor-level combinations. The number of improved subwatersheds always increases when the R-factor level decreases (moving down on the *y*-axis) or the C-factor level decreases (moving from left to right). The general lack of effect from increasing rainfall intensity is also evident when values for the two intensity levels are compared within a given combination of R- and C-factor levels. However, a small effect is apparent when the C-factors are unchanged and the R-factors are either at the predicted or at the low level. For predicted R-factors, the number of improved subwatersheds with increased road maintenance

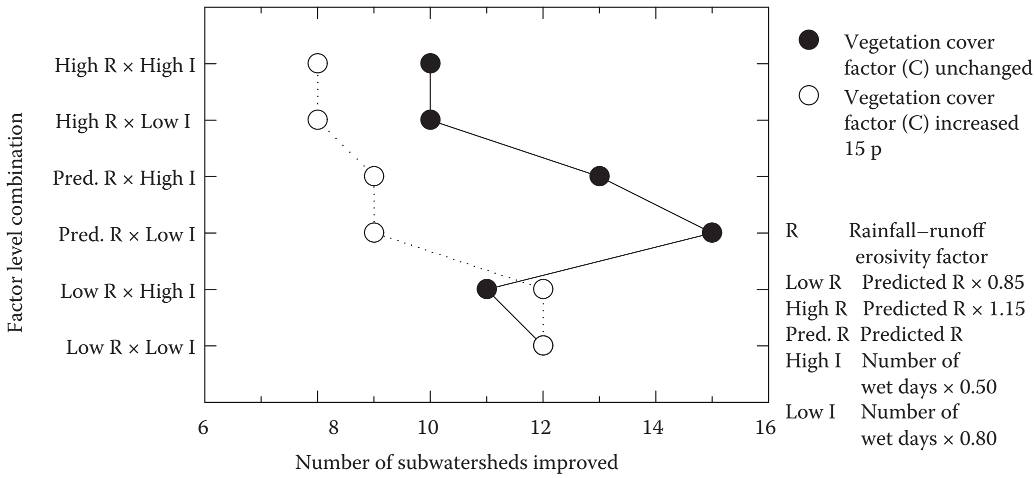


FIGURE 9.29 Effect of increased road maintenance on the number of Ouachita National Forest subwatersheds with improved sediment risk rankings—higher to lower risk class—under varying future rainfall and forest conditions. The rainfall–runoff erosivity and vegetation cover factors (Wischmeier and Smith 1965), number of wet days per month (Elliot 2004), and road maintenance levels are used within the Aquatic Cumulative Effects model (Clingenpeel and Crump 2005) to predict sediment risk ranking. Rainfall–runoff erosivity and number of wet days are based on precipitation predicted for the 2041–2060 period using an ensemble general circulation model and the A1B emissions storyline (representing moderate population growth and high-energy use) (Mote and Shepherd 2011).

decreases by two when rainfall intensity increases to the high level, but it decreases by one for the low R condition.

In summary, results from these simulations indicate that changes in mean annual precipitation predicted from a climate projection for the Ouachita from 2041 to 2060 and an assumed increase in rainfall intensity based on 20% fewer wet days are not sufficient by themselves to worsen the current sediment risk ratings. However, if these changes happen concurrently with decreased vegetative cover, then such a situation is possible. Decreased vegetative cover is a distinct possibility given the increased air temperatures predicted for this period. To a substantial degree, potential increases in sediment risk could be mitigated by increasing road maintenance within the forest.

SUMMARY

Forest managers and policy makers in the South will face growing challenges in mitigating or adapting to the effects of climate change on water resources in the near future. This chapter identified the key water resource characteristics related to forestlands that are mostly likely to be affected and how they will respond based on past research and new analyses. Future climate conditions were projected using climate scenarios derived from commonly used general circulation models and emission storylines. The region-wide sections show that in the future, important increases in evapotranspiration, water temperature, soil erosion, and coastal salinity intrusion are projected to occur either across broad areas of the South or within specific localities. The discussion of mitigation and adaptation options notes which BMPs are likely to be effected by changing climate conditions and how they may need to be modified. The case studies provide more detailed assessments of impacts to particular water resources within each of the subregions of the South. The information in this chapter should provide forest managers and policy makers a sound knowledge base from which to identify the potential issues they will need to consider in their future, location-specific plans, as well as a rich set of examples of analysis approaches they might use in accomplishing their own assessments.

REFERENCES

Alcamo, J., T. Henrichs, and T. Rosch. 2000. World water in 2025: Global modeling and scenario analysis for the World Commission on Water for the 21st century. Kassel World Water Series, Kassel, Germany. Report No 2.

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

Amatya, D.M., T.J. Callahan, C.C. Trettin et al. 2009. Hydrologic and water quality monitoring on Turkey Creek Watershed, Francis Marion National Forest, SC. *Annual ASABE International Meeting*, Reno, Nevada. Paper 09–5999.

Amatya, D.M., G.M. Chescheir, G.P. Fernandez et al. 2003. Lumped parameter models for predicting nitrogen transport in lower Coastal Plain watersheds. Water Resources Research Institute, Raleigh, North Carolina. Report 347.

Amatya, D.M., G.M. Chescheir, G.P. Fernandez et al. 2004. DRAINWAT-based methods for estimating nitrogen transport on poorly drained watersheds. *Transactions of the ASAE* 43(3): 677–687.

Amatya, D.M., J.D. Gregory, and R.W. Skaggs. 2000. Effects of controlled drainage on storm event hydrology in a loblolly pine plantation. *Journal of the American Water Resources Association* 36(1): 175–190.

Amatya, D.M., M. Miwa, C.A. Harrison et al. 2006a. Hydrology and water quality of two first order watersheds in coastal South Carolina. *American Society of Agricultural Engineers*. St. Joseph, Michigan. Paper 062182.

Amatya, D.M. and A. Radecki-Pawlik. 2007. Flow dynamics of three forested watersheds in coastal South Carolina, U.S.A. *Acta Scient. Polonorum—Formatio Circumiectus* 6(2): 3–17.

Amatya, D.M. and R.W. Skaggs. 2011. Long-term hydrology and water quality of a drained pine plantation in North Carolina, U.S.A. *Transactions of the ASABE* 54(6): 2087–2098 (special issue "Advances in Forest Hydrology").

Amatya, D.M., R.W. Skaggs, and J.D. Gregory. 1996. Effects of controlled drainage on the hydrology of a drained pine plantation in the North Carolina coastal plains. *Journal of Hydrology* 181: 211–232.

Amatya, D.M., R.W. Skaggs, and J.D. Gregory. 1997. Evaluation of a watershed scale forest hydrologic model. *Journal of Agricultural Water Management* 32(1997): 239–258.

Amatya, D.M., G. Sun, R.W. Skaggs et al. 2006b. Hydrologic effects of global climate change on a large drained pine forest. *In*: Williams and J.E. Nettles (eds). *Proceedings of the International Conference on Hydrology and Management of Forested Wetlands*. pp. 383–394. ASABE, St. Joseph, Michigan.

Amatya, D.M. and C.C. Trettin. 2007. Development of watershed hydrologic studies at Santee Experimental Forest, South Carolina. *In:* M.J. Furniss et al. (eds). *Proceedings of the Forest Service National Earth Sciences Conference*, Volume I. pp. 180–190. US Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon. PNW-GTR-689.

Amatya, D.M. and R.W. Skaggs. 2001. Hydrologic modeling of pine plantations on poorly drained soils. *Forest Science* 47(1): 103–114.

Appelboom, T.W., G.M. Chescheir, R.W. Skaggs et al. 2008. Nitrogen balance for a plantation forest drainage canal on the North Carolina coastal plain. *Transactions of the ASABE* 51(4): 1215–1233.

Arnoldus, H.M.J. 1977. Methodology used to determine the maximum potential average annual soil loss due to sheet and rill erosion in Morocco. *Food and Agriculture Organization Soils Bulletin* 34: 39–51.

Baron, J.S., L. Gunderson, C.D. Allen et al. 2009. Options for National Parks and Reserves for adapting to climate change. *Environmental Management* 44: 1033–1042, doi:10.1007/s00267-009-9296-6.

Bear, J., A.H.D. Cheng, S. Sorek et al., eds. 1999. *Seawater Intrusion in Coastal Aquifers—Concepts, Methods and Practices*. Kluwer Academic Publishers, Dordrecht, the Netherlands. 640 p.

Beerling, D.J. 1996. Ecophysiological responses of woody plants to past CO_2 concentrations. *Tree Physiology* 16: 389–396.

Bicknell, B.R., J.C. Imhoff, J.L. Kittle Jr. et al. 2001. *Hydrological Simulation Program—Fortran, HSPF User's Manual, Version 12*. US EPA, Office of Research and Development, National Exposure Research Laboratory, Athens, Georgia.

Bogan, T., O. Mohseni, and H.G. Stefan. 2003. Stream temperature—equilibrium temperature relationship. *Water Resources Research* 39(9): 1245.

Breshears, D.D., N.S. Cobb, P.M. Rich et al. 2005. Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences* 102(42): 15144–15148.

Bryant, M.L., S. Bhata, and J.M. Jacobs. 2005. Measurements and modeling of throughfall variability for five forest communities in the Southeastern US. *Journal of Hydrology* 312: 95–108.

Burton, T.M. and G.E. Likens. 1973. The effect of strip-cutting on stream temperature in the Hubbard Brook Experimental Forest, New Hampshire. *BioScience* 23(7): 433–435.

Caldwell, P.V., G. Sun, S.G. McNulty et al. 2012. Impacts of impervious cover, water withdrawals, and climate change on river flows in the conterminous U.S. *Hydrology and Earth System Sciences* 16: 2839–2857.

Carlisle, D.M., D.M. Wolock, and M.R. Meador. 2010. Alteration of streamflow magnitudes, and potential ecological consequences: A multiregional assessment. *Frontiers in Ecology and the Environment* 2010, doi:10.1890/100053.

Cech, P.G., S. Pepin, and C. Körner. 2003. Elevated CO_2 reduces sap flux in mature deciduous forest trees. *Oecologia* 137: 258–268.

Chapman, D.S. and M.G. Davis. 2010. Climate change: Past, present, and future. *EOS* 91(37): 325–326.

Chen, G., M. Notaro, Z. Liu et al. 2012. Simulated local and remote biophysical effects of afforestation over the southeast United States in boreal summer. *Journal of Climate* 25: 4511–4522.

Clingenpeel, J.A. and M.A. Crump. 2005. *A manual for the Aquatic Cumulative Effects model*. Unpublished paper. Ouachita National Forest, Supervisors Office, Hot Springs. Arkansas.

Conrads, P.A. and E.A. Roehl Jr. 2007. Analysis of salinity intrusion in the Waccamaw River and the Atlantic Intracoastal Waterway near Myrtle Beach, South Carolina, 1995–2002. US Geological Survey. Scientific Investigations Report 2007–5110.

Coulson, D.P., L.A. Joyce, D.T. Price et al. 2010a. Climate scenarios for the conterminous United States at the county spatial scale using SRES scenarios B2 and PRISM climatology. US Forest Service, Rocky Mountain Research Station. http://www.fs.fed.us/rm/data_archive/dataaccess. Accessed: November 1, 2012.

Coulson, D.P., L.A. Joyce, D.T. Price et al. 2010b. Climate scenarios for the conterminous United States at the county spatial scale using SRES scenarios A1B and A2 and PRISM climatology. US Forest Service, Rocky Mountain Research Station. http://www.fs.fed.us/rm/pubs_other/rmrs_2010_coulson_d005.pdf. Accessed: November 1, 2012.

Crockford, R.H. and D.P. Richardson. 2000. Partitioning of rainfall into throughfall, stemflow and interception: Effect of forest type, ground cover and climate. *Hydrological Processes* 14: 2903–2920.

Cruise, J.F., A.S. Limaye, and N. Al-Abed. 1999. Assessment of impacts of climate change on water quality in the southeastern United States. *Journal of the American Water Resources Association* 35(6): 1539–1550.

Dai, Z., D. Amatya, G. Sun et al. 2011a. Climate variability and its impact on forest hydrology on South Carolina Coastal Plain of USA. *Atmosphere* 2: 330–357, doi:10.3390/atmos2030330.

Dai, Z., C. Li, C. Trettin et al. 2010a. Bi-criteria evaluation of MIKE SHE model for a forested watershed on South Carolina coastal plain. *Hydrology and Earth Systems Science*. 14: 179–219, doi:10.5194/hess-14-1033-2010.

Dai, Z., C.C. Trettin, and D.M. Amatya. 2011b. Impacts of climate variability on forest hydrology and carbon sequestration at Santee Experimental Forest in coastal South Carolina. U.S. Department of Agriculture, Forest Service, Southern Research Station. General Technical Report (in review).

Dai, Z., C. Trettin, C. Li et al. 2010b. Sensitivity of stream flow and water table depth to potential climatic variability in a coastal forested watershed. *Journal of the American Water Resources Association* 46(5): 1036–1048, doi:10.1111/j.1752-1688. 2010.

Daly, C., R.P. Neilson, and D.L. Phillips. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. *Journal of Applied Meteorology* 33: 140–158.

Danish Hydraulic Institute. 2005. MIKE SHE Technical Reference, Version 2005. DHI Water and Environment. Danish Hydraulic Institute, Denmark.

David, J.L., B.A. Middleton, and R.O. Anyah. 2010. Modeled climate change impacts on the production of bald cypress swamps across a latitudinal gradient. *Proc. Society of Wetland Scientists Conference*, Salt Lake City, Utah.

Dissmeyer, G.E. and G.R. Foster. 1984. A guide for predicting sheet and rill erosion on forestland. U.S. Department of Agriculture, Forest Service. Gen. Tech. Publ. R8-TP 6. 40 p.

Dissmeyer, G.E. and R.F. Stump. 1978. Predicted erosion rates for forest management activities and conditions sampled in the Southeast. Administrative report. U.S. Department of Agriculture Forest Service, State and Private Forestry, Southern Region, Atlanta, Georgia. 26 p.

Donogian, A.S., J.C. Imhoff, B.R. Bicknell et al. 1984. Application guide for hydrological simulation program-FORTRAN (HSPF). EPA, Athens, Georgia. EPA-600/3-84-065.

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

Easterling, D.R., G.A. Meehl, C. Parmesan et al. 2000b. Climate extremes: Observations, modeling, and impacts. *Science* 289: 2068–2074, doi:10.1126/science.289.5487.2068.

Eaton, G.J. and R.M. Scheller. 1996. Effects of climate warming on fish thermal habitat in streams of the United States. *Limnology & Oceanography* 41(5): 1109–1115.

Elliot, W.J. 2004. WEPP internet interfaces for forest erosion prediction. *Journal of the American Water Resources Association* 40(2): 299–309.

Elmore, A.J, S.M. Guinn, B.J. Minsley et al. 2012. Landscape controls on the timing of spring, autumn, and growing season length in mid-Atlantic forests. *Global Change Biology* 18: 656–674. doi: 10.1111/j.1365-2486.2011.02521.x.

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

Fernandez, G.P., G.M. Chescheir, R.W. Skaggs et al. 2006. DRAINMOD-GIS: A lumped parameter watershed scale drainage and water quality model. *Agricultural Water Management* 81(2006): 77–97.

Fernandez, G.P., G.M. Chescheir, R.W. Skaggs et al. 2007. Application of DRAINMOD-GIS to a lower coastal plain watershed. *Transactions of the ASABE* 50(2): 439–447.

Ford, C.R., R.M. Hubbard, and J.M. Vose. 2011a. Quantifying structural and physiological controls on variation in canopy transpiration among planted pine and hardwood species in the southern Appalachians. *Ecohydrology* 4: 183–195.

Ford, C.R., S.H. Laseter, W.T. Swank et al. 2011b. Can forest management be used to sustain water-based ecosystem services in the face of climate change? *Ecological Applications* 21: 2049–2067.

Franks, P.J. and D.J. Beerling. 2009. Maximum leaf conductance driven by CO_2 effects on stomatal size and density over geologic time. *Proceedings of the National Academy of Sciences* 106(25): 10343–10347.

Furlow, J., J.D. Scheraga, R. Freed et al. 2002. The vulnerability of public water systems to sea level rise. *In:* J.R. Lesnik, (ed). *Proceedings of the Coastal Water Resource Conference*. pp. 31–36. American Water Resources Association, Middleburg, Virginia.

Galloway, G.E. 2011. If stationarity is dead, what do we do now? *Journal of the American Water Resources Association* 47(3): 563–570, doi: 10.1111/j.1752-1688.2011.00550.x.

Gedney, N., P.M. Cox, R.A. Betts et al. 2006. Detection of a direct carbon dioxide effect in continental river runoff records. *Nature* 439: 835–838.

Gibson, W.P., C. Daly, T. Kittel et al. 2002. Development of a 103-year high-resolution climate data set for the conterminous United States. *In: Proceedings, 13th AMS Conference on Applied Climatology*. pp. 181–183. American Meteorological Society, Portland, Oregon.

Groisman, P.Y., R.W. Knight, T.R. Karl et al. 2003. Contemporary changes of the hydrological cycle over the contiguous United States: Trends derived from *in situ* observations. *Journal of Hydrometeorology* 5(1): 64–85.

Guedon, N.B. and J.V. Thomas. 2004. State of Mississippi Water Quality Assessment Section 305(b) Report Appendum. Mississippi Department of Environmental Quality, Jackson, Mississippi.

Gunn, J.S., J.M. Hagan, and A.A. Whitman. 2009. Forestry adaptation and mitigation in a changing climate. A forest resource manager's guide for the northeastern United States. Natural Capital Initiative at Manomet. Ecosystem Services Resources Center Report NCI-2009-1.

Guttman, N.B. 1999. Accepting the standardized precipitation index: A calculation algorithm. *Journal of the American Water Resources Association* 35: 311–322.

Hänninen, H. and K. Tanino. 2011. Tree seasonality in a warming climate. *Trends in Plant Science* 16: 412–416.

Hansen, J., M. Sato, R. Ruedy et al. 2006: Global temperature change. *Proceedings of the National Academy of Sciences* 103: 14288–14293.

Hansen, J.E., R. Ruedy, M. Sato et al. 2010. Global surface temperature change. *Reviews of Geophysics* 48: RG4004, doi:10.1029/2010RG000345.

Hanson, P.J., S.D. Wullschleger, R.J. Norby et al. 2005. Importance of changing CO_2, temperature, precipitation, and ozone on carbon and water cycles of an upland oak forest: Incorporating experimental results into model simulations. *Global Change Biology* 11: 1402–1423.

Harder, S.V., D.M. Amatya, T.J. Callahan et al. 2007. A hydrologic budget of a first order forested watershed, South Carolina. *Journal of the American Water Resources Association* 43(3): 1–13.

Hatch, U., S. Jagtap, J. Jones et al. 1999. Potential effects of climate change on agricultural water use in the southeast US. *Journal of the American Water Resources Association* 35(6): 1551–1561.

Helvey, J.D. 1967. Interception by eastern white pine. *Water Resources Research* 3(3): 723–729.

Helvey, J.D. and J.H. Patric. 1965. Canopy and litter interception of rainfall by hardwoods of eastern United States. *Water Resources Research* 1: 193–206.

Hook, D.D., M.A. Buford, and T.M. Williams. 1991. The impact of Hurricane Hugo on the South Carolina coastal plain forest. *Journal of Coastal Research*, special issue 8: 291–300.

Hossian, F., D. Niyogi, J. Adegoke et al. 2011. Making sense of the water resources that will be available for future use. *EOS* 92(17): 144–145.

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. 2001. *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. J.T. Houghton, Y. Ding, D.J. Griggs et al. (eds). Cambridge University Press, Cambridge, United Kingdom and New York, New York.

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*: Core Writing Team, R.K. Pachauri, and A. Reisinger (eds). *Climate Change 2007: Synthesis Report*. Geneva, Switzerland.

Iverson, L.R., A.M. Prasad, S.N. Matthews et al. 2008. Estimating potential habitat for 134 eastern US tree species under six climate scenarios. *Forest Ecology and Management* 254: 390–406.

Jackson, R.B., E.G. Jobbagy, R. Avissar et al. 2005. Trading water for carbon with biological carbon sequestration. *Science* 310: 1944–1947, doi:10.1126/science.1119282.

Jackson, R.B., J.T. Randerson, J.G. Canadell et al. 2008. Protecting climate with forests. *Environmental Research Letters* 3: 044006, doi:10.1088/1748-9326/3/4/044006.

Jensen, B.A. 1994. *Expert Systems—Neural Networks, Instrument Engineers' Handbook*, Third Edition. Chilton, Radnor, Pennsylvania.

Jung, M., M. Reichstein, P. Ciais et al. 2010. Recent deceleration of global land evapotranspiration due to moisture supply limitation. *Nature* 467: 951–954.

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

Karl, T.R., R.W. Knight, and N. Plummer. 1995. Trends in high-frequency climate variability in the twentieth century. *Nature* 377: 217–220.

Karl, T.R., J.M. Mellillo, and T.C. Peterson (eds). 2009. *Global Climate Change Impacts in the United States*. Cambridge University Press, Cambridge, United Kingdom.

Kaushal, S.S., G.E. Likens, N.A. Jaworski et al. 2010. Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment* 8: 461–466.

Kinouchi, T., H. Yaki, and M. Miyamoto. 2007. Increase in stream temperature related to anthropogenic heat input from urban wastewater. *Journal of Hydrology* 335: 78–88.

Kittel, T.G.F., J.A. Royle, C. Daly et al. 1997. A gridded historical (1895–1993) bioclimate dataset for the conterminous United States. *In: Proceedings of the 10th Conference on Applied Climatology*, American Meteorological Society, Boston, Massachusetts. pp. 219–222.

Krakauer, N.Y. and I. Fung. 2008. Mapping and attribution of change in streamflow in the conterminous United States. *Hydrology and Earth Systems Science* 12: 1111–1120.

Kunkel, K.E., D.R. Easterling, K. Hubbard et al. 2004. Temporal variations in frost-free season in the United States: 1895–2000. *Geophysical Research Letters* 31: L03201.

Labat, D., Y. Godderis, J.L. Probst et al. 2004. Evidence for global runoff increase related to climate warming. *Advances in Water Resources* 27(6): 631–642.

Lammertsma, E.I., H. Jan de Boer, S.C. Dekker et al. 2011. Global CO_2 rise leads to reduced maximum stomatal conductance in Florida vegetation. *Proceedings of the National Academy of Sciences* 1100371108v1–201100371.

Laseter, S.H., C.R. Ford, J.M. Vose et al. 2012. Long-term temperature and precipitation trends at the Coweeta Hydrologic Laboratory, Otto, North Carolina, USA. *Hydrology Research,* 43: 890–901.

La Torre Torres, I., D.M. Amatya, T.J. Callahan et al. 2011. Seasonal rainfall-runoff relationships in a lowland forested watershed in the southeastern U.S.A. *Hydrological Processes* 25: 2032–2045.

Leopold, D.J., G.R. Parker, and W.T. Swank. 1985. Forest development after successive clearcuts in the Southern Appalachians. *Forest Ecology and Management* 13: 83–120.

Leuzinger, S. and C. Korner. 2007. Water savings in mature deciduous forest trees under elevated CO_2. *Global Change Biology* 13: 2498–2508.

Leuzinger, S. and C. Korner. 2010. Rainfall distribution is the main driver of runoff under future CO_2 concentration in a temperate deciduous forest. *Global Change Biology* 16: 246–254.

Liao, C., Y. Luo, C. Fang et al. 2010. Ecosystem carbon stock influenced by plantation practice: Implications for planting forests as a measure of climate change mitigation. *PLoS ONE* 5: e10867, doi:10.1371/journal.pone.0010867.t001.

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

Lins, H.F. and J.R. Slack. 2005. Seasonal and regional characteristics of US streamflow trends in the United States from 1940–1999. *Physical Geography* 26: 489–501.

Liu, Y.-Q. 2011. Hydrological impacts of forest restoration in the southern United States. *Ecohydrology* 4: 299–314, doi:10.1002/eco.178.

Liu, Y.-Q., J. Stanturf, and H. Lu. 2008. Modeling the potential of the northern China forest shelterbelt in improving hydroclimate conditions. *Journal of the American Water Resources Association* 44(5): 1176–1192.

Lockaby, G., C. Nagy, J.M. Vose et al. (in press). Water and Forests. *In:* D.N. Wear and J.G. Greis (eds). The Southern Forest Futures Project: Technical Report. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina. General Technical Report.

Lu, J., G. Sun, S.G. McNulty et al. 2009. Sensitivity of pine flatwoods hydrology to climate change and forest management in Florida, USA. *Wetlands* 29: 826–836.

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

Marion, D.A. and J.A. Clingenpeel. 2012. Methods used for analyzing the cumulative watershed effects of fuel management on sediment in the eastern United States. *In*: LaFayette, R., M.T. Brooks, J.P. Potyondy, L. Audin, S.L. Krieger, and C.C. Trettin (eds). *Cumulative Watershed Effects of Fuel Management in the Eastern United States. Forest Service*, Southern Research Station, Asheville, North Carolina. General Technical Report SRS-161: 308–326.

Matthews, K.R. and N.H. Berg. 1997. Rainbow trout responses to water temperature and dissolved oxygen stress in two southern California stream pools. *Journal of Fish Biology* 50(1): 50–67, doi: 10.1111/j.1095-8649.1997.tb01339.x.

McCabe, G.J. and D.M. Wolock. 2002. A step increase in streamflow in the conterminous United States. *Geophysical Research Letters* 29: 2185.

McNulty, S.G., J.A. Moore Myers, P. Caldwell et al. (in press). Climate Change. *In*: D.N. Wear and J.G. Greis (eds). *The Southern Forest Futures Project: Technical Report*. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina. General Technical Report.

Meisner, J.D. 1990. Effect of climatic warming on the southern margins of the native range of brook trout, *Salvelinus fontinalis*. *Canadian Journal of Fisheries and Aquatic Science* 47(6): 1065–1070.

Milly, P.C.D., J. Betancourt, M. Falkenmark et al. 2008. Stationarity is dead: Whither water management. *Science* 319: 573–574.

Mississippi Department of Environmental Quality. 2008. *Sediment TMDL for the Yalobusha River Yazoo River Basin*. Mississippi Department of Environmental Quality, Jackson, Mississippi.

Mohseni, O. and H.G. Stefan. 1999. Stream temperature air temperature relationship: A physical interpretation. *Journal of Hydrology* 218(3–4): 128–141.

Mohseni, O., H.G. Stefan, and T.R. Erickson. 1998. A nonlinear regression model for weekly stream temperatures. *Water Resources Research* 34(10): 2685–2692.

Mote, T.L. and J.M. Shepherd. 2011. Monthly ensemble downscaled climate model data for the Southeast U.S. http://shrmc.ggy.uga.edu/downscale.php. Accessed: November 07, 2011.

Murdoch, P.S. and J.S. Baron. 2000. Potential effects of climate change on surface-water quality in North America. *Journal of the American Water Resources Association* 36: 347–366.

National Research Council of the National Academies. 2008. *Hydrologic Effects of a Changing Forest Landscape*. National Research Council of the National Academies, The National Academies Press, Washington, District of Columbia.

Nearing, M.A. 2001. Potential changes in rainfall erosivity in the U.S. with climate change during the 21st century. *Journal of Soil and Water Conservation* 56(3): 229–232.

Neilson, R.P., L.F. Pitelka, A. Solomon et al. 2005. Forecasting regional to global plant migration in response to climate change: Challenges and directions. *BioScience* 55: 749–759.

Nelson, K.C. and M.A. Palmer. 2007. Stream temperature surges under urbanization and climate change: Data, models, and responses. *Journal of the American Water Resources Association* 43(2): 440–452.

Neubauer, C.P., G.B. Hall, E.F. Lowe et al. 2008. Minimum flows and levels method of the St. Johns River Water Management District, Florida. *Environmental Management* 42(6): 1101–1114.

Noormets, A., M. Gavazzi, S.G. McNulty et al. 2010. Response of carbon fluxes to drought in a coastal plain loblolly pine forest. *Global Change Biology* 16: 272–287.

O'Gorman, P.A. and T. Schneider. 2009. The physical basis for increases in precipitation extremes in simulations of 21st-century climate change. *Proceedings of the National Academy of Sciences* 106: 14773–14777, doi:10.1073/pnas.0907610106.

Ollinger, S.V., C.L. Goodale, K. Hayhoe et al. 2008. Potential effects of climate change and rising CO_2 on ecosystem processes in northeastern US forests. *Mitigation and Adaptation Strategies for Global Change* 13: 467–485.

O'Neal, K. 2002. *Effects of Global Warming on Trout and Salmon in US Streams*. Defenders of Wildlife, Washington, District of Columbia.

Pacala, S. and R. Sokolow. 2004. Stabilization wedges: Solving the climate problem for the next 50 years with current technologies. *Science* 305: 968–972, doi:10.1126/science.1100103.

Peterson, J.T. and T.J. Kwak. 1999. Modeling the effects of land use and climate change on riverine smallmouth bass. *Ecological Applications* 9(4): 1391–1402.

Phillips, D.L., D. White, and C.B. Johnson. 1993. Implications of climate change scenarios for soil erosion potential in the United States. *Land Degradation and Rehabilitation* 4: 61–72.

Poole, G.C. and C.H. Berman. 2001. An ecological perspective on in-stream temperature: Natural heat dynamics and mechanisms of human-caused thermal degradation. *Environmental Management* 27(6): 787–802.

Postel, S. and S. Carpenter. 1997. Freshwater ecosystem services. *In:* G.C. Daily (ed). *Nature's Services: Societal Dependence on Natural Ecosystems*. pp. 195–214. Island Press, Washington, District of Columbia.

Postel, S. and B. Richter. 2003. *Rivers for Life: Managing Water for People and Nature*. Island Press, Washington, District of Columbia. 220 p.

Prentice, I.C. and S.P. Harrison. 2009. Ecosystem effects of CO_2 concentration: Evidence from past climates. *Climate of the Past* 5: 297–307.

Pruski, F.F. and M.A. Nearing. 2002a. Climate-induced changes in erosion during the 21st century for eight U.S. locations. *Water Resources Research* 38(12): 1298, doi:10.1029/2001WR000493.

Pruski, F.F. and M.A. Nearing. 2002b. Runoff and soil loss responses to changes in precipitation: A computer simulation study. *Journal of Soil and Water Conservation* 57(1): 7–16.

Pyzoha, J.E., T.J. Callahan, G. Sun et al. 2007. A conceptual hydrologic model for a forested Carolina bay depressional wetland on the Coastal Plain of South Carolina, USA. *Hydrologic Processes* 22(14): 2689–2698, doi:10.1002/hyp.68.

Reilly, C.F. and C.N. Kroll. 2003. Estimation of 7-day, 10-year low-streamflow statistics using baseflow correlation. *Water Resources Research* 39: 1236.

Renard, K.G. and V.A. Ferreira. 1993. RUSLE model description and database sensitivity. *Journal of Environmental Quality* 22: 458–466.

Renard, K.G. and J.R. Freimund. 1994. Using monthly precipitation data to estimate the R factor in the revised USLE. *Journal of Hydrology* 157: 287–306.

Renard, K.G., G.R. Foster, G.A. Weesies et al. 1997. *Predicting Soil Erosion by Water: A Guide To Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). Agriculture Handbook* Number 703. US Department of Agriculture, Washington, District of Columbia.

Rosenblatt, F. 1958. The perceptron: A probabilistic model for information storage and organization in the brain. *Psychological Review* 65: 386–408.

Ryan, M.G., M.E. Harmon, R.A. Birdsey et al. 2010. A synthesis of the science on forests and carbon for US forests. *Issues in Ecology* 13: 1–16.

Sassaman, R.W. 1981. Threshold of concern: A technique for evaluating environmental impacts and amenity values. *Journal of Forestry* 79: 84–86.

Savenije, H.G. 2004. The importance of interception and why we should delete the term evapotranspiration from our vocabulary. *Hydrological Processes* 18: 1507–1511.

Schwartz, M.D. and J.M. Hanes. 2010. Continental-scale phenology: Warming and chilling. *International Journal of Climatology* 30: 1595–1598.

Shelby, J.D., G.M. Chescheir, R.W. Skaggs et al. 2005. Hydrologic and water quality response of forested and agricultural lands during the 1999 extreme weather conditions in eastern North Carolina. *Transactions of the ASAE* 48: 2179–2188.

Sheridan, J.M. 2002. Peak flow estimates for coastal plain watersheds. *Transactions of the ASAE* 45(5): 1319–1326.

Shields, F.D., Jr., C.M. Cooper, S. Testa III et al. 2008. Nutrient transport in the Yazoo River Basin. Research Report 60. US Department of Agriculture, Agricultural Research Service, National Sedimentation Laboratory, Oxford, Mississippi.

Slack, J.R., A.M. Lumb, and J.M. Landwehr. 1993. Hydro-Climatic Data Network (HCDN) streamflow data set, 1874–1988. US Geological Survey, Water Resources Investment Report 93–4076.

Smakhtin, V.U. 2001. Low flow hydrology: A review. *Journal of Hydrology* 240: 147–186.

St. Johns River Water Management District. 2010. St. Johns River Water Management District, Chapter 40c-8, F.A.C. Minimum Flows and Levels. St. Johns River Water Management District, Palatka, Florida.

Stoy, P.C., G.G. Katul, M.B.S. Siqueira et al. 2006. Separating the effects of climate and vegetation on evapotranspiration along a successional chronosequence in the southeastern US. *Global Change Biology* 12: 2115–2135.

Sullivan, K., J. Tooley, K. Doughty et al. 1990. *Evaluation of Prediction Models and Characterization of Stream Temperature Regimes in Washington*. Timber/Fish/Wildlife Rep. No. TFW-WQ3-90-006. Washington Department of Natural Resources, Olympia, Washington. 224 pp.

Sun, G., K. Alstad, J. Chen et al. 2011a. A general predictive model for estimating monthly ecosystem evapotranspiration. *Ecohydrology* 4: 245–255, doi:10.1002/eco.194.

Sun, G., D.M. Amatya, S.G. McNulty et al. 2000a. Climate change impacts on the hydrology and productivity of a pine plantation. *Journal of the American Water Resources Association* 36(2): 367–374.

Sun, G., P. Caldwell, A. Noormets et al. 2011b. Upscaling key ecosystem functions across the conterminous United States by a water-centric ecosystem model. *Journal of Geophysical Research* 116: G00J05, doi:10.1029/2010JG001573.

Sun, G., S.G. McNulty, D.M. Amatya et al. 2002. A comparison of the watershed hydrology of coastal forested wetlands and the mountainous uplands in the southern US. *Journal of Hydrology* 63: 92–104.

Sun, G., S.G. McNulty, J.A. Moore Myers et al. 2008. Impacts of multiple stresses on water demand and supply across the southeastern United States. *Journal of the American Water Resources Association* 44: 1441–1457.

Sun, G., A. Noormets, M. Gavazzi et al. 2010. Energy and water balances of two contrasting loblolly pine plantations on the lower coastal plain of North Carolina, USA. *Forest Ecology and Management* 259: 1299–1310.

Sun, G., H. Riekerk, and L.V. Kornhak. 2000b. Ground-water table rise after forest harvesting on cypress-pine flatwoods in Florida. *Wetlands* 20(1): 101–112.

Sun, G., M. Riedel, R. Jackson et al. 2004. Influences of management of southern forests on water quantity and quality. *In*: H.M. Rauscher and K. Johnsen (eds). *Southern Forest Sciences: Past, Current, and Future*. pp. 195–226. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina. Gen. Tech. Rep SRS-75.

Swift, L.W. 1982. *Duration of Stream Temperature Increases Following Forest Cutting in the Southern Appalachian Mountains*. International Symposium on Hydrometeorology, Denver, Colorado.

Swift, L.W., Jr. 1973. Lower water temperatures within a streamside buffer strip. US Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, North Carolina. Res. Note SE-193.

Swift, L.W., Jr. and J.B. Messer. 1971. Forest cuttings raise temperatures of small streams in the Southern Appalachians. *Journal of Soil and Water Conservation* 26(3): 111–116.

Tague, C., L. Seaby, and A. Hope. 2009. Modeling the eco-hydrologic response of a Mediterranean type ecosystem to the combined impacts of projected climate change and altered fire frequencies. *Climatic Change* 93: 137–155.

Telis, P.A. 1992. Techniques for estimating 7-day 10-year low flow characteristics for ungauged sites on streams in Mississippi. US Geologic Survey. Water-Resources Investigations Report 91–4130.

Tian, S., M.A. Youssef, R.W. Skaggs et al. 2011. Modeling water, carbon and nitrogen dynamics for two drained pine plantations under intensive management practices. *Forest Ecology and Management* 264(2012): 20–36.

Trenberth, K.E., P.D. Jones, P. Ambenje et al. 2007. Observations: Surface and atmospheric climate change. Chapter 3 *In:* S. Solomon, D. Qin, M. Manning et al. (eds). *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, New York. http://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch3.html. Accessed: November 1, 2012.

Trimble, S.W., F.H. Weirich, and B.L. Hoag. 1987. Reforestation and the reduction of water yield on the Southern Piedmont since circa 1940. *Water Resources Research* 23(3): 425–437, doi:10.1029/WR023i003p00425.

US Department of Agriculture. 2006. *Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin.* US Department of Agriculture, Natural Resource Conservation Service, Washington, District of Columbia. Agriculture Handbook 296.

US Department of Agriculture, Forest Service. 2005. Draft environmental impact statement for the proposed revised land and resource management plan: Ouachita National Forest. US Department of Agriculture. Management Bulletin R8-MB 124 C.

US EPA. 2010. BASINS 4.0 (Better Assessment Science Integrating Point & Non-Point Sources) Description. http://water.epa.gov/scitech/datait/models/basins/BASINS4 _index.cfm. Accessed: November 1, 2012.

van Vliet, M.T.H., F. Ludwig, J.J.G. Zwolsman et al. 2011. Global river temperatures and sensitivity to atmospheric warming and changes in river flow. *Water Resources Research* 47: W02544, doi:10.1029/2010WR009198.

Vörösmarty, C.J., P. Green, J. Salisbury et al. 2000. Global water resources: Vulnerability from climate change and population growth. *Science* 289: 284–288.

Walter, M.T., D.S. Wilks, J.-Y. Parlange et al. 2004. Increasing evapotranspiration from the conterminous United States. *Journal of Hydrometeorology* 5: 405–408.

Warren, J.M., E. Pötzelsberger, S.D. Wullschleger et al. 2011. Ecohydrologic impact of reduced stomatal conductance on forests exposed to elevated CO_2. *Ecohydrology* 4: 196–210.

Watson, T. 2009. Climate plan calls for forest expansion, USA Today. www.usatoday.com/news/nation/environment/2009–08-19-forest_N.htm. Accessed: November 1, 2012.

Wear, D.N. and J.G. Greis. 2002. Southern forest resource assessment: Summary report. Gen. Tech. Rep. SRS 53. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina. 103 p.

Webb, B.W. and F. Nobilis. 1995. Long term water temperature trends in Austrian rivers. *Hydrological Sciences Journal* 40: 83–96.

Webb, B.W. and F. Nobilis. 2007. Long-term changes in river temperature and the influence of climatic and hydrological factors. *Hydrological Sciences Journal* 52(1): 74–85.

Webb, B.W. and D.E. Walling. 1993. Longer-term water temperature behavior in an upland stream. *Hydrological Processes* 7(1): 19–32.

Webb, B.W., D.M. Hannah, R.D. Moore et al. 2008. Recent advances in stream and river temperature research. *Hydrological Processes* 22(7): 902–918, doi: 10.1002/hyp.6994.

West, B. 2002. Water quality in the South. *In*: D.N. Wear and J.G. Greis (eds). *Southern Forest Resource Assessment*. pp. 455–476. US Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina. General Technical Report SRS-54.

White, E. 2011. Trends in heavy precipitation over the southern region. *Southern Climate Monitor* 1(7): 2–7.

Whitehead, P.G., R.L. Wilby, R.W. Battarbee et al. 2009. A review of the potential impacts of climate change on surface water quality. *Hydrological Sciences Journal* 54: 101–123.

Williams, T., D.M. Amatya, A. Jayakaran et al. 2012. Runoff generation from shallow water table southeastern forests: Unusual behavior of paired watersheds following a major disturbance. ASABE Paper # 121337045, St. Joseph, Michigan: ASABE. http://elibrary.asabe.org/azdez.asp?JID=5&AID=42110&CID=dall2012&T=2. Accessed: June 26, 2013.

Williams, T.M. 1978. Response of shallow water tables to rainfall. *In*: W.E. Balmer (ed). *Proceedings: Soil Moisture, Site Productivity Symposium*. pp. 363–370. US Department of Agriculture, Forest Service, Southeastern Area, State and Private Forestry.

Wischmeier, W.H. and D.D. Smith. 1965. *Predicting Rainfall Erosion Losses from Cropland East of the Rocky Mountains: Guide for Selection of Practices for Soil and Water Conservation*. US Department of Agriculture, Washington, District of Columbia. Agriculture Handbook No. 282.

Wischmeier, W.H. and D.D. Smith. 1978. *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*. US Department of Agriculture, Washington, District of Columbia. Agriculture Handbook No. 537.

Wooldridge, D.D. and D. Stern. 1979. Relationships of silvicultural activities and thermally sensitive forest streams. University of Washington, College of Forest Resources, Seattle, Washington. Rep. DOE 79-5a-5.

Young, C.E. and R.A. Klawitter. 1968. Hydrology of wetland forest watersheds. *In:* F.C. Alley, K. Lehotsky, and J.T. Ligon (eds). *Proceedings of Hydrology in Water Resources Management*. pp. 29–38. Water Resources Research Institute Report #4. Clemson University, Clemson, South Carolina.

Zhang, X., D. Tarpley, and J.T. Sullivan. 2007. Diverse responses of vegetation phenology to a warming climate. *Geophysical Research Letters* 34: L19405.

Zhao, M. and S.W. Running. 2010. Drought-induced reduction in global terrestrial net primary production from 2000 through 2009. *Science* 329: 940–943.