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# Southern Forest Outlook **WATER**

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# WATER

## An Analysis to Support the Southern Forest Outlook

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# Abstract

The linkages among forests, water, and people have long been recognized by natural resources professionals and policymakers. This report, in support of the U.S. Department of Agriculture, Forest Service, Southern Research Station's *Southern Forest Outlook*, provides a contemporary assessment of current and future forest and water resource issues in the Southern United States, a region encompassing the 13 Southern States from Virginia to Texas. This report uses a combination of literature review, hydrologic modeling, and data synthesis to address pressing research questions, leveraging scenarios of future climate, land use, and forest conditions developed as part of the 2020 Resources Planning Act Assessment. Results show that climate and land-use change have and will continue to alter streamflow regimes and water quality through the 21<sup>st</sup> century, with consequences for drinking water treatment costs, flood protection, and other ecosystem services. These impacts will present broad-scale and complex challenges for land and water resources managers. Fortunately, increasing evidence suggests that in many cases, management actions can be effective at helping forests adapt to climate and land-use change, which can ensure the availability and quality of water resources in the future.

**Keywords:** Climate change, water yield, water supply, ecosystem services, land-use change, disturbance interactions, management actions.



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## Preface

The *Southern Forest Outlook* (SFO) adds to the regional landscape-level Southern Forest Assessment library. The South has a long history of regional assessments to address uncertainties within the forestry sector. Wheeler published *The South's Third Forest* in 1970, which evaluated trends and synthesized the peer-reviewed literature to examine the future of timber supply given increasing demand and concerns of underinvestment in private forest land. That report was followed by *The South's Fourth Forest* in 1988 (USDA Forest Service 1988), which focused on re-evaluating many of the same questions. However, research for *The South's Fourth Forest* relied on using timber market models and policy analyses to examine alternatives for reversing underinvestment in nonindustrial private forests. By the late 1990s, growth in forest management and timber production largely anticipated by those reports, coupled with the emergence of satellite chip mills, raised concerns about the sustainability of forests in the South, which motivated the *Southern Forest Resource Assessment* (SFRA) in 2002 (Wear and Greis 2002). *The Southern Forest Futures Project* (SFFP) (Wear and Greis 2013) was completed in 2013; it updated and expanded the analysis conducted for the SFRA. The SFFP was driven by increasing urbanization, land ownership dynamics, and markets and included integrated change scenarios based on population, economic factors, and climate change.

Analyses and results detailed in the Southern Forest Assessment library have been used to support planning and policy discussions for decades. *The South's Third Forest* identified strategies to encourage planting and increase management on private forests; protect forests from insects, diseases, and fires; and build stronger institutions for forestry training, technology transfer, and research. Its forecasts of population-driven urbanization and expansion of tree planting and timber production have been realized in the South. *The South's Fourth Forest* anticipated the growth in timber production realized through 2000 and informed a suite of programs and policies to encourage reforestation, management, and forest protection. The SFRA led to improved best management practices monitoring, helped to expand the Forest Inventory and Analysis Program, provided a wealth of information for the first State Forest Action Plans, and led to broader



engagement across the forestry community. Findings from the SFFP helped set research priorities at the U.S. Department of Agriculture, Forest Service, Southern Research Station; provided motivation for a high priority regional working forest conservation initiative known as Keeping Forests; and increased emphasis on understanding socioecological systems related to water resources.

In 2019, the Forest Service and Southern Group of State Foresters (SGSF) leadership recognized a need to update the Southern Forest Assessment library with a new regional assessment—the SFO. Recent changes in land use, forest disturbances, forest management, and forest product markets have led to increasing uncertainty about the future of the sector. Though much of the assessment library remains relevant, new data, research results, and projection models allow for new insights into critical issues facing the South.

The SFO is a four-part synthesis of the following issues: (1) timber market conditions and futures, (2) water and forest interactions, (3) fire in a changing ecological and social landscape, and (4) forest dynamics and land-use change. These issue areas each leverage the work of the 2020 Resources Planning Act Assessment in terms of overall drivers (climate, population, and income), land-use change, forest condition (dynamics), and markets. The Southern Research Station and Southern Region, along with the SGSF, led the SFO with the objective of informing forest sector decisionmakers and the interested public about observed trends, anticipated futures, and critical issues based on authoritative synthesis and interpretation of existing science, data, and 50-year projections.

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## Key Messages

- Projected changes in annual and monthly streamflow varied among and within States. There is general agreement among scenarios in the projected direction of change in streamflow for some parts of the South. Portions of Texas, Oklahoma, Arkansas, and Louisiana may be most vulnerable to climate-driven streamflow reductions.
- Future projections suggest that the observed historical trends in climatic precipitation may continue through the 21<sup>st</sup> century with corresponding changes in streamflow extremes under some scenarios. Historical observations suggest that increases in the magnitude and frequency of precipitation extremes (low and high) have occurred in the South since the 1980s. Compound dry-hot extremes have also become more frequent.
- Projected land-use change will alter streamflow regimes and water quality in developing watersheds, with consequences for drinking water treatment costs, flood protection, and other ecosystem services.
- Changes in hydrologic conditions due to climate and land-use change will likely have complex feedbacks and interactions with other disturbances such as wildfire, drought, sea-level rise, hurricanes, nonnative invasive plants, and insect and disease outbreaks. In some cases, these interactions could exacerbate hydrologic changes due to climate and land-use change alone.
- More extreme precipitation events present challenges for water quality. Identification of areas that contribute most to nonpoint source pollution can help prioritize implementation of forest best management practices that have been widely used in the Southern United States.
- Many mitigation and adaptation management actions used today will likely benefit water quantity, including prescribed burning, thinning, and eradication of invasive species.



# Introduction

The linkages among forests, water, and people have long been recognized by natural resource professionals and policymakers in the United States (National Research Council 2008).

Among the primary purposes of the Organic Administration Act (1897) was

*securing favorable conditions of water flows, and to furnish a continuous supply of timber for the use and necessities of citizens of the United States.*

The subsequent Weeks Act (1911) was prompted, at least in part, by the expressed need to protect and restore degraded forest watersheds in the Eastern United States (Vose et al. 2019). These Acts were followed by decades of research in Federal agencies, universities, research institutes, the forest products industry, and State forestry agencies to develop best management practices (BMPs) to protect and restore water resources, while managing forests for multiple uses. Much of this research has been summarized in international (Amatya et al. 2016, Creed and van Noordwijk 2018), national (Ice and Stednick 2004, NCASI 2009), and regional syntheses (Jackson et al. 2004, Sun and Vose 2016).

Over the past 2 decades, researchers and public and private land managers in the Southern United States partnered to conduct detailed analyses of contemporary and future forest and water resource issues as part of a larger multiresource analysis (Wear and Greis 2002, 2013). These assessments were unique in that they were driven by questions and concerns identified in public forums, listening sessions, and written comments. In the most recent assessment, *The Southern Forest Futures Project* (SFFP), the following key findings were presented in the water resources chapter (Lockaby et al. 2013):

- Forest conversion to agriculture or urban use consistently causes increased discharge, peak flow, and velocity of streams. Subregional differences in hydrologic responses to urbanization are substantial.
- Sediment, water chemistry indices, pathogens, and other substances often become more concentrated after forest conversion to other land

uses. If the conversion is to an urban use, the resulting additional increases in discharge and concentrations will produce even higher loads.

- Although physiographic characteristics such as slope and soil texture play key roles in hydrologic and sediment responses to land-use conversion, land use (rather than physiography) is the primary driver of water chemistry responses.
- Conversion of forest land to urban uses may decrease the supply of water available for human consumption and increase potential threats to human health.
- Increases in urbanization by 2060 in the Appalachians, Piedmont, and Coastal Plain will increase imperviousness and further reduce hydrologic stability and water quality indices in the headwaters of several major river basins and small watersheds along the Atlantic Ocean and Gulf of Mexico.
- On average, water supply model projections indicate that water stress due to the combined effects of population and land-use change will increase in the Southern United States by 10 percent by 2050.
- Water stress will likely increase significantly by 2050 under four climate change scenarios, largely because higher temperatures would result in more water loss by evapotranspiration and because of decreased precipitation in some areas.
- Approximately 5,000 miles (17 percent) of the southern coastline is highly vulnerable to sea-level rise.

Many of the issues identified in the SFFP remain relevant today; however, our understanding of forest water interactions has increased since the SFFP was published, and new challenges and stronger rationale for protecting forests and water have emerged locally and globally (Creed and van Noordwijk 2018). For example, though the importance of forest cover to water quantity, quality, and supply has long been recognized (e.g., Anderson et al. 1976), new studies have linked forested lands with drinking water intakes to quantify the relationship more directly at national



and regional scales (e.g., Liu et al. 2022). Further, additional analyses have included the expected impacts of land-use change, urbanization, climate change, and their interactions on water resources (Caldwell et al. 2012; Li, C. et al. 2020; Martin et al. 2017). Though increases in streamflow associated with greater impervious cover may mitigate some of the climate-driven decreases in water yield (Li, C. et al. 2020), increases in demand for drinking water and reduced water quality may offset those benefits if the excess water can be captured and utilized (Gorelick et al. 2020). Hence, forest conversion to other land uses represents a continuing, and in some cases an accelerating, threat to water resources—and future forest loss is likely to exacerbate those threats. Recent research suggests that watersheds comprised of large amounts of highly fragmented, privately owned forests are most vulnerable to urbanization (Riitters and Costanza 2019) and have the greatest sensitivity to climate change (Martin et al. 2017), whereas watersheds with a larger proportion of protected forest land are less vulnerable to climate change (Suttles et al. 2018). Understanding how to keep these fragmented forests healthy, resilient, and intact represents an ongoing challenge for both land and watershed managers.

Even in areas where forest land area is stable or increasing, new evidence points to how changing disturbance regimes pose increasing threats to water resources in the South. Disturbances such as wildfire, drought, and hurricanes (Jayakaran et al. 2014, Worley et al. 2022) can have direct and indirect effects on water resources (Caldwell et al. 2020a, Hallema et al. 2018, Vose et al. 2016, Zhang et al. 2022), and the frequency and severity of these disturbances are expected to increase in the future due—at least in part—to climate change (Bedel et al. 2013, Mitchell et al. 2014, Vose et al. 2018). Furthermore, recent studies have disentangled how multiple climate disturbances, in combination with other stressors that are responding to those disturbances (e.g., wildfire, insects, pathogens)—in some places—create stress complexes that may overwhelm the inherent resiliency of forests and challenge effective management responses to mitigate impacts and maintain ecosystem services (Vose et al. 2018).

Trends in measured climate variables suggest that many of what were once projected changes in future climate are already observable. For example, the past 2 decades have been the warmest on record for the Southeastern United States, with a large part of that warming being driven by higher average nighttime temperatures (Carter et al. 2018). Recent climate projections suggest increasing hydrologic intensification in the future (Langner et al. 2020) with potentially significant consequences for the environment and people. For example, an increase in the magnitude and frequency of heavy precipitation events, tidal storm surges, and sea-level rise has increased the risk of inland and coastal flooding (Carter et al. 2018). More extreme heat, dry spells, and flooding pose direct health threats to humans, but also threaten the natural and built infrastructure that provides important services, such as drinking water.

It is increasingly recognized that active forest management will be instrumental in keeping forest watersheds healthy and resilient in the face of current and future stressors, providing sustained water-related ecosystem services (McNulty et al. 2021, Sun and Vose 2016, Sun et al. 2017) (fig. 1). Knowledge gained from a combination of new experiments, analyses of long-term paired watershed studies, eddy covariance and sapflow measurements, and computer simulations (Sun et al. 2023) will be critical for management efforts. For example, improvements in understanding of how ecohydrological responses to climate change vary by species are due in large part to a greater number of sapflow studies, which are now prolific enough to populate a global network of sapflow data (e.g., SAPFLUXNET, Poyatos et al. 2016). In addition, recent analyses of long-term paired watersheds indicate that increases in the abundance of mesophytic species (species such as red maple [*Acer rubrum*] and tulip poplar [*Liriodendron tulipifera*]) after disturbance (Elliott et al. 2017, Jackson et al. 2018) and long-term succession (Caldwell et al. 2016) can reduce watershed-scale streamflow through increased evapotranspiration. New insights have also been obtained from a growing network of eddy covariance towers in forest ecosystems in the South. Like the sapflow network, an eddy covariance network (e.g., Ameriflux) facilitates

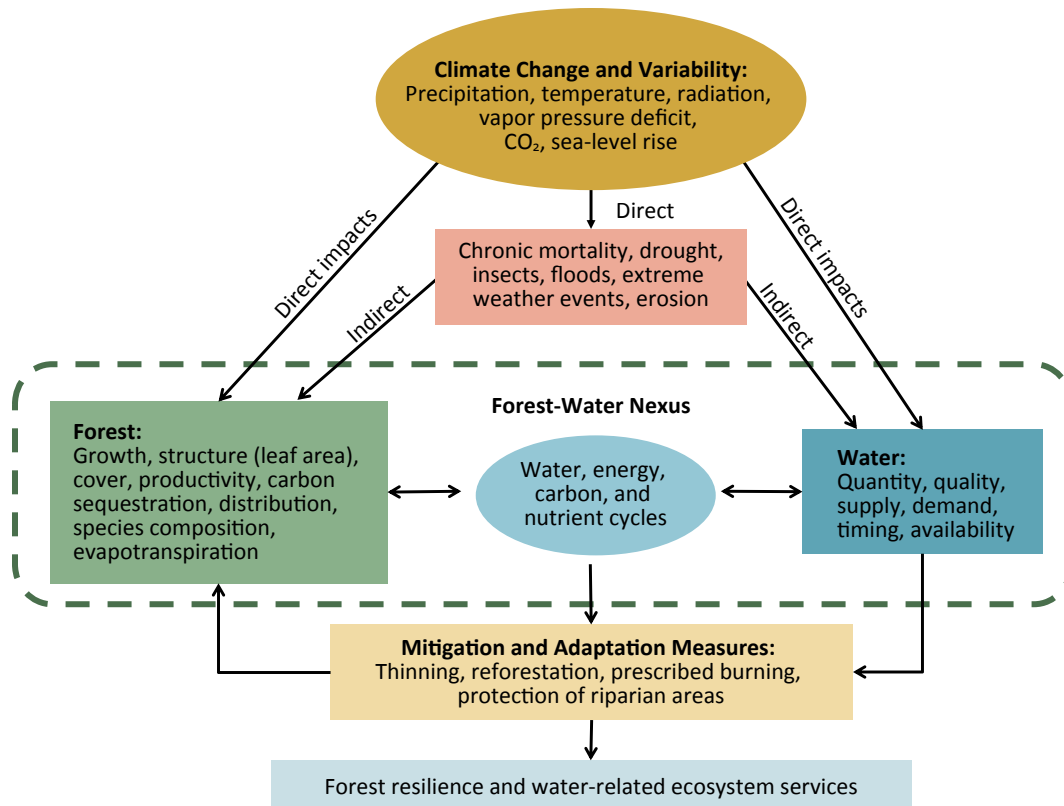


Figure 1—Illustration of the processes and interactions among climate change, forests, and water resources and climate change mitigation and adaptation measures. Adapted from Zhang et al. (2022).

understanding of coupled carbon and water cycling processes among different ecosystems and biophysical conditions. For example, eddy covariance data have helped quantify differences in evapotranspiration between pine and hardwood stands in the Piedmont and improved understanding of how forest structure (e.g., age, leaf area, rooting depth, and more) regulates evapotranspiration in coastal pine stands (Aguilos et al. 2021). Furthermore, though some recent studies suggest that managed pine plantations (i.e., those that receive regular forest management operations such as thinning, species selection, herbicide and fertilizer application, etc.) may be more vulnerable to drought than unmanaged pine stands (those that do not receive regular forest management operations) or hardwood forests (Domec et al. 2015), the results of a regionwide throughfall exclusion study (PINEMAP) suggest that drought vulnerability in loblolly pine (*Pinus taeda*) plantations is highly site specific (Will et al. 2015). Hence, in some regions, loblolly pine may be more vulnerable to drought than other species. In

contrast, other studies have indicated that managed loblolly pine is more resilient to drought-related mortality than unmanaged stands of loblolly pine or other species (Klockow et al. 2020a). It is likely that further insights will be gained as new sensors and analytical tools reveal new patterns of responses and ecohydrological processes. Examples of such technologies include emerging techniques to estimate evapotranspiration using high resolution remote sensing (Liu et al. 2021, Yang et al. 2020) and machine learning using large datasets (Guswa et al. 2020).

As noted above, knowledge and science-based tools are available to help provide the necessary information to maintain watershed function and continue provisioning of water-related ecosystem services into the future; however, integrating new science findings and tools into planning and management actions presents a challenge for both researchers and natural resource managers. To help meet this challenge, we used a research question approach, with questions formulated from the Southern Forest Resource Assessment and SFFP

public comments, input from State and national forestry agencies, and discussion with the Southern Group of State Foresters Water Resources Committee. Draft research questions were shared with the Water Resources Committee and revised; these revised questions were submitted for public comment and finalized based on all feedback received. The iterative processes identified four questions that frame the sections in this report (fig. 2):

**Q1:** How will changes in climate, land uses, and forest conditions impact water resources in the future?

**Q2:** How might these projected changes in climate and water resources interact with other disturbances such as wildfire, drought, sea-level rise, hurricanes, invasive plants, and insect and disease outbreaks?

**Q3:** What are the likely implications of these changes on water supply, flood protection, and other water-related ecosystem services?

**Q4:** What management actions best minimize impacts of and increase resiliency to climate change and human population increase?

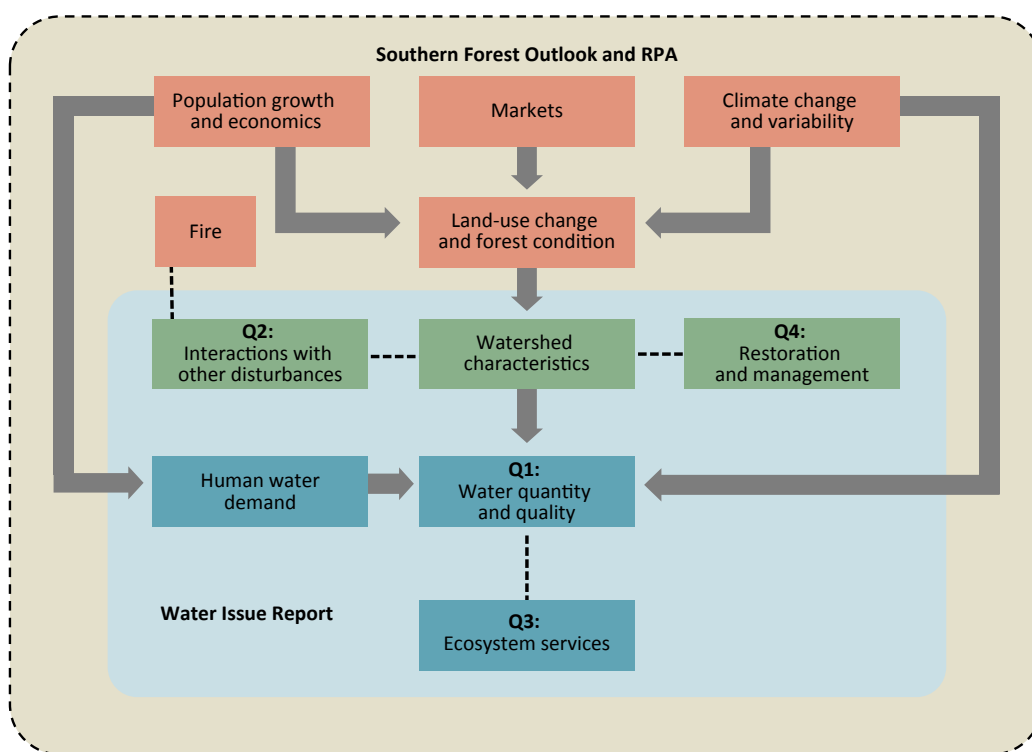


Figure 2—Functional diagram showing this report (light blue area) and the questions (Q1–4) that it addresses within the broader context of the *Southern Forest Outlook* (tan area). Orange boxes are data and analyses originating from the 2020 Resources Planning Act (RPA) Assessment or other *Southern Forest Outlook* reports, green boxes are elements included in this report pertaining to changes on the land that affect water resources, and blue boxes are elements in this report pertaining to changes in water-related drivers, condition, and ecosystem services. Solid grey arrows signify data streams that serve as inputs to models used to project changes in water quantity and quality. Dashed lines are informational linkages between elements that are discussed in this report, but not directly used in model projections.

# Methods

We used a combination of a literature review, hydrologic modeling, and data synthesis to address the four research questions. This report leverages scenarios of future climate, land use and forest conditions developed and described elsewhere in the *Southern Forest Outlook* (SFO) and as part of the U.S. Department of Agriculture, Forest Service 2020 Resources Planning Act (RPA) Assessment (fig. 2). Many of these projections served as inputs in our integrated hydrological modeling approach to address research question 1. The projections also provided context for the other three research questions, which were addressed primarily through a literature review. We used these modeling tools in combination with the RPA scenarios to project future long-term annual and monthly changes in the water balance and water supply. We reviewed and summarized literature describing potential future changes in submonthly extreme events (e.g., extreme precipitation, drought, high and low flows) that could not be assessed with our monthly time-step modeling approach. In addition to modeling streamflow, we used projections of county-level freshwater consumptive use developed by Warziniack et al. (2022) for the RPA scenarios and compared those projections to modeled water supply projections to identify areas of potential water supply stress. The consumptive use estimates include water use from both surface and groundwater sources, and project water use annually from the 2015 base year to 2070 for domestic, industrial and mining, thermoelectric power, irrigated agriculture, livestock, and aquaculture water use sectors.

## Water Balance and Streamflow Modeling Overview

We used the Water Supply Stress Index modeling framework (WaSSI) (Caldwell et al. 2012, Sun et al. 2011b, Wear and Greis 2013) to integrate future projections and investigate potential effects on water resources in the South between 1981 and 2070. WaSSI was developed by the Forest Service, Southern Research Station and has been extensively tested, validated, and compared

to other hydrological models (Caldwell et al. 2012, 2015, 2020b; Li, C. et al. 2020; Schwalm et al. 2015; Sun et al. 2011a, b, 2015a). We focused on understanding responses at the annual and monthly scales to examine the implications of both wet and dry conditions.

Precipitation was partitioned into rain and snow in WaSSI using an air temperature-based conceptual snow accumulation and melt model (McCabe and Wolock 1999). The WaSSI model calculates monthly infiltration, surface runoff, soil moisture, and baseflow for each watershed land-use type using algorithms of the Sacramento Soil Moisture Accounting Model (Burnash et al. 1973, Burnash 1995). The soil profile is divided into a relatively thin upper layer and a much thicker lower layer that supplies moisture to meet evapotranspiration demands (Koren et al. 2003). Each layer consists of tension water storage (i.e., between soil water tensions of field capacity and the plant wilting point) and free water storage (i.e., soil water tension greater than field capacity) that interact to generate surface runoff, lateral water movement from the upper soil layer to the stream (interflow), percolation from the upper soil layer to the lower soil layer, and lateral water movement from the lower soil layer to the stream (baseflow). Monthly evapotranspiration is calculated as a function of potential evapotranspiration (Hamon 1963), precipitation, and leaf area index (LAI) by using empirical relationships derived from multisite eddy covariance measurements (Sun et al. 2011a, b). Storage and evapotranspiration for impervious cover in each watershed were assumed to be negligible; thus, all precipitation falling on the impervious portion of a watershed was assumed to be runoff and was routed directly to the watershed outlet. Though this assumption may overestimate the effect of impervious cover on water yield in some locations, it was necessary because we could not be certain about the amount of water storage on impervious surfaces or their connectivity to surface water at the regional scale. Water yield was calculated for each land-use type in a given watershed as the sum of surface runoff from pervious and impervious surfaces, interflow, and



baseflow after accounting for losses that include changes in water storage in the soil, evaporation, and transpiration from vegetation. Water yield for each watershed was then calculated as the sum of the area-weighted averages of water yield of each land-use type present and expressed in depth units of mm/year. Water yield for each watershed was then routed and accumulated from upstream to downstream watersheds along the river network to estimate the surface water supply at the outlet of each respective watershed. The surface water supply was the sum of the water yield generated in all watersheds upstream of a given location on the river network and expressed in volume units of m<sup>3</sup>/year.

## Model Modifications for This Assessment

WaSSI was implemented at the 12-digit (sixth-level) hydrologic unit code (HUC12) watershed scale. There are 22,093 HUC12s in the South with a mean area of 100 km<sup>2</sup> per HUC12. We computed monthly evapotranspiration for each land-use type as a function of Food and Agriculture Organization (FAO) reference potential evapotranspiration (Allen et al. 1994), precipitation, and LAI by using empirical relationships derived from multisite eddy covariance measurements (Sun et al. 2011a) and the FAO reference potential evapotranspiration estimates available with the climate projections (Abatzoglou and Brown 2012). In addition to the HUC12s in the South, watersheds upstream of the region and within the United States (e.g., upper Mississippi River, Missouri River, and others) as well as watersheds outside the United States on the Mexico side of the Rio Grande basin and the Canada side of the Missouri River basin were included in our modeling approach. It was necessary to account for the water supply contribution of areas outside the South so that we would not underestimate water supply volume and overestimate percentage of change in water supply in affected watersheds in the South. Those HUC12 watersheds within the United States, but outside the South were simulated using climate and land-use projections; however, variables that relate forest condition projections to the water balance (i.e., forest LAI) were assumed stationary at 2010–2016 levels because forest condition projections were not available outside the South. The effects of this assumption were limited

primarily to HUC12s on the mainstems of the Rio Grande/Pecos River, Canadian/Arkansas River, Ohio River, and Mississippi River (<2 percent of all HUC12s in the South). HUC12 watersheds in Mexico and Canada were not simulated using climate, land-use, or forest condition projections because those projections were not available for areas outside the United States. For watersheds outside the United States, we included the 2001–2015 mean monthly water yield from Liu et al. (2022) in the streamflow calculations for HUC12s in the South in all years of model simulations and thus the water contribution to streamflow from those watersheds was assumed to be stationary for future projections. The effects of this assumption were limited primarily to HUC12s that were on the mainstem of the Rio Grande and Mississippi Rivers (<1 percent of all HUC12s in the South).

## Scenarios

We used the WaSSI model to project potential changes in water resources at the monthly time step between 1981 and 2070 under multiple scenario-climate model combinations (table 1). The scenarios combine varying assumptions about socioeconomics and atmospheric warming. In this manner, a range of plausible potential futures can be evaluated. The SFO relies on the same core scenarios as the 2020 RPA Assessment. Within the RPA Assessment, a set of core scenarios was developed from the Intergovernmental Panel on Climate Change (IPCC) scenarios. The IPCC provides a global context for a range of potential futures, with the climate futures defined in terms of differing representative concentration pathways (RCPs) (IPCC 2014). The RCPs are defined by radiative forcing (W/m<sup>2</sup>). In simple terms, radiative forcing is the difference between energy being absorbed by the Earth and energy being reflected and emitted back into space. When the balance is greater than zero, warming occurs. For the 2020 RPA Assessment, RCP 4.5 and RCP 8.5 were selected. RCP 4.5 has a radiative forcing of 4.5 W/m<sup>2</sup> at stabilization (after year 2100), with a carbon dioxide (CO<sub>2</sub>)-equivalent concentration of approximately 650 ppm. RCP 8.5 has a radiative forcing of more than 8.5 W/m<sup>2</sup> in 2100, with a CO<sub>2</sub>-equivalent concentration of >1370 ppm (Moss et al. 2010).

**Table 1—Characteristics of the four 2020 Resources Planning Act Assessment scenarios**

| Scenario characteristic                      | Scenario                                           |                                              |                                                   |                                               |
|----------------------------------------------|----------------------------------------------------|----------------------------------------------|---------------------------------------------------|-----------------------------------------------|
|                                              | LM<br>Lower warming<br>and moderate<br>U.S. growth | HL<br>High warming<br>and low<br>U.S. growth | HM<br>High warming<br>and moderate<br>U.S. growth | HH<br>High warming<br>and high<br>U.S. growth |
| Global warming and U.S. socioeconomic growth | Low                                                | High                                         | High                                              | High                                          |
| Global real GDP growth, 2020–2070            | Medium (4.9x)                                      | Low (3.2x)                                   | Medium (4.6x)                                     | High (6.9x)                                   |
| Global population growth, 2020–2070          | Low <sup>1</sup> (1.2x)                            | High (1.6x)                                  | Medium (1.4x)                                     | Low (1.2x)                                    |
| U.S. real GDP growth, 2020–2070              | Medium (3.0x)                                      | Low (1.9x)                                   | Medium (2.8x)                                     | High (4.7x)                                   |
| U.S. population growth, 2020–2070            | Medium (1.5x)                                      | Low (1.0x)                                   | Medium (1.4x)                                     | High (1.9x)                                   |
| Global emissions                             | Lower                                              | High                                         | High                                              | High                                          |
| Technology change                            | High                                               | Low                                          | Moderate                                          | High                                          |
| Bioenergy preferences                        | High                                               | Low                                          | Moderate                                          | High                                          |
| Trade openness                               | Moderate                                           | Low                                          | Moderate                                          | High                                          |
| Global scenario links                        | RCP 4.5–SSP1                                       | RCP 8.5–SSP3                                 | RCP 8.5–SSP2                                      | RCP 8.5–SSP5                                  |

LM = lower warming-moderate economic growth; HL = high warming-low economic growth; HM = high warming-moderate economic growth; HH = high warming-high economic growth. Numbers in parentheses are the factors of change in the projection period. For example, U.S. real gross domestic product increases by a factor of 3.0 between 2020 and 2070 in scenario LM. GDP = gross domestic product based on estimates by the International Institute of Applied Systems Analysis (2019).

<sup>1</sup> Low population involves initial increase with declines in the later decades of the projection period.

Source: Langner et al. (2020).

Scientists are challenged to align socioeconomic paths that are consistent with RCP paths (van Vuuren et al. 2014). Shared socioeconomic pathways (SSPs) were developed after the RCPs, and the SSPs offer economic and demographic storylines at a global scale that generally align with the RCP levels (O'Neill et al. 2014). The four SSPs used in the RPA Assessment (SSP1, SSP2, SSP3, and SSP5) include a wealthy and increasingly economically equal world that has a focus on climate change mitigation and slowing population growth (SSP1), an economically wealthy world with more focus on adaptation and high U.S. population growth but slower global population growth (SSP5), a less wealthy world with continued inequality and low U.S. population growth and high global population growth (SSP3), and an intermediate case intended to roughly replicate current rates of economic and population change (SSP2).

The RPA scenarios are based on combining RCPs and SSPs (fig. 3). Langner et al. (2020) provides a description of the scenarios for the United States. These scenarios were also evaluated as part of the SFO. The LM scenario (low warming-moderate U.S. growth) is based on the combination of RCP 4.5 and SSP1 and describes a future with medium growth (3.0 times current) in U.S. real gross domestic

product (GDP), medium population growth (1.5 times current), and lower global emissions. The HL scenario, combining RCP 8.5 and SSP3, suggests low U.S. population growth (1.0 times current) and low growth in U.S. real GDP (1.9 times), but high emissions (table 1). The HH scenario, the combination of RCP 8.5 and SSP5, describes a future with high warming and high U.S. socioeconomic growth: U.S. real GDP increases 4.7 times current levels, and population growth increases by 1.9 times. Under the HH scenario, these high levels of GDP and population growth are coupled with

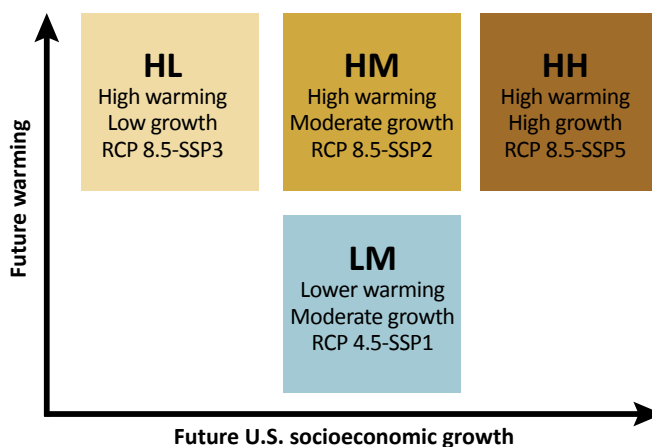


Figure 3—Scenarios developed for the 2020 Resources Planning Act. Source: USDA Forest Service (2023: 3-3).

high global emissions. Between the HL and HH scenarios is the HM scenario (RCP 8.5 and SSP2), which has high global emissions and, consequently, high warming. Under the HM scenario, U.S. growth is moderate: a 2.8 times increase in U.S. real GDP and a 1.4 times increase in U.S. population growth. More information about these scenarios is available in Langner et al. (2020).

Wear and Prestemon (2019) downscaled U.S. population and income projections to the county scale. Based on those projections the Southern U.S. population is expected to increase between 1.5 (SSP1) and 1.9 (SSP5) times, which suggests that population growth in the South is expected to outpace the rest of the country. Though downscaled GDP projections were not available from Wear and Prestemon (2019), their per capita income projections suggest increases of between 1.7 (SSP3) and 2.4 (SSP5) times in the South.

RCPs were designed to capture the range in projected radiative forcing ( $\text{W/m}^2$ ) found in the scientific literature. Radiative forcing is also used as an input into general circulation models (GCMs), which are used to model future climates. There are numerous competing GCMs; for the RPA Assessment and for the SFO, five of those GCMs were selected to pair with each scenario. The five GCMs can generally be classified into least warm, hot, dry, wet, and middle climate futures for the United States (table 2).

It is important to note that the GCM classification into least warm, hot, dry, wet, and middle is based on long-term average departures (1971–2000 versus 2041–2070) over the entire United States. This classification is relevant to the South (fig. 4). In the South, under the least warm model, average annual temperature is projected to increase by between 1.1 °C (RCP 4.5) and 2.75 °C (RCP 8.5). The hot model suggests average annual temperature will increase by between 2.6 and 3.9 °C. Under the dry model, average annual precipitation will decrease by between 9 and 13 percent. The wet model suggests a future where average annual precipitation will increase by between 7 and 8 percent. The middle climate model represents a future where the departures are relatively close to the average departures for mean annual temperature and precipitation.

The four different RPA scenarios (table 1) are combined with the five different climate models (table 2) to produce 20 different scenario-climate futures. When specified, these are denoted with a RPA scenario-climate model format. These scenario-climate futures represent a broad range of socioeconomic futures, warming futures, and temperature and precipitation futures. Together, these futures create an envelope under which the forests of the Southern United States might exist.

**Table 2—General Circulation Models used with each Resources Planning Act Assessment scenario**

|                           | Least warm                               | Hot                                      | Dry                                   | Wet                                                | Middle                           |
|---------------------------|------------------------------------------|------------------------------------------|---------------------------------------|----------------------------------------------------|----------------------------------|
| General circulation model | MRI-CGCM3                                | HadGEM2-ES                               | IPSL-CM5A-MR                          | CNRM-CM5                                           | NorESM1-M                        |
| Source institution        | Meteorological Research Institute, Japan | Met Office Hadley Centre, United Kingdom | Institut Pierre Simon Laplace, France | National Centre of Meteorological Research, France | Norwegian Climate Center, Norway |

Source: Langner et al. (2020).

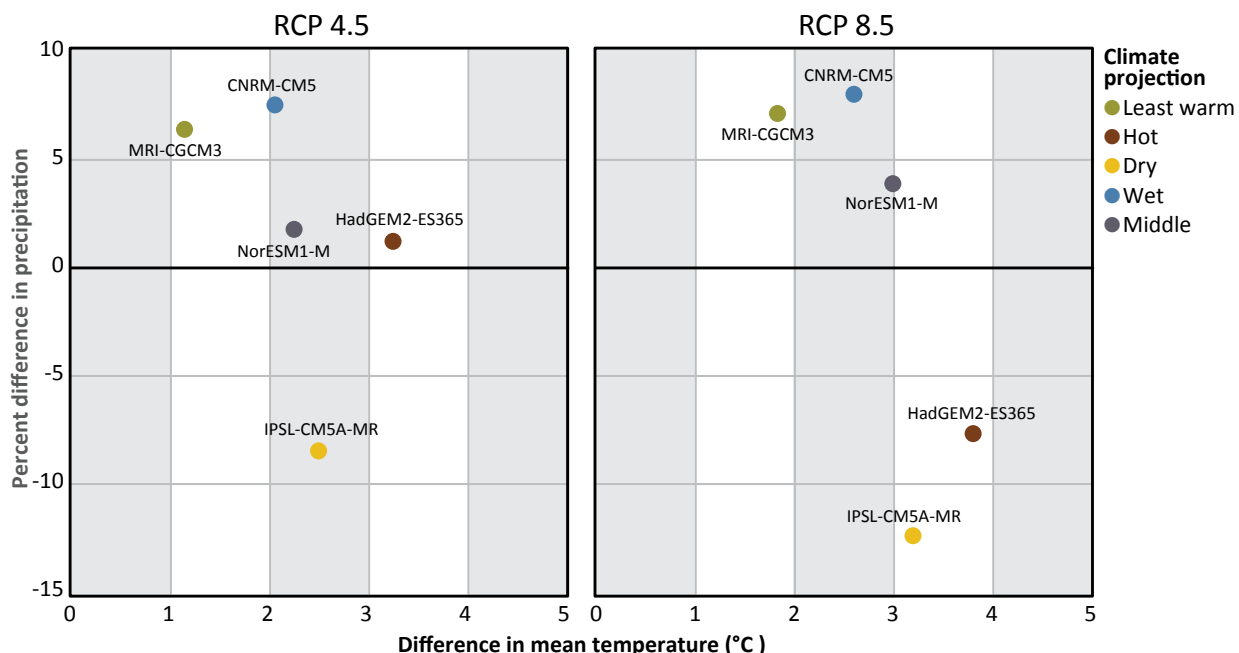


Figure 4—Comparison of the climate projections used in the *Southern Forest Outlook* for two representative concentration pathways (RCP, emissions future), showing differences in mean annual temperature and percent differences in mean total annual precipitation between the 1971–2000 and 2041–2070 time periods in the U.S. South. Model names corresponding to the projections are listed under each colored dot. The descriptive names of the projections are in the legend. Source: Joyce and Coulson (2020: 84).

## Interpretation of Scenarios

Scenarios are not predictions of potential future conditions, but rather a range of plausible socioeconomic and climate conditions (Langner et al. 2020) that can be used to assess how systems may respond to different rates of change or how different decisions may alter resource trajectories (Langner et al. 2020, Moss et al. 2008). Scenarios are not assigned likelihoods, nor are any scenarios intended to be accurate per se (Langner et al. 2020). As designed, the scenarios address uncertainty in RCP, SSP, and the climate system but not uncertainty in the linkage between forests and water. The latter is incorporated into the WaSSI modeling approach and is informed by decades of forest hydrological science. In addition to projected changes over time among scenarios, projected changes over space carry uncertainty and should not be viewed in

a probability context nor as a prediction with a certain level of accuracy. This spatial uncertainty can increase with increasing spatial resolution (Brooks et al. 2020), for example when downscaling from the county to a fine-scale grid.

## Model Inputs

The WaSSI model is parameterized using national-scale soil, land-use, and climate data (table 3). All input datasets were spatially rescaled using an area-weighted averaging scheme to match the scale of analysis (i.e., HUC12 watershed scale). Land use, impervious cover, and forest LAI projections for each month or year were calculated by interpolating between available decadal projections. All inputs were used as-is without modification, except for projections of impervious cover and forest LAI, which were developed as part of this study.



**Table 3—Data inputs for the Water Supply Stress Index (WaSSI) model**

| Data or database                                                  | Source                                                                                                                                                                                                                   | Resolution     | Time period |
|-------------------------------------------------------------------|--------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|----------------|-------------|
| Watershed boundaries                                              | WBD                                                                                                                                                                                                                      | HUC12          | N/A         |
| Soil properties                                                   | State Soil Geographic-based Sacramento Soil Moisture Accounting Model soil parameters, National Oceanic and Atmospheric Administration, National Weather Service, Office of Hydrologic Development, Hydrology Laboratory | 1 x 1 km grid  | N/A         |
| Climate (monthly precipitation, temperature, and PET)             | Downscaled MACA MACAv2-METDATA (Abatzoglou and Brown 2012): HadGEM2-ES365, MRI-CGCM3, CNRM-CM5, IPSL-CM5A-MR, and NorESM1-M GCMs under the 4.5 and 8.5 RCPs                                                              | 4 x 4 km grid  | 1981–2070   |
| Land use (crop, developed, forest, other natural, pasture, water) | Spatial realizations of county level land-use projections, case 10 (Brooks et al. 2023)                                                                                                                                  | 90 x 90 m grid | 2020–2070   |
| Impervious cover within developed land use                        | See impervious cover projection development section below                                                                                                                                                                | HUC12          | 2020–2070   |
| Monthly mean LAI by land use for nonforest land use <sup>1</sup>  | MODIS                                                                                                                                                                                                                    | 1 x 1 km grid  | 2006–2015   |
| Monthly mean LAI for forest land use                              | See forest LAI projection development section                                                                                                                                                                            | 90 x 90 m grid | 2020–2070   |

WBD = watershed boundary dataset; HUC12 = 12-digit hydrologic unit code; PET = potential evapotranspiration; MACA = multivariate adaptive constructed analogs; GCM = general circulation model; RCP = representative concentration pathway; LAI = leaf area index; MODIS = moderate resolution imaging spectroradiometer.

<sup>1</sup> LAI for nonforest land-use types (i.e., developed, crop, pasture, other natural) held at 2006–2015 monthly mean values for future projections. Thus, projected changes in evapotranspiration and effects on water yield from land-use change associated with these land-use types are a function of climate and changing land-use distribution only. LAI effects for these land-use types are not represented in model simulations.

WBD: <https://www.usgs.gov/core-sciencesystems/ngp/national-hydrography/watershed-boundary-dataset>.

MODIS: <https://modis.gsfc.nasa.gov/>.

## Impervious Cover Projections

The observed fraction of impervious cover within developed land use was calculated by first overlaying the 2019 National Land Cover Database impervious fraction (Dewitz 2021) pixels with developed land use from the RPA 2020 basemap (Brooks et al. 2023) to estimate the fraction of impervious cover within RPA developed land use in 2020. We then fit empirical beta regression models to the 2020 fraction of impervious cover and used those models to project the future developed fraction impervious cover using socioeconomic variables, land use, and climate change under the 20 scenario-climate futures. We first estimated regression models at the county scale using county-level independent variables, then incorporated the information predicted from the county-level model, along with spatial variables summarized by HUC, to predict average developed impervious fraction at the HUC scale.

County-scale models were estimated because the form of development, including the amount of impervious cover within developed land use, is in part a result of regulations and restrictions related to zoning and building, which differ by jurisdiction. In addition, we wanted to incorporate socioeconomic variables (population and income) into the empirical models, and those variables for 2020 and future time steps were only available by county and not at a finer spatial resolution (e.g., HUC12). In the county-level models, independent variables included population, income, population per area of developed land use, and proportion of forest, crop, pasture, and other land uses in each county. Independent variables in HUC-level models included elevation, slope, the average impervious fraction of developed land in the county in which the majority of the HUC falls, the proportion of each land use in each HUC, and landscape metrics describing the spatial pattern of land uses within the HUC (Riitters et al. 2009) derived from the RPA 2020 basemap (Brooks et al. 2023).

Separate county- and HUC-level models were fit for each RPA subregion (Langner et al. 2020) that overlapped the WaSSI modeling domain. By RPA subregion for both county- and HUC-level models, we used a model averaging approach, examining the full set of models that used all possible combinations of predictor variables and retaining the models within two Akaike Information Criterion (AIC) units from the model with the lowest AIC value. We used the fitted models to predict decadal fraction of impervious cover within developed land from 2020 to 2070 for each of the 20 scenario-climate futures based on projections of land use, socioeconomic variables, and landscape metrics. The final time series of projected impervious fraction were computed by adjusting the observed HUC-average 2020 impervious fraction by the difference between the future and 2020 modeled values.

## Forest LAI Projections

Monthly average forest leaf area index (LAI) projections by decade for all HUC12s in the South were based on results from the Forest Dynamics Model (FDM) which projects the Forest Service, Forest Inventory and Analysis (FIA) database forward based on the scenarios and climate models (Coulston et al. 2023). Specifically, hardwood and softwood biomass projections informed monthly LAI projections. To leverage FIA data and the FDM projections we (1) summarized FIA plot data at the HUC12 scale, and (2) translated mean HUC12 hardwood and softwood biomass to monthly mean HUC12 LAI on a decadal basis consistent with land use and forest condition projections.

We used spatial imputation methods similar to those of Wilson et al. (2012) to downscale the circa 2010–2016 FIA data so it could be summarized to the HUC12 scale. We then used regression modeling

approaches to translate 2010–2016 HUC12 biomass to monthly mean HUC12 LAI. We used the monthly MODIS LAI product (NASA 2024) in the regression model to calculate the mean 2006–2015 monthly forest LAI for each HUC12; this served as the response variable. We used Fourier regression to construct forest LAI-based phenology curves for each watershed as one explanatory variable based on the MODIS LAI summaries. We then used linear regression to estimate monthly forest LAI by HUC12 as a function of mean HUC12 softwood biomass, and the interaction of mean HUC12 hardwood biomass and the HUC12-specific phenology curve shape parameters. The resulting model had an adjusted  $R^2$  of 0.99.

By combining the imputation techniques offered by Wilson et al. (2012), the land-use projections techniques described by Brooks et al. (2020) with the FDM results for biomass, and summarizing those results to the HUC12 scales we had the requisite information to use the forest LAI models. The resulting projections of decadal mean HUC12 monthly forest LAI accounted for shifts in softwood and hardwood biomass (shifts in species composition), disturbance effects (natural and human mediated), and land-use change. Because of data and modeling limitations, phenology shifts associated with climate change were not included in the LAI models. The final time series of decadal monthly mean forest LAI for each HUC12 in the South were computed by adjusting the observed 2006–2015 MODIS-based forest LAI (assumed to approximate forest LAI in 2020, the base land use and forest condition year) by the difference between the future modeled and 2006–2015 modeled values. Forest LAI in HUC12s outside but draining to the South were held at 2006–2015 observed MODIS values because forest biomass was not projected outside the region.

# Results and Discussion

## (1) How Will Changes in Climate, Land Uses, and Forest Conditions Impact Water Resources in the Future?

### Projected changes in annual and monthly water yield and streamflow

The water yield response to climate and land-use change for the South varied across the four scenarios and five climate models, with future water yield projected to increase in some scenarios and climate models and decrease in others (fig. 5A). This variation was not unexpected, because the climate and land-use change scenarios were purposefully selected to represent a range of plausible futures that affect water yield. For example, southwide annual water yield was projected to increase by between 3.1 and 9.4 percent in 2041–2070 compared to 1981–2010 across all scenarios with the least warm

and wet climate futures and was projected to decrease by between 4.7 and 29.6 percent across all scenarios with the dry and middle climate futures. In general, there was greater variability in projected changes in annual water yield across climate futures than across scenarios (fig. 5A), suggesting that the effects of variability among climate futures were greater than the effects of emission and land-use scenario at the regional scale. Projected annual water yield was relatively insensitive to RCP for the wet, least warm, and middle climate futures, but was sensitive to RCP for the hot and dry climate futures. For example, annual water yield was projected to increase by 0.4 percent under the low warming scenario (LM) and decrease approximately 15 percent under the high warming scenarios (HL, HM, and HH) for

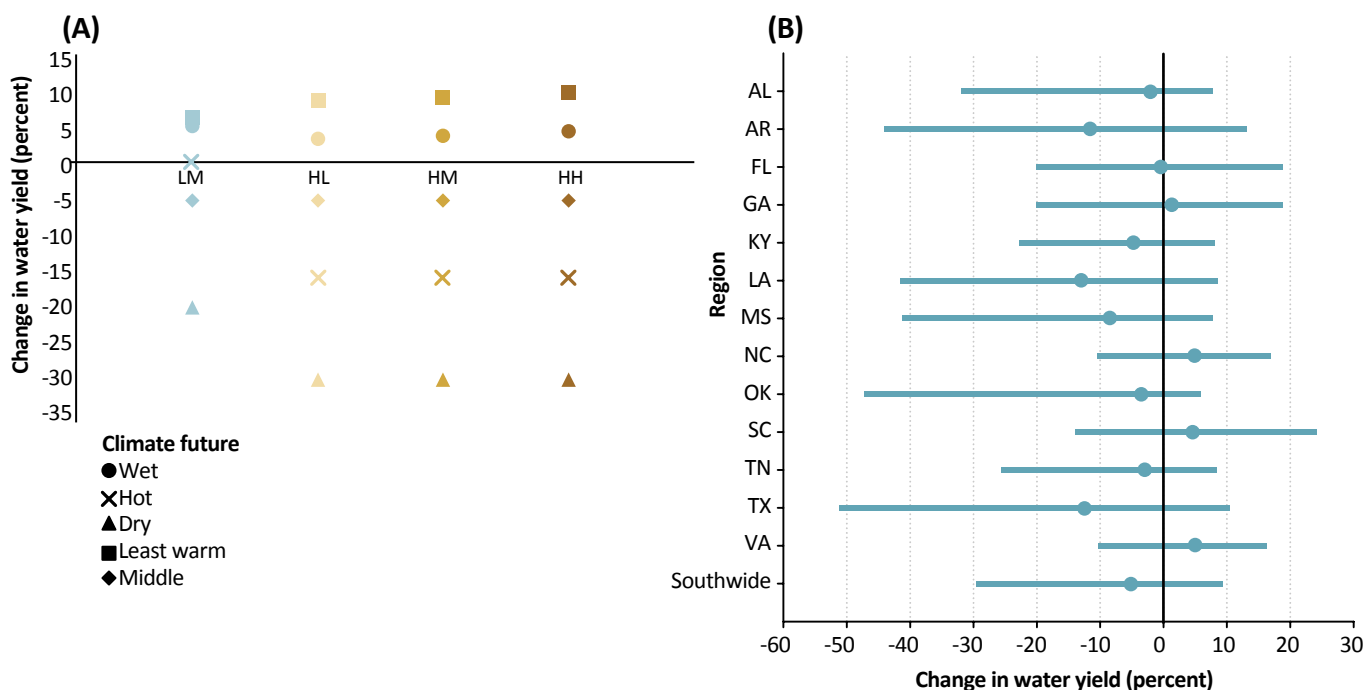


Figure 5—Projected change in southwide annual water yield in 2041–2070 compared to 1981–2010 by (A) scenario and (B) the range (bars) and median (dots) change in annual water yield across all scenario-climate futures by State. Scenarios in (A) are lower warming and moderate U.S. growth (LM), high warming and low U.S. growth (HL), high warming and moderate U.S. growth (HM), and high warming and high U.S. growth (HH) (tables 1 and 2). Projected changes in State-level annual water yield for individual scenario-climate futures are shown in [table A1 on page 58](#).

the hot climate future, yet there was virtually no difference in projected change among LM, HL, HM, and HH scenarios for the middle climate future. Across all scenario-climate futures, the median projected southwide change in annual water yield by 2041–2070 was a decrease of 5.3 percent (HM-middle), ranging from an increase of 9.4 percent (HH-least warm) to a decrease of 29.6 percent (HL-dry) (fig. 5A).

Projected changes in annual water yield and streamflow are not uniform across the South, rather they vary from State to State (fig. 5B). For example, the median change in projected annual water yield across scenarios by 2041–2070 relative to 1981–2010 in South Carolina is an increase of 4.3 percent under HH-wet and that of Texas was a decrease of 11.7 percent under HH-middle. Similar to results at the regional scale, projected changes in annual water yield vary by scenario-climate future, with at least one scenario-climate future projecting an increase and one scenario projecting a decrease in each State, and with larger variability across scenario-climate futures in some States than others. For example, projected changes in annual water yield in Texas ranged from a decrease of 51 percent to an increase of 10.4 percent, and changes in Virginia ranged from a decrease of 10.3 percent to an increase of 16.2 percent (fig. 5B).

The wide range in projected changes in streamflow and water yield across scenarios noted above makes it difficult to interpret or draw conclusions on potential impacts. We addressed this in two ways. First, we used the HM-middle scenario-climate future as an example that generally reflects the central tendency of potential future changes in water yield and streamflow. We also reported the most extreme scenario-climate futures when discussing low and high flows. Second, for some analyses, we summarized results where at least 80 percent of the scenario-climate futures were in agreement on the direction of change (i.e., increasing or decreasing). The results from all scenario-climate futures are provided in the appendix. Under HM-middle, the southwide 1981–2010 annual water yield was approximately 394 mm (15.5 inches), or 865 billion m<sup>3</sup> (228,566 billion gallons) per year and was projected to decrease by 16.1 billion m<sup>3</sup> (4,256 billion gallons, or 1.8 percent) per decade between 2011 and 2070 (fig. 6).

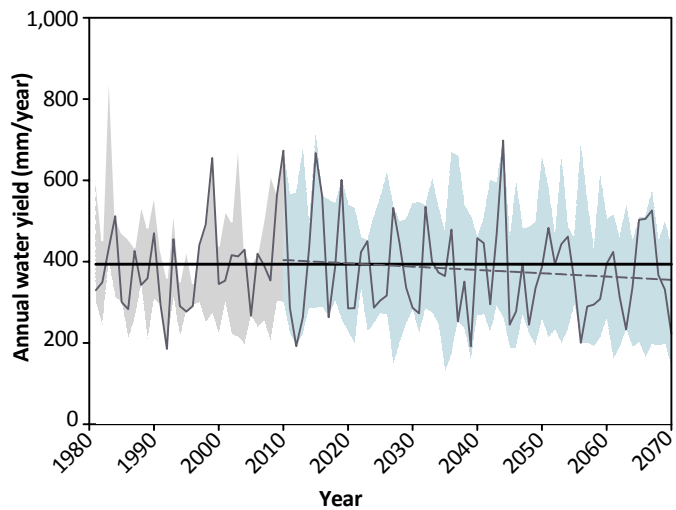


Figure 6—Southwide annual water yield under the HM-middle scenario-climate future (solid line) and range in projected annual water yield across scenario-climate futures (shaded area). Horizontal solid line is the 1981–2010 mean annual water yield, and the dashed line is the future trend in 2011–2070 mean annual water yield for HM-middle. Projected southwide changes in annual water yield for HM-middle are the median across all scenario-climate futures considered (tables 1 and 2). Similar charts for each of the scenario-climate futures can be found in [figure A1 on page 59](#).

Projected changes in streamflow (i.e., accumulated water yield for each subwatershed, including those upstream) also varied spatially within and among States (fig. 7A). For example, though projected changes in annual flow were variable across scenario-climate futures, there was general agreement in the direction of projected change across scenario-climate futures in some regions (fig. 7B). Annual streamflow was projected to decrease by 2041–2070 across eastern and central Texas and Oklahoma, southern Louisiana and Mississippi, northern Arkansas, and western Tennessee and Kentucky (fig. 7B). Annual flow was projected to increase across central and eastern North Carolina and Virginia (fig. 7B).

In addition to changes in annual streamflow, climate and land-use change were projected to affect the magnitude of annual low and high monthly flows in the future (fig. 8). The annual low monthly streamflow across scenarios by 2041–2070 was projected to decrease relative to 1981–2010 across much of the region under HM-middle (fig. 8A) and there was agreement among at least 80 percent of the scenario-climate futures in the direction of this change across 44.5 percent of the region (fig. 8B). Areas where the annual low monthly streamflow across scenario-climate futures was projected to increase were generally



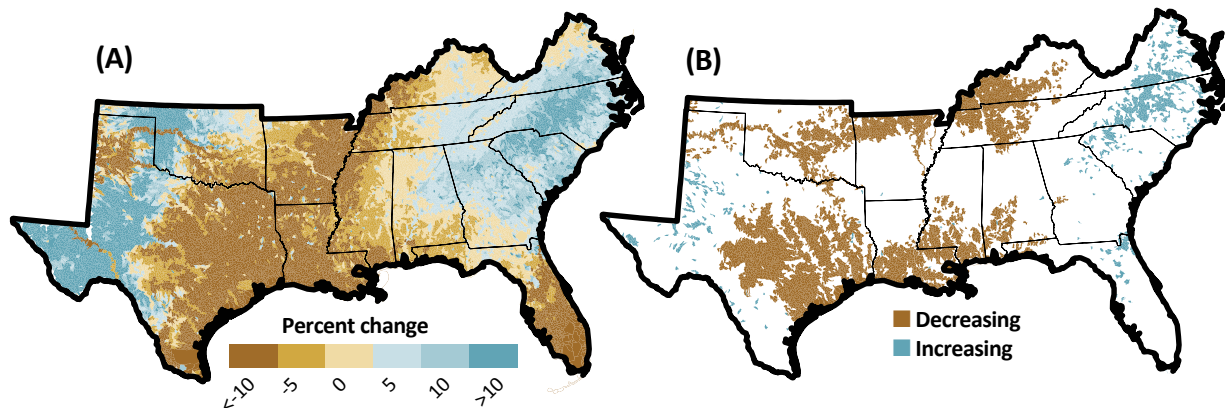


Figure 7—(A) Percent change in mean annual flow between 2041–2070 and 1981–2010 for the HM-middle scenario-climate future, and (B) areas where  $\geq 80$  percent of scenario-climate futures (tables 1 and 2) agree on the projected direction of change. Note that though projected changes in annual flow under HM-middle in (A) are the median changes for the South, this scenario-climate future may or may not be the median change in flow for individual States or watersheds. Maps similar to (A) for other scenario-climate futures can be found in [figure A2 on page 60](#).

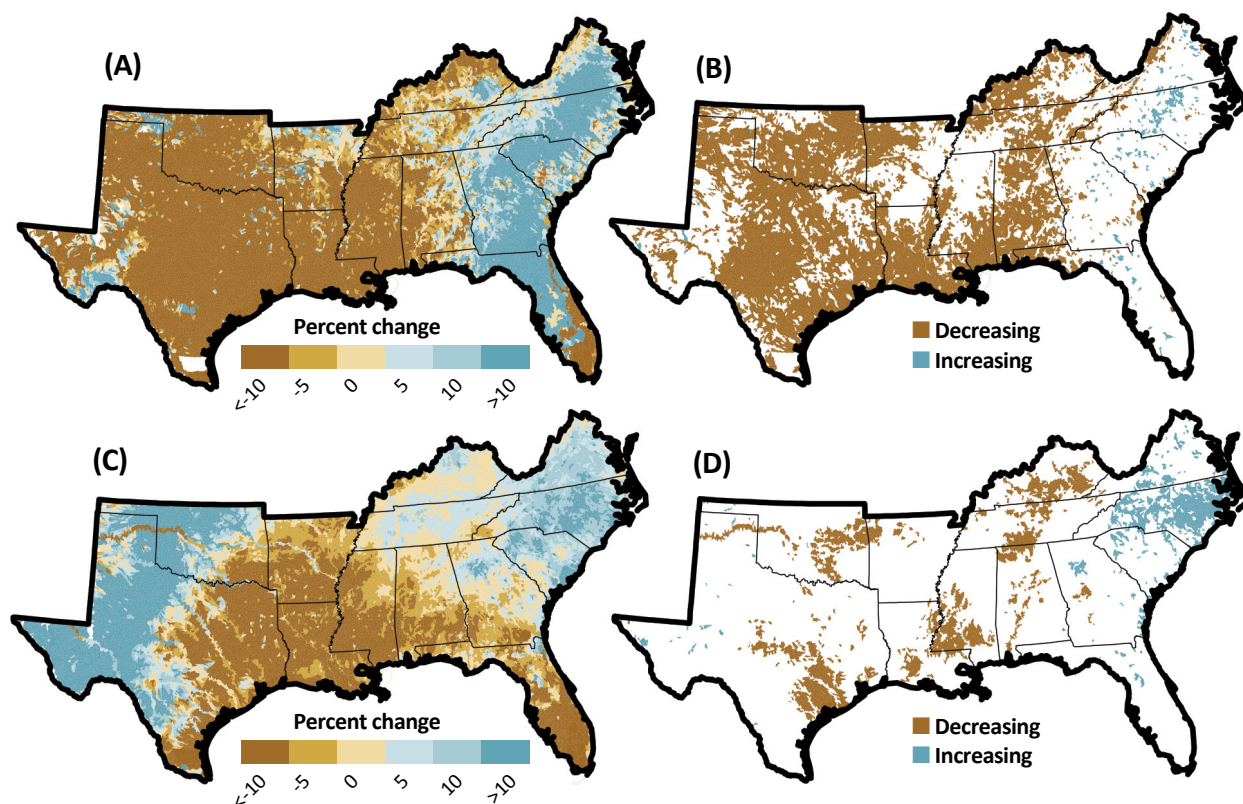


Figure 8—(A) Percent change in annual minimum monthly flow for 2041–2070 compared with 1981–2010 for the HM-middle scenario-climate future, (B) areas where  $\geq 80$  percent of the scenario-climate futures (tables 1 and 2) agree on the projected direction of change, (C) percent change in annual maximum monthly flow for 2041–2070 compared with 1981–2010 for HM-middle, and (D) areas where  $\geq 80$  percent of the scenario-climate futures agree on the projected direction of change. Note that though projected changes in minimum and maximum monthly flow under HM-middle in (A) and (B) are the median changes across scenario-climate futures for the South, this scenario-climate future may or may not be the median change in flow for individual States or watersheds. White areas in (A) had 1981–2010 mean annual minimum monthly flows equal to zero.



constrained to portions of central Florida and across the piedmont region of Georgia, South Carolina, and North Carolina, but there was agreement among scenario-climate futures in the direction of this change across only 1.3 percent of the region. The annual high monthly streamflow across scenarios by 2041–2070 was generally projected to decrease relative to 1981–2010 across much of the central and western portion of the South under HM-middle (fig. 8C), however there was agreement among scenario-climate futures in the direction of this change across only 9.2 percent of the entire region (fig. 8D). Annual high monthly streamflow was generally projected to increase in parts of western Texas and Oklahoma, Virginia, and South Carolina, and across North Carolina, however there was agreement in the direction of this change among scenario-climate futures across only 4.9 percent of the region, constrained mostly to North Carolina.

Of importance for water supply, aquatic ecosystems, and other water-related ecosystem services is the amount of time when there is low or no flow. By definition, perennial streams have flowing water year-round during a typical year (Messenger et al. 2021) but flow may cease even in perennial streams under extremely dry conditions. Though there are multiple quantitative definitions of perennial streams used in practice, we used a threshold of <5 percent of all months

over a 30-year period with zero flow that was compatible with our monthly time-step modeling approach. We used that threshold to identify potential streams that were perennial between 1981 and 2010, but not perennial by 2041–2070 under the southwide median HM-middle scenario-climate future and the southwide driest HL-dry scenario-climate future (fig. 9). Most of the streams that were projected to become nonperennial in the future were in Texas and Oklahoma, with some smaller streams or stream reaches becoming nonperennial along the Atlantic Coastal Plain. For example, the Clear Fork of the Trinity River at the Benbrook Lake dam southwest of Arlington, TX was classified as perennial between 1981 and 2010 (zero flow in 3.6 percent of months), but was projected to become nonperennial by 2041–2070 (14.2 percent of months with zero flow) under the HM-middle scenario-climate future. Benbrook Lake is a water source for the city of Arlington, a public water system serving 379,370 people. Similarly, the Colorado River at the Lake Travis dam northwest of Austin, TX was classified as perennial between 1981 and 2010 (zero flow 3.9 percent of months), but was projected to become nonperennial by 2041–2070 (11.9 percent of months with zero flow) under the HL-dry scenario-climate future. Lake Travis is a water source for the city of Austin, a public water system serving 973,338 people.

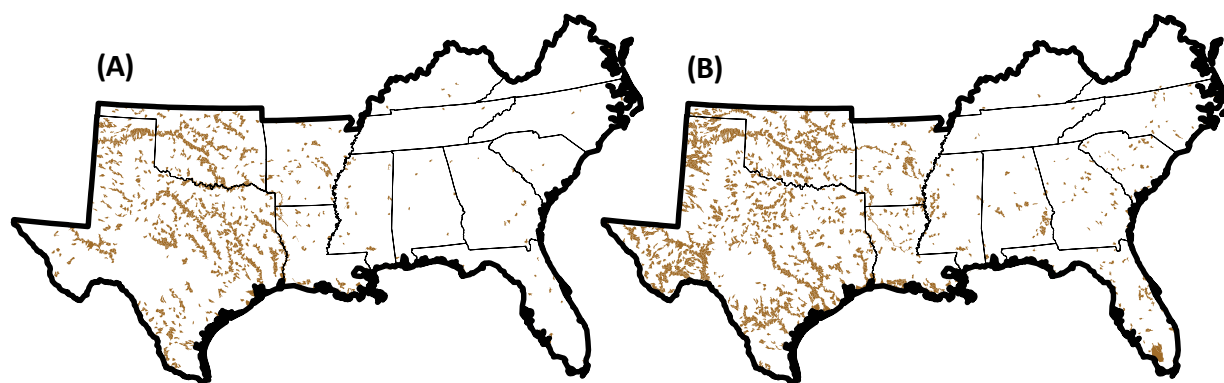


Figure 9—Watersheds with perennial flow (<5 percent of months with zero flow) between 1981 and 2010 that were projected to not have perennial flow by 2041–2070 under the (A) HM-middle, and (B) HL-dry scenario-climate futures (tables 1 and 2).

## Projected changes in water consumptive use

Total consumptive water use from both surface and groundwater sources across the South in 2015 was 58.6 billion m<sup>3</sup> (15,479 billion gallons) per year. The irrigation sector comprised 58 percent, domestic 21 percent, and industrial/commercial 10 percent of total consumptive use, with the remaining used by aquaculture, livestock, and thermoelectric sectors. Areas of the greatest consumptive water use include peninsular Florida, eastern Arkansas, southern Louisiana, and west and coastal Texas (fig. 10A). Regionwide consumptive water use was projected to increase in 2041–2070 from 2015 levels for all scenario-climate futures, ranging from an increase of 15 percent (9.2 billion m<sup>3</sup>/year) for LM-hot to an increase of 154 percent (89.6 billion m<sup>3</sup>/year) for HH-hot (median 50-percent increase under HL-middle). The thermoelectric sector water use is projected to have the largest relative increase (67 to 361 percent), followed by irrigation (3.6 to 195 percent), and domestic (-0.5 to 91 percent), depending on the scenario. The irrigation sector water is projected to have the largest increase in volume (1.2 to 66.6 billion m<sup>3</sup>/year), followed by domestic (-0.06 to 11.1 billion m<sup>3</sup>/year), and thermoelectric (1.0 to 5.3 billion m<sup>3</sup>/year). There was general agreement in the direction and magnitude of projected change in consumptive water use among scenario-climate futures, with >68 percent of counties having ≥80 percent of scenario-climate futures in agreement

and 99.8 percent of these were projected to have increases in consumptive use (fig. 10B). There were only two counties in the South where total consumptive water use was projected to decrease in ≥80 percent of scenario-climate futures. The largest projected percentage increases in consumptive water use were in east Texas, southern Louisiana, eastern Arkansas, and Florida, where consumptive use was also greatest in 2015 (fig. 10A). Some of these areas, including east Texas and southern Louisiana, are also projected to have decreases in annual streamflow (fig. 7C) and annual minimum monthly flow (fig. 7B) in 2041–2070. These results suggest potential increases in water supply stress in the future, when water demand could be limited by available supply or increased river flow depletion could occur with consequences for downstream users and aquatic ecosystem health.

## Changes in submonthly wet and dry extremes

Historical observations suggest increases in the magnitude and frequency of climate extremes over time in the South. Carter et al. (2018) showed that the mean number of days with precipitation above 3 inches (76 mm) has increased between 1900 and 2016. Much of that increase has occurred since the 1980s, and the variability in the mean number of days per year with precipitation above that level across the South has also increased (fig. 11A, B). The number of consecutive dry days has also increased from 1880 to 2010 (Roque-Malo and Kumar 2017)

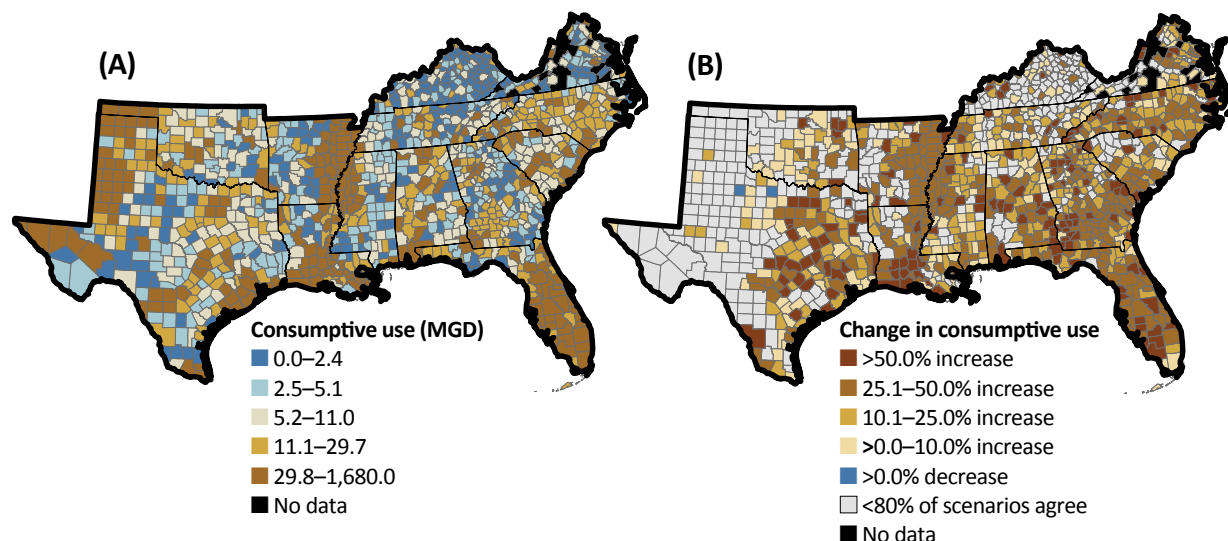


Figure 10—(A) County-level consumptive freshwater use from both surface and groundwater sources in 2015, and (B) counties where ≥80 percent of projections under all scenario-climate futures agree on the magnitude of change in consumptive water use by 2041–2070. Base data from Warziniack et al. (2022). MGD = million gallons per day.

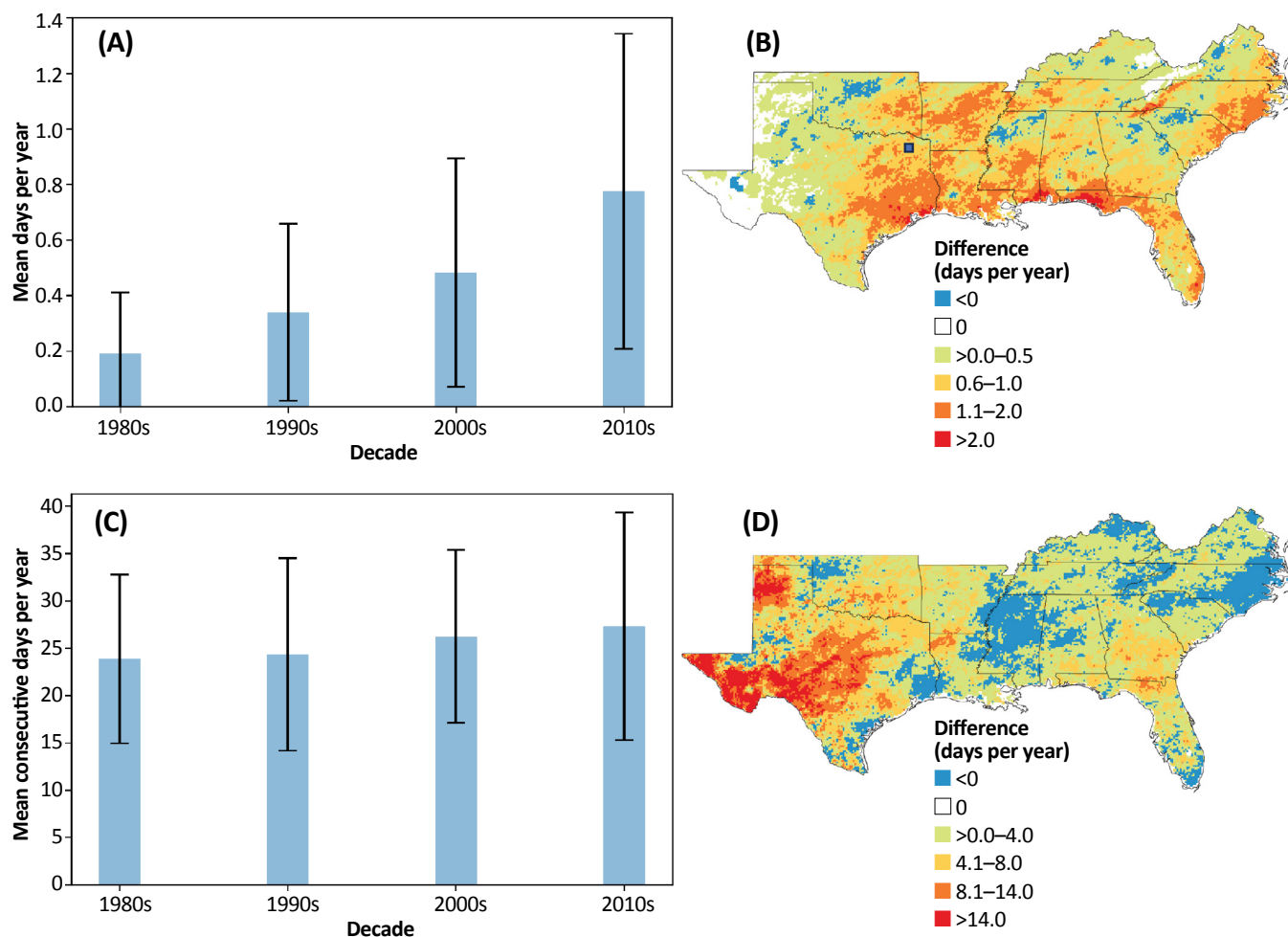


Figure 11—(A) Mean number of days per year with precipitation >76 mm by decade across the South from 1980 to 2018, (B) the difference in the number of days per year with precipitation >76 mm between 2010–2018 and 1980–1990, (C) mean number of consecutive days per year with <1 mm precipitation by decade across the South from 1980 to 2018, and (D) difference in the number of consecutive days per year with precipitation <1 mm between 2010–2018 and 1980–1990. These results are based on unpublished analyses (Bian 2024) using GRIDMET historical daily precipitation data ([www.climatologylab.org/gridmet.html](http://www.climatologylab.org/gridmet.html)). Error bars are one standard deviation from the mean.

across much of the South. Recent data suggest that the number of consecutive days without rain may have increased from 1980 to 2010, particularly in central and west Texas (fig. 11C, D).

Concurrent with dry climate extremes, compound dry-hot extremes have increased in frequency from 1896 to 2017 across portions of the South, including peninsular Florida, west Texas, southern Virginia, central North Carolina, and Mississippi (Alizadeh et al. 2020). For example, North Carolina's worst drought on record in 2007 led to water shortages causing restrictions in municipalities and water supply available for residents (Davis 2015) over much of the State. These conditions were driven not only by low precipitation but also high temperatures, with August 2007 being the second warmest and second driest on record at that time

(Davis 2015). The 2011 exceptional drought in Texas killed an estimated 300 million trees (Moore et al. 2016) and led to similar water shortages associated with severe reductions in water tables and stream flows. Summer temperatures were far above any previous summer of record, coinciding with record low precipitation (Hoerling et al. 2013). Though every region of Texas was impacted, the relative impact on forests provided insights into future resilience of forests to a warmer climate. The 2011 Texas drought disproportionately affected large mature trees and trees of certain species. Interestingly, even those genera known for their drought resilience (*Juniperus*, *Prosopis*) were affected (Moore et al. 2016). Managed pine stands fared better than unmanaged stands, which can be attributed to management practices that

avored lower stand density and smaller, younger trees (Klockow et al. 2020a). The long-term and cumulative effects of drought-related mortality on forest composition, water use, and productivity deserves further study (Klockow et al. 2018).

Future projections suggest that these trends in climatic extremes may continue through the 21<sup>st</sup> century, but the magnitude and direction of projected change can depend on the scenario-climate future. For example, Zobel et al. (2018) found that the frequency of days with no precipitation increased under two of three climate futures evaluated, and the frequency of days with precipitation greater than the historical 95<sup>th</sup> percentile increased under all three climate futures in the Southeastern United States by 2085–2094 compared to 1995–2005, all under RCP 8.5 (i.e., high emission scenario). Similarly, Gao et al. (2012) report that the mean annual number of days with extreme rainfall events (defined as the 95<sup>th</sup> percentile of all the rainy days) will increase from 5.2 days per year in 2012 to 6.7 days per year by the 2050s, and total extreme precipitation will increase by 35 percent under the high emission scenario. Hence, models suggest a future that has greater frequency of both no precipitation days as well as days with very high precipitation (fig. 12). Likewise, extreme high temperatures in both winter and summer are expected to increase in magnitude

and frequency over the 21<sup>st</sup> century, particularly under the high emission scenario (Vose et al. 2017). Though examples from the literature are based on different climate futures than those in this study, the overall projected direction of change in extreme events is likely similar.

Despite observed and projected increases in the frequency of wet extremes, corresponding broad-scale patterns of increasing trends in high streamflow events are difficult to detect and quantify. For example, Archfield et al. (2016) analyzed trends in flood metrics across 345 streamflow gauges in reference watersheds across the United States with flow measurements in at least 20 years between 1940 and 1969 and between 1970 and 2013, finding a “complex, fragmented pattern of flood change” that prevented generalizations of trends in the magnitude and frequency of high flow events across broad geographic regions. Others (Lins and Slack 1999, McCabe and Wolock 2014, Rice et al. 2015) analyzed trends in the magnitude of annual maximum daily flows but did not identify consistent, broad-scale increasing trends in high flow extremes across the South. Factors complicating the evaluation of historical trends in high flow extremes include the length of observation record as well as variable watershed properties, climate patterns, and human influences within aggregated regions. As suggested

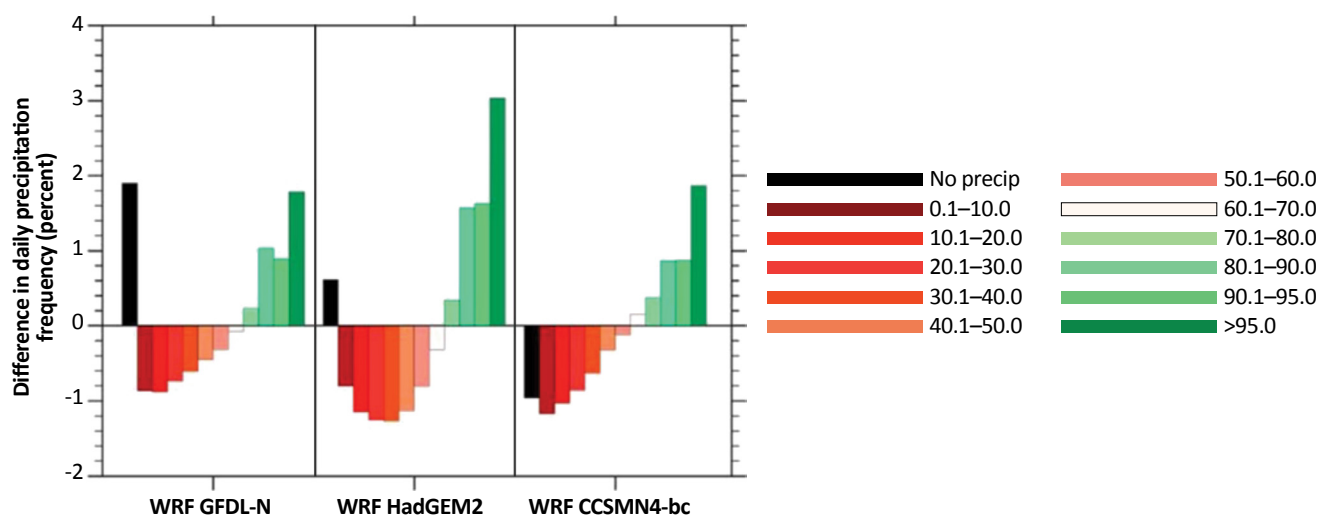


Figure 12—Difference in the daily precipitation frequency bins between historical (1995–2005) and high emission scenario (2085–2094) simulations in the Southeastern United States using regional climate models informed by Earth System Models (adapted from Zobel et al. 2018). The Southeastern region in these analyses included Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, and Virginia. A difference of +1.8 percent, for example in the case of the >95<sup>th</sup> percentile precipitation for the WRF GFDL-N model, indicates that the percentage of future days in that precipitation percentile range increased from the historical percentage of days in that percentile by a difference of 1.8 percentage points.

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by Archfield et al. (2016), the catchment scale may be the necessary resolution to quantify and attribute changes in high flow extremes over time. Despite these challenges, changes in high flow extremes have been detected in some catchments across the South and linked to changes in high precipitation extremes (table 4, fig. 13).

Unlike high flow extremes that are typically responses to short duration precipitation events, flow responses to low precipitation extremes manifest over relatively longer periods of time. Intuitively, the observed and projected increases in the duration of consecutive dry days and number of days without precipitation would result in increasing frequency of low flow extremes. These trends in dry climate extremes have not

resulted in a consistent pattern of observed low flow extremes across or within subregions of the South, but significant decreasing trends in low flow magnitudes are evident for some sites (table 4) (McCabe and Wolock 2014, Rice et al. 2015). Low precipitation effects on low flow extremes in a given watershed depend on the timing of low precipitation (e.g., growing versus dormant season), magnitude of precipitation deficit, and watershed characteristics (e.g., soil properties, vegetation type, and more) (Vose et al. 2016). In addition, anthropogenic flow alterations such as withdrawals and flow augmentation from reservoir releases and wastewater discharges vary by site and may compound or mask the effect of dry climate extremes on low flows at some sites.

**Table 4—Summary of studies on extreme flow events**

| Study                          | Flow extreme | Time period                                | Location(s)                                                                                     | Scenario                                                                  | Relevant findings                                                                                                                                                                                                                                                             |
|--------------------------------|--------------|--------------------------------------------|-------------------------------------------------------------------------------------------------|---------------------------------------------------------------------------|-------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|
| Dykstra and Dzwonkowski (2021) | High         | Historical: 1930–2019                      | Pascagoula, Tombigbee-Alabama, and Apalachicola watersheds in Mississippi, Alabama, and Florida | Historical observations                                                   | On average, the frequency of bankfull flow nearly doubled from once every 0.88 years in 1930–1959 to once every 0.46 years in 1990–2019. Flow for events larger than bankfull also tended to be larger in the more recent period, but these differences were not significant. |
| Slater et al. (2021)           | High         | Historical: 1970–2019                      | Global                                                                                          | Historical observations                                                   | Some evidence of increases in the magnitude and frequency of 20-, 50-, and 100-year high flow extremes since the 1970s for parts of the South.                                                                                                                                |
| Maurer et al. (2018)           | High         | Future: 2070–2099<br>Historical: 1971–2000 | Western U.S. including Texas, Oklahoma, and Arkansas                                            | Modeling using ensemble of GCMs and the high emission scenario            | The 100-year flow increased 43% on average and the future return period of the historical 100-year flow decreased to an average of 39.7 years, a 2.5-fold increase in recurrence probability.                                                                                 |
| Li et al. (2022)               | High         | Future: 2071–2100<br>Historical: 1976–2005 | Conterminous U.S.                                                                               | Modeling using ensemble of GCMs and the high emission scenario            | A larger fraction of catchments will have increasing peak flows, shorter lag times between precipitation events and peak flows, shorter event durations, and higher measures of flashiness across the South in the future.                                                    |
| Sun et al. (2015b)             | Low          | Historical: 1962–2012                      | 170 national forests in the conterminous U.S. (33 in the South)                                 | Modeling of historical conditions                                         | Areas of low water yield became larger in extent between 2002 and 2012 compared to 1962 through 2001 and the top five droughts for each national forest decreased water yield by 37% on average.                                                                              |
| Brunner et al. (2021)          | Low          | Historical: 1981–2018                      | Conterminous U.S.                                                                               | Historical observations                                                   | The spatial extent of low flows and the effects of high temperature on low flows have both increased over time.                                                                                                                                                               |
| Asadieh and Krakauer (2017)    | Low          | Future: 2070–2099<br>Historical: 1971–2000 | Global                                                                                          | Modeling using ensemble of GCMs under the high and 2.6 emission scenarios | Much of the South will be at higher risk of low flow extremes under RCP 8.5 scenarios, however there will be little change in risk under lower RCP scenarios.                                                                                                                 |

GCM = general circulation model; RCP = representative concentration pathway.

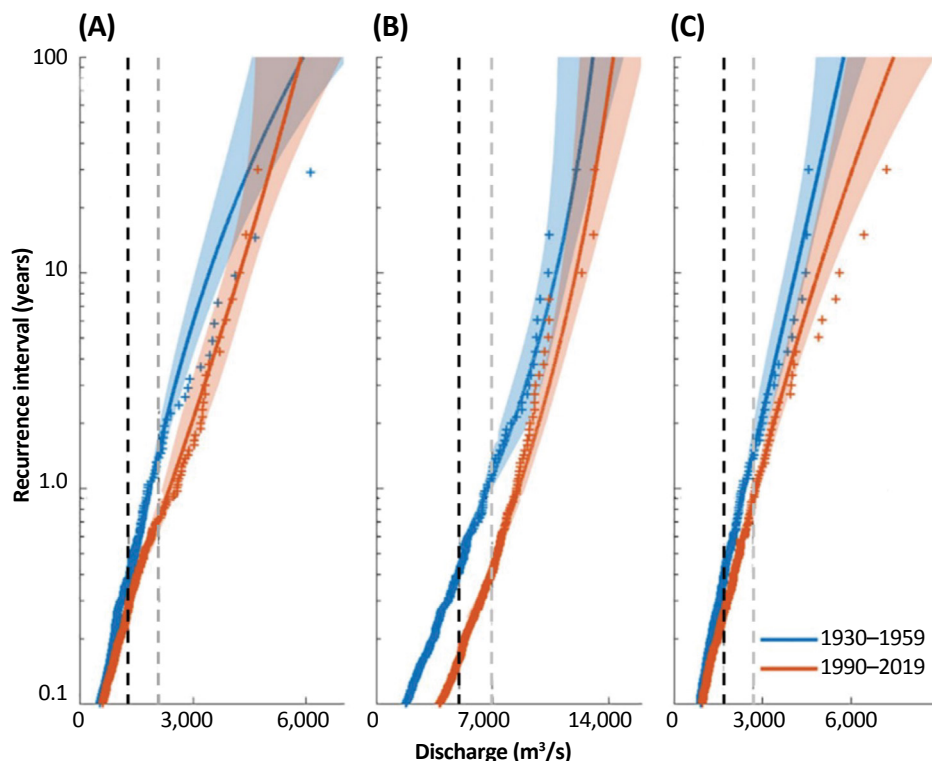


Figure 13—Estimated recurrence interval of flow events for sites on the (A) Pascagoula, (B) Tombigbee-Alabama, and (C) Apalachicola watersheds in Mississippi, Alabama, and Florida (adapted from Dykstra and Dzwonkowski 2021). The estimated distribution of discharge event recurrence interval (solid lines) were compared for two periods with a 95-percent confidence interval (shading), and a rank-order distribution of every event is shown with a + symbol. Bankfull discharge for a site within tidal influences and a site upstream of tidal influences for each watershed are represented by the black and grey dashed lines, respectively.

## Water quality changes due to climate and land-use change

Land use is a major control on water quality and changes in the proportion of land use or the proportion of flow originating on various land uses can be used to infer potential impacts on water quality in the future (Sun and Lockaby 2012). Among all land uses in the South, forests provide the best water quality (Caldwell et al. 2023), and in general, the larger the proportion of forested land upstream or water originating on forested land, the better the water quality and the less expensive it is to treat the water for municipal water supplies (Warziniack et al. 2017). In contrast, the larger the proportion of agricultural or developed land upstream, the lower the water quality. The projected changes in land use across the South, therefore, could have consequences for water quality in areas where change from forested to developed land is greatest.

Conversion of less intensive land uses (i.e., forests, grasslands, and wetlands) to developed land can result in water quality degradation due in part to a reduction in pollutant uptake or trapping efficiency (Neary et al. 2009), but also due to the addition of pollutant sources in a developing watershed (Carey et al. 2013, O'Driscoll et al. 2010). For example, lawn fertilizers can increase nutrient loads to receiving waters, as can point sources associated with human population and industry, such as wastewater treatment plants. Reductions in soil infiltration capacity and exposed soil associated with development and construction activities can increase surface runoff and soil erosion, leading to increased sediment loading in streams (O'Driscoll et al. 2010). This reduction in soil infiltration capacity, in addition to the introduction of impervious surfaces (e.g., roads, parking lots, and rooftops), can result in increases in flow volumes and stream flow velocities during storms, resulting in stream channel erosion and sedimentation. Impervious surfaces also become sources of many pollutants as they are collected, stored, and flushed to surface

waters during storms (Arnold and Gibbons 1996, Brabec et al. 2002). Some of these pollutants, including heavy metals, pesticides, and organic compounds such as polychlorinated biphenyls and polycyclic aromatic hydrocarbons, can be toxic to aquatic ecosystems and humans. Aside from pollutants that are directly related to land-use change, pollutants such as antibiotics, analgesics, narcotics, psychotherapeutics, pathogenic microbial populations, and other emerging contaminants may also increase with increasing human population and activities (Deo 2014, Wilkinson et al. 2017). Collectively, these pollutants can induce symptoms of “urban stream syndrome”, which results in ecological degradation of streams draining from developed land (Walsh et al. 2005).

Climate change could magnify these land-use change driven effects on water quality through increases in the frequency and magnitude of extreme precipitation events (Gao et al. 2012, Zobel et al. 2018), because a majority of nonpoint source pollutant loads from developed and agricultural land uses are delivered to surface water supplies during precipitation events (EPA 2003). For example, Johnson et al. (2015) projected changes in streamflow, total nitrogen, total phosphorus, and suspended sediment loading by mid-21<sup>st</sup> century in 20 river basins across the United States including the Neuse, Suwanee, Apalachicola, Amite, and Trinity in the South, finding that the effects of future urbanization on water quality is small compared to the effects of climate change overall, but the urbanization effects could be greater in subbasins where development is concentrated. These water quality impacts driven by climate and land-use change have consequences for both human populations and aquatic communities that depend on water resources in the South.

Areas where the percentage of upstream forested land is projected to decrease may be vulnerable to water quality degradation in the future (Caldwell et al. 2023). Similarly, areas where the percentage of upstream impervious surface is projected to increase may also be most vulnerable to water quality degradation (Brabec et al. 2002, Kim et al. 2016, Schueler 1994, Schueler et al. 2009). We adopted the impervious cover watershed stress classifications used in the Environmental Protection Agency (EPA) Integrated Climate and Land Use projections (EPA 2009) to infer potential future quality and aquatic ecosystem degradation

resulting from development across the South. In this classification, the percentage of upstream area in impervious cover is 0–0.9 percent (unstressed), 1.0–4.9 percent (lightly stressed), 5.0–9.9 percent (stressed), 10.0–24.9 percent (impacted), and >25.0 percent (degraded). We then mapped HUC12 watersheds where the total upstream impervious cover comprised >5 percent of the upstream area (i.e., stressed or greater) under future scenarios.

More than 50 percent of HUC12 watersheds across the South had ≥50 percent of their upstream area in forested land in 2020 (fig. 14A). Of those HUC12 watersheds with ≥50 percent upstream forested land, 24 percent were projected to have a >5 percent forest loss by 2070 under the HM-middle scenario-climate future (fig. 14B). These forest losses were projected to occur primarily in east Texas, western Louisiana, and southwestern Arkansas, and across portions of the piedmont and Coastal Plain from Virginia to Mississippi. Upstream forested land was projected to decrease for all scenarios between 2020 and 2070 across forested watersheds in the South, with the percentage of forested HUC12 watersheds projected to have >5 percent upstream forest loss by 2070, ranging from 18 to 23 percent under the HL scenario to 21 to 27 percent under the HH scenario (fig. 14C).

Across the South in 2020, 7.8 percent of HUC12 watersheds had >5 percent impervious cover in their upstream area (fig. 15A). These areas tended to contain major urban centers as well as downstream watersheds—some a long distance downstream. In 2070, 14.9 percent of HUC12s were projected to exceed 5 percent upstream impervious cover under the HM-middle scenario-climate future (fig. 15B). In some cases, future impervious cover extended the length of major rivers exceeding the 5 percent upstream impervious cover threshold much further downstream. For example, the Chattahoochee River had >5 percent impervious cover upstream in 2020 from Johns Creek, GA north of Atlanta to just downstream of Georgetown, GA (>400 km of river), but this threshold was projected to extend another 119 km further downstream by 2070 under the HM-middle scenario-climate future. Upstream impervious cover was projected to increase for all scenarios between 2020 and 2070, with the percentage of HUC12s with >5 percent upstream impervious cover more than doubling by 2070 under the HH scenario (fig. 15C).

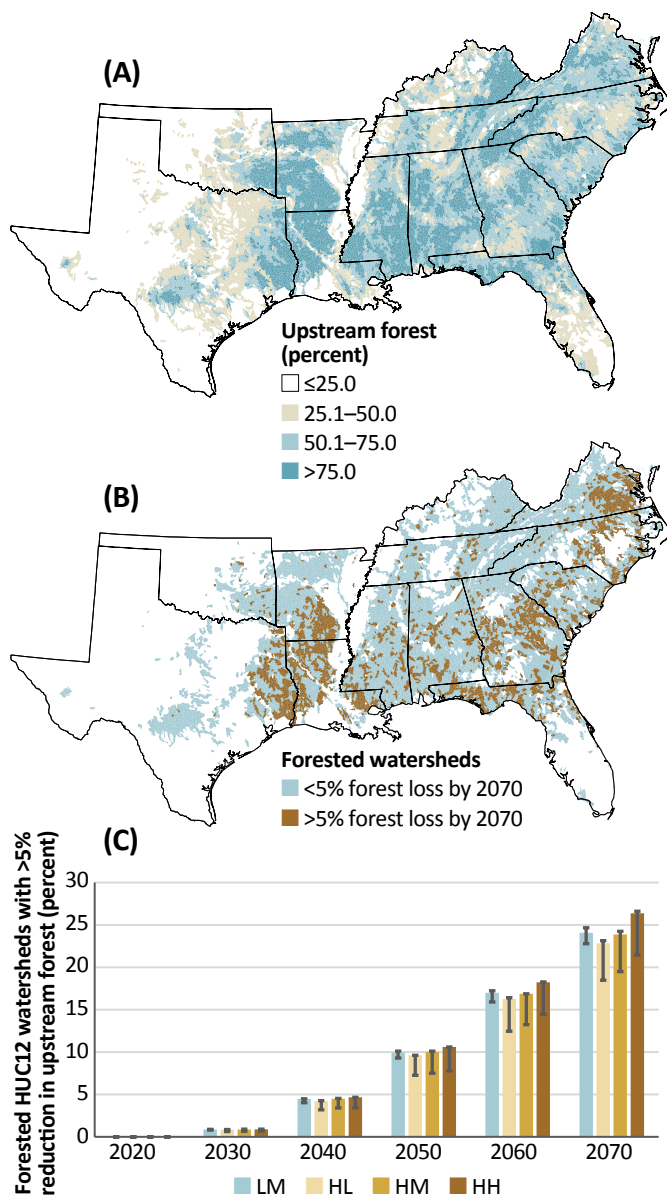


Figure 14—(A) Percentage of forest land upstream of watersheds in 2020, (B) forested watersheds (i.e., >50 percent upstream forest land in 2020) that were projected to have >5 and <5 percent forest loss by 2070 for the HM-middle scenario-climate future (tables 1 and 2), and (C) percentage of forested watersheds that were projected to have >5 percent upstream forest loss by decade and scenario. Bars in (C) are the percentage of watersheds for HM-middle; error bars are the range across climate futures for each scenario.

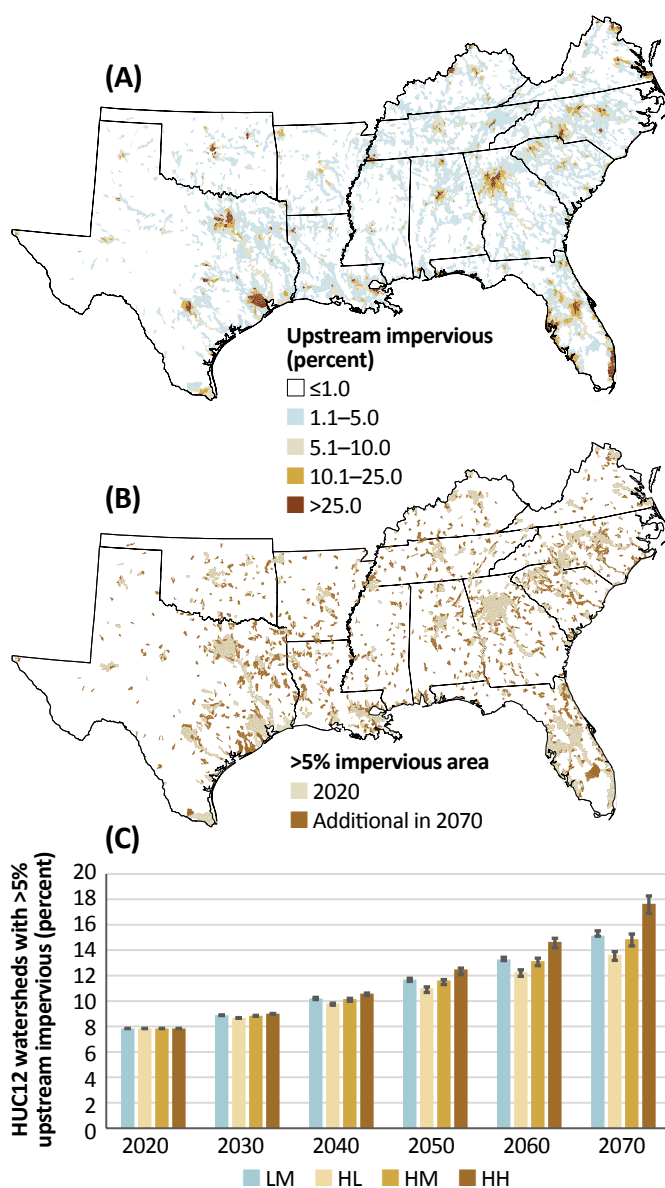


Figure 15—(A) Percentage of impervious cover upstream of watersheds in 2020, (B) watersheds that had <5 percent upstream impervious cover in 2020 but were projected to have >5 percent upstream impervious cover in 2070 under the HM-middle scenario-climate future (tables 1 and 2), and (C) percentage of all watersheds in the South that had >5 percent upstream impervious cover by decade and scenario. Bars in (C) are the percentage of watersheds for the middle climate future; error bars are the range across climate futures for each scenario.



## Potential effects of projected changes on public water systems

The projected changes in future annual flow, minimum monthly flow, and upstream land use could have consequences for downstream public water systems. Across the South, there were 1,361 public water systems with a total of 1,905 surface water intakes that directly provided drinking water to more than 46 million people in 2017. Projections of reduced annual water supply and decreases in minimum flows in the future (figs. 7, 8) could stress these water systems and limit their capacity to support population growth. Further, less upstream forested land and more upstream impervious cover (figs. 14, 15) could have implications for raw water quality and treatment costs. We linked public water system intakes to projections of flow and land use, and identified those that may be most vulnerable to climate and land-use change in the future for the HM-middle scenario-climate future (fig. 16). Those identified were projected to have a decrease in annual flow of >10 percent (fig. 14A), a decrease

in annual minimum monthly flow of >10 percent (fig. 16B), a decrease in the percentage of upstream forested lands of more than 5 percentage points for intakes with >50 percent upstream forest land use in 2020 (fig. 16C), and upstream impervious cover exceeding 5 percent for intakes with <5 percent upstream impervious cover in 2020 (fig. 16D). Many of these at-risk facilities were located in Texas, Oklahoma, and Arkansas from a water quantity perspective, and many of these were projected to also experience increases in upstream impervious cover with implications for water quality. There were also 181 public water system intakes scattered across the region that were projected to have declines in the percentage of upstream forested lands of >5 percent and 143 that were projected to have impervious cover exceed 5 percent of the upstream area under this scenario-climate future. For example, the city of Jacksonville, TX public water system serves 14,500 people, and maintains a surface water intake on Lake Jacksonville. This intake was projected to have decreases in annual

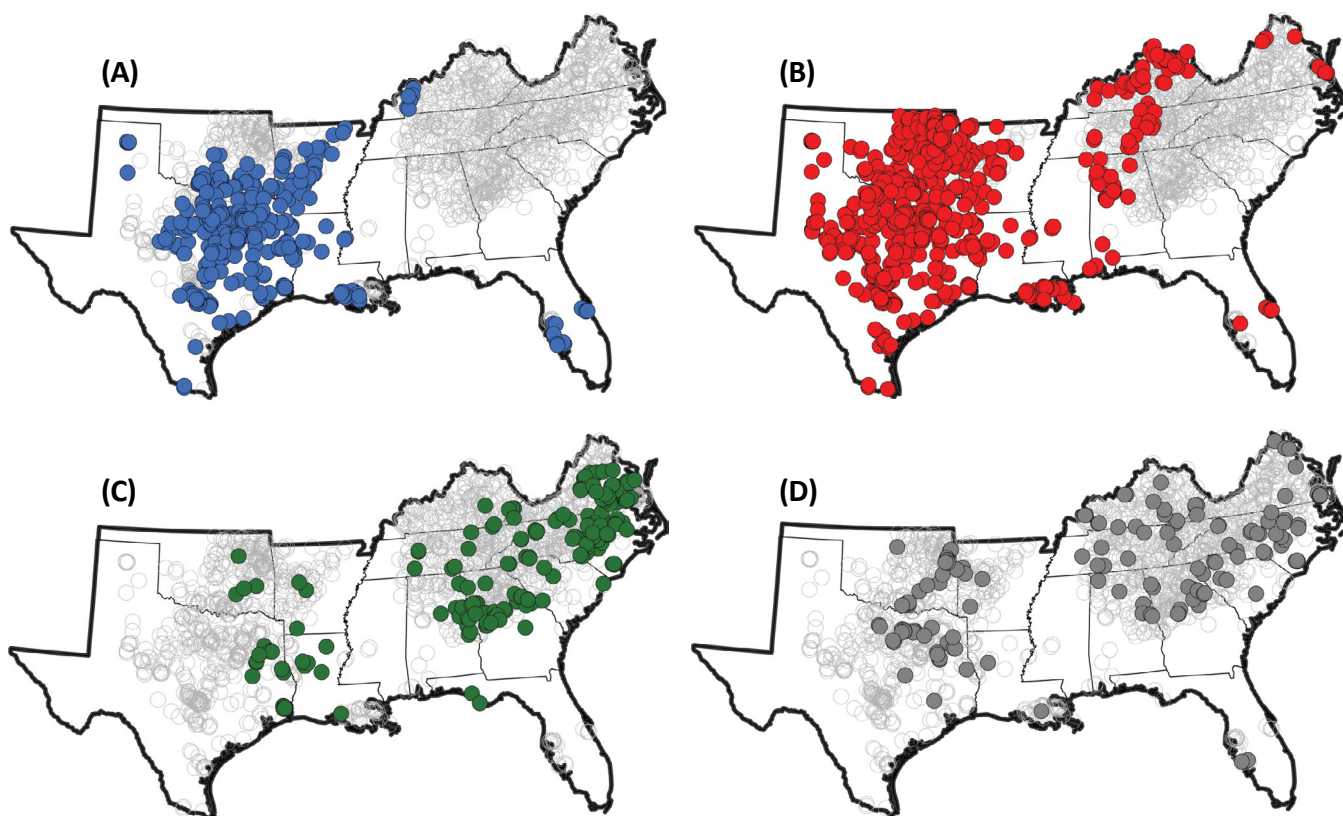


Figure 16—Drinking water intakes where (A) the mean annual surface water supply was projected to decrease >10 percent, (B) the annual minimum monthly flow was predicted to decrease >10 percent, (C) the percentage of upstream forest land use was >50 percent in 2020 but was predicted to decrease by >5 percent, and (D) the percentage of upstream impervious surface was <5 percent in 2020 but predicted to exceed 5 percent by 2041–2070 relative to 1981–2010 for the HM-middle scenario-climate future (tables 1 and 2).

flow and annual minimum monthly flow by 2041–2070 of 3.8 and 24 percent, respectively, a decrease in upstream forested lands from 68 percent in 2020 to 43 percent in 2070, and an increase in upstream impervious surfaces from 3 percent in 2020 to 15 percent in 2070 under the HM-middle scenario-climate future.

### Groundwater recharge and depletion

Changes in surface water and water use discussed above could affect groundwater recharge, storage, and water supply that relies on groundwater systems. Groundwater recharge could decrease in areas where water yield is projected to decrease (figs. 7, 8), both locally and in upgradient recharge areas. Concurrent with decreases in recharge, groundwater pumping could increase in these and other areas due to increased irrigation, industrial, and domestic water demand under a warmer, and sometimes drier climate (Brown et al. 2019). Indeed, groundwater pumping already approaches or exceeds recharge rates for some aquifers across the South, including west and coastal Texas, central Florida, and the Mississippi

valley in Arkansas (Reitz et al. 2017). Increased groundwater pumping could decrease ground water supplies and reduce surface streamflow beyond levels resulting from climate change alone, where groundwater and surface water are tightly coupled (Qi et al. 2020). Where groundwater becomes depleted or other pumping-related effects occur (e.g., land subsidence, saltwater intrusion), water users may shift to surface water supplies. For example, Houston, TX shifted from primarily groundwater sources to surface water in the 20<sup>th</sup> century due to widespread land subsidence associated with groundwater pumping (Menard 2016). Alternatively, some water users may shift from surface to groundwater when surface water sources become unreliable during drought or too expensive to treat. For example, water users in the California Central Valley supplemented or replaced surface water with groundwater sources during drought in the early 21<sup>st</sup> century (Famiglietti et al. 2011). However, alternative surface or groundwater sources can be constrained in some places limiting opportunities for conjunctive use (i.e., the coordinated use of surface and groundwater).

## (2) How Might These Projected Changes in Climate and Water Resources Interact With Other Disturbances Such as Wildfire, Drought, Sea-Level Rise, Hurricanes, Invasive Plants, and Insect and Disease Outbreaks?

Changes in hydrologic conditions due to climate and land-use change will likely have complex feedbacks and interactions with other disturbances such as wildfire, insect and disease outbreaks, and hurricanes (fig. 1). These disturbances are common in southern forests and are drivers of ecosystem dynamics; however, climate change can alter natural disturbance regimes by increasing (or decreasing) severity, frequency, and size of disturbances (Vose et al. 2018). Furthermore, multiple co-occurring disturbances over short time scales can have compounding effects on forests and thus the water resources provided by forests (Goeking and Tarboton 2020). In general, because evapotranspiration is a large component of the forest hydrologic cycle in the South (Sun et al. 2011a), disturbances that alter evapotranspiration through changes in species composition or the

structure of forests will have the greatest effects on streamflow amount and timing in forested watersheds. And, because forest ecosystems sequester nutrients, minimize erosion, and forest cover shades water, disturbances that change species composition, structure, or disrupt nutrient cycling processes and alter infiltration rates will have the greatest effects on water quality. In this section, we discuss potential interactions among disturbances, changing climate and land use, and water resources, and examine their implications.

### Wildfire

Across most of the South, forests are adapted to—or in some cases, require—low severity, frequent fire (Guyette et al. 2012, Noss et al. 2015, Stambaugh et al. 2015), for maintaining open forest understory and midstory conditions, and promoting a range of

ecosystem services. To maintain these conditions, forest managers and private landowners across the region implement prescribed fires, which tend to be relatively low severity and small in areal extent. However, larger, higher severity wildfires also occur in the region, especially in association with dry periods and drought conditions (Barbero et al. 2015, Caldwell et al. 2020a, James et al. 2020, Nielsen-Gammon 2012, Schulte et al. 2019), and several notable large wildfires have occurred recently, including in the southern Appalachian Mountains, Texas, Florida, and in areas with peat soils in the Coastal Plain.

Warmer and drier conditions in some areas of the South are likely to increase fire risk (Liu et al. 2009) and lightning-caused wildfires may increase across many parts of the South (Prestemon et al. 2016, Romps et al. 2014), including in the southern Appalachian Mountains (James et al. 2020) and elsewhere. At the same time, the ability for managers to conduct prescribed fires may become more constrained due to climate change (Kupfer et al. 2020) as well as a growing human population, especially at the wildland-urban interface where there is greater risk to property and people including exposure to fine particulates in smoke (Mitchell et al. 2014). Projections under the 20 RPA scenario-climate futures indicate increases in the annual volume of trees killed by wildfire in the South between 2020 and 2050, with larger increases expected under the higher warming scenarios (RCP 8.5: HL, HM, HH) (Costanza et al. 2023a). Increases in the areas of both moderate and high severity fires were also projected for the South over the same time period (Costanza et al. 2023a).

An increase in wildfire occurrence in the South, especially if those wildfires burn with medium to high severity, could affect streamflow, at least in the short term. In a review of streamflow responses to wildfires across the United States, Paul et al. (2022) reported a median duration of postfire streamflow increase of 3.5 years, with wide variation among studies. Large, severe wildfires have historically been relatively rare in the South and thus observations of fire impacts on hydrology are scarce (Hallema et al. 2018). However, the few studies examining hydrologic responses to severe wildfires in the South suggest that water yield may increase immediately following wildfire. For example, water quantity substantially increased in the 2 years following the southern Appalachian

wildfires in 2016, and the amount of increase was related to the extent of the watershed that was burned with high severity (Caldwell et al. 2020a). Similarly, following the large Bastrop wildfire in Texas in 2011, burned areas had higher soil moisture (Cardenas and Kanarek 2014), and evapotranspiration was lowest in forest stands that burned with highest severity (Cooper et al. 2019), also suggesting the potential for increased water yield from those stands. Alternatively, any changes in the frequency of low severity wildfires or prescribed burns may have little direct effect on water quantity at the scale of large watersheds. For example, infrequent low severity (either wildfire or prescribed) burns typically have minimal direct effects (i.e., low magnitude and short duration) on water quantity because they do not alter soil hydraulic properties or cause substantial tree mortality (Moody et al. 2016). In addition, most of these fires are relatively small relative to watershed area, and therefore do little to change flow across an entire watershed (Hallema et al. 2018, 2019). Although localized ash and sediment movement may occur and soil solution chemistry may be altered in the short term (Knoepp and Swank 1993, Knoepp et al. 2005), prescribed fires typically have little impact on water quality (Elliott and Vose 2005, Hahn et al. 2019, Uzun et al. 2020). Water quality can be affected by wildfires (e.g., Caldwell et al. 2020a), but the magnitude and duration of impact depends on numerous factors such as fire severity, fire area, postfire weather conditions, topography, riparian condition, and proximity to surface water (Thompson et al. 2019). In addition to the direct effects of fires, both high severity wildfires and frequent prescribed fires in the South can affect water yield indirectly by abruptly or gradually changing forest structure, stand age, or species composition. Frequent prescribed fire can lower basal area and reduce density of midstory vegetation, which can lead to decreased evapotranspiration, and thus increased water yield (McLaughlin et al. 2013). If regional restoration initiatives are successful in using frequent fire within increasing constraints to restore more open conditions in pine forests across the South, regional water flows could increase in the long term (e.g., Qi et al. 2022, Younger et al. 2023). For example, forests shifted from pine- to oak-dominated communities after recent wildfires in Texas, especially where fires burned with



high severity, and these species shifts have led to decreases in evapotranspiration (Cooper et al. 2018, 2019). If more frequent wildfires in Texas and other portions of the South such as the southern Appalachian Mountains are severe enough to cause tree mortality, they could support a widespread transition to xerophytic tree species such as oaks and hickories, from the mesophytic species like maples that have begun to dominate forests without fire (Vose and Elliott 2016). Because those xerophytic species have lower growing season evapotranspiration rates than mesophytic species (Ford et al. 2011a), such a transition could result in increased water yield. On sites with organic-rich peat wetland soils, high-severity fires spread vertically downward into the soil profile and can smolder for days to months in the ground, leading to changes in species composition or even wholesale ecosystem shifts from forested wetlands to marsh, as has been demonstrated in Florida (Casey and Ewel 2006).

## Drought

Drought can be defined as a period of moisture deficit resulting from below average precipitation, high temperatures, or both (Clark et al. 2015). Lack of moisture or increased temperatures or both can have direct effects on streamflow in the short term (days to months) (Vose et al. 2016). Drought can reduce forest productivity over longer time scales, causing shifts in forest species composition, and affecting the ability of forests to regenerate, all of which can affect the capacity of forests to provide water-related ecosystem services (Vose et al. 2016). Drought can also lead to large, high severity wildfires, which can affect water quantity and quality.

Future projections of meteorological drought, defined as a deficit in precipitation relative to a long-term mean, indicate an increase in the frequency at which forests in the South are likely to be exposed to drought conditions, and in the severity of those conditions, especially by mid-century (2041–2070) (Costanza et al. 2023a). Costanza et al. (2023b) examined changes in forest exposure to meteorological drought under the 10 climate futures included here (5 GCMs and 2 RCPs) and found that all 10 futures agreed that exposure could be especially pronounced in portions of west Texas. Other areas in the western and northern portions of the South, specifically Oklahoma,

Louisiana, and portions of Mississippi, Kentucky, and Tennessee show high exposure to drought under at least one climate future.

In general, more drought in these places implies reduced streamflow and possibly elevated nutrients and warmer water; however, droughts accompanied by warmer conditions could lead to greater tree mortality, which may offset some of the direct effects of reduced precipitation on streamflow (Allen et al. 2015). Generalizing the effect of drought on forests and water quantity and quality is difficult because it depends on a number of interacting local factors, including forest stand characteristics, physiographic region, and the specific climate conditions that cause drought (Crausbay et al. 2017, 2020; Luce et al. 2016). In other words, a meteorological drought may not necessarily lead to an ecological drought (a reduction in the amount of available water for ecological processes) or a hydrological drought (a substantial reduction in streamflow). Thus, to determine the effects of drought on forests, hydrologic conditions, and water quantity and quality, projections of exposure to meteorological drought conditions must be coupled with knowledge about the differential sensitivities and adaptive capacities of tree species and forest ecosystems at local scales.

Evidence from droughts in the Eastern United States show that some tree species are more sensitive to drought than others (Novick et al. 2022). For example, Elliott et al. (2015) analyzed the drought versus growth response of six prominent Appalachian hardwood species and observed that individual tree growth relationships varied according to xylem anatomy, with diffuse porous mesophytic species (e.g., maples) being generally more susceptible to reduced growth during drought than ring porous xerophytic species (e.g., oaks). Similar patterns were observed by Brzostek et al. (2014) using a broad scale examination of FIA data. Species-level variation in drought-related mortality is more complex and difficult to quantify (Clark et al. 2015), but differences in growth sensitivity to drought suggest a transition to more xerophytic species over the long term. Thus, similar to the fire effects discussed above, more frequent drought in the future could mediate a shift toward more xerophytic species, and higher resulting streamflow due to lower water use by those species. The 2011 Texas drought affected all regions and



most tree taxa, killing 6.2 percent of trees across the State by the end of the year (Moore et al. 2016). Larger trees were especially prone to die (Klockow et al. 2020b, Moore et al. 2016). Some mesic species died in greater proportion in the drier western extent of their range, but mortality was also higher than expected in species such as honey mesquite (*Prosopis glandulosa*), which is classified as drought tolerant (Moore et al. 2016). Pines, on the other hand, were relatively resilient to drought-caused mortality, especially in managed stands (Klockow et al. 2020a); however, loblolly pine mortality did not increase significantly until 2013, 2 years following the worst of the drought conditions (Klockow et al. 2018). The delayed mortality observed in Texas and elsewhere suggests that drought effects on streamflow may be prolonged.

In the Coastal Plain, management efforts emphasize restoration and maintenance of open pine forests and woodlands, including longleaf pine (*Pinus palustris*). Longleaf pines seem to be relatively drought tolerant because they use water more conservatively compared with other pine species such as loblolly pine that are not as efficient (Samuelson et al. 2012, 2019). Thus, in places where loblolly pine stands have been converted to longleaf, there may be only small effects of drought on forests, and thus on forest-driven effects on water quantity and quality.

Increased drought in the future can also lead to more risk of other disturbances such as wildfire, and insect and disease outbreaks, which can alter forest vegetation and thus alter hydrologic processes. Indeed, at least two recent large wildfires in the South have occurred following drought or long dry spells (Caldwell et al. 2020a; Nielsen-Gammon 2012; Williams et al. 2014, 2017), both of which altered water supply, especially where fires burned with high severity.

The effects of drought on insects and diseases in eastern forests are highly complex and evidence for strong and generalizable connections are weak (Kolb et al. 2016). For example, southern pine beetle (*Dendroctonus frontalis*) outbreaks can have devastating impacts on loblolly pine (and other pine species), but evidence of a strong linkage between drought and widespread southern pine beetle outbreaks is generally lacking (McNulty et al. 1998). This may be partially explained by the nonlinear relationship between resin

flow and drought (Kolb et al. 2016): moderate drought increases resin flow (making trees less susceptible to beetle colonization) and severe drought decreases resin flow (making trees more susceptible to beetle colonization) (Lombardero et al. 2000). Furthermore, incidence of forest diseases that thrive under wetter conditions may decrease during drought, whereas incidence of diseases that infect stressed trees may increase during drought (Kolb et al. 2016). In contrast, correlations between insect outbreaks and warmer temperatures are stronger, where warming may increase both the range and activity of certain harmful forest insects (Olatinwo et al. 2013, Weed et al. 2013).

## Sea-level rise

In coastal regions of the South, the effects of climate and land-use change on water resources will be further exacerbated by sea-level rise. Globally on average, sea level is projected to rise by between 0.4 and 2.5 m by 2100 (Oppenheimer et al. 2019). However, the rate of sea-level rise varies locally. For example, the coasts of Virginia and North Carolina north of Cape Hatteras are part of a hotspot of sea-level rise in which sea level is rising three times faster than the global average (Sallenger et al. 2012). Since 2010, the rate of sea-level rise along the coast from south of Cape Hatteras and extending into the western Gulf of Mexico has accelerated to more than 10 mm per year, a rate that is unprecedented compared to the last 120 years (Dangendorf et al. 2023).

As a result of saltwater intrusion from sea-level rise forests are already being converted to marsh, mangrove, or other intertidal vegetation along the Atlantic and Gulf of Mexico coasts of the South. This replacement can be observed in the appearance of “ghost forests”, which are dead standing trees and stumps surrounded by wetland vegetation, signifying an area where a forest once stood (Kirwan and Gedan 2019). Both ghost forests and changes in forest species composition have been reported along the Atlantic Coast where sea level is rising faster than average, including in North Carolina and Virginia (Kirwan and Gedan 2019; Ury et al. 2019, 2021). Conversion of forests to wetland vegetation or ghost forests is also closely associated with other disturbances such as fire, which can help maintain forests (Poulter et al.

2009), and hurricanes, which cause tree mortality and have been associated with the appearance of ghost forests (Ury et al. 2021).

Marshes, mangroves, and other intertidal vegetation provide benefits for water quality, and research has shown that intertidal vegetation likely can resist further sea-level rise better than upland vegetation. Therefore, where wetlands are able to migrate and establish inland, they can provide necessary protection from inundation to nearby upland vegetation and human land uses. In a simulation study, when marshes were allowed to migrate up to 400 m into nearby agricultural fields, the benefits to crop yields due to prevention of saltwater inundation under large storm surges outweighed the costs of lost agricultural land (Guimond and Michael 2021). However, urban development near low lying vegetation along the coast can present barriers to wetland migration in what is known as “coastal squeeze,” preventing inland migration of wetlands. Tampa Bay (FL), Galveston Bay (TX), and Atchafalaya/Vermillion Bays (LA) are among those with greatest risk of coastal squeeze in the future (Borchert et al. 2018). Though marsh migration can help prevent further inundation and erosion, saltwater inundation and conversion to wetlands can have negative effects on water quality. Simulation studies in Virginia and South Carolina suggest increases in salinization that could place drinking water sources at risk (Paul et al. 2019). Also, as upland vegetation or agricultural lands convert to wetlands, relatively quick releases of nutrients that have accumulated in upland soils can occur (Tully et al. 2019).

These negative effects of sea-level rise on water resources may be exacerbated in areas where human population and land use are already concentrated or projected to increase in the future. Flood risk will increase where vegetation transitions to marshes or mangroves cannot occur because of coastal structures, flood-control structures, or urban/residential land uses. This will be especially true in coastal urban areas where impervious cover is high, stormwater infrastructure is inadequate, and human populations are growing. Tidal floods, otherwise known as nuisance floods, in coastal cities are already becoming more common in some cities such as Charleston, SC and Savannah, GA (Carter et al. 2018), and are projected to become more common as sea level rises. Forests within urban

areas can help reduce flooding via increased evapotranspiration compared to other land uses and impervious surfaces (Wan et al. 2021).

## Hurricanes

Climate change is likely to lead to an increase in hurricane intensity, and some global climate models suggest an increase in frequency of the most intense hurricanes (Hayhoe et al. 2018). More intense hurricanes not only bring higher winds but can also be accompanied by extreme precipitation, and, along with sea-level rise, cause more intense storm surge and flooding. When hurricanes occur, and especially if they are accompanied by extreme precipitation or near urban and other human land uses, tree mortality and defoliation results in decreased evapotranspiration and increased runoff, streamflow, and sediment loads (Miller et al. 2019, Worley et al. 2022). Decreased evapotranspiration can also alter regional precipitation patterns downwind because less water is released into the atmosphere (Miller et al. 2019). Lower evapotranspiration and increased streamflow in forested watersheds can be detected from months and years to decades following a hurricane (Amatya et al. 2021a, Jayakaran et al. 2014, Worley et al. 2022). The degree of tree mortality from a hurricane and thus the resulting effects on streamflow are determined in part by tree species and age composition. Hurricanes tend to affect larger, mature trees more than seedlings and saplings, and large-diameter pines may be most susceptible (Jayakaran et al. 2014, Kupfer et al. 2008). There are also major differences among tree species in terms of susceptibility to hurricanes with species such as baldcypress (*Taxodium distichum*) and live oak (*Quercus virginiana*) showing strong resilience compared to pines and some other species (Joffrion 2007). Evidence from a paired watershed retrospective study of the effects of Hurricane Hugo suggests that due to this differential mortality, immediately following a hurricane, streamflow increases more in forests that had more older than younger trees prior to the hurricane. Over years to decades following a hurricane, sites with a mix of age classes prior to the hurricane recover to prehurricane streamflow rates quicker, as younger trees replace the older trees that were killed (Jayakaran et al. 2014). Though it is difficult to know the distributions of tree ages within each forest stand in the South,

forests in the Eastern United States are becoming older on average (Coulston et al. 2023, Oswalt et al. 2019), which suggests increased susceptibility in southern forests to tree mortality from hurricanes. In addition, the occurrence of other disturbances such as drought either preceding or following a hurricane can cause additional stress or mortality and lead to standing dead trees, or can delay forest recovery following a hurricane (Edgar et al. 2019, Gang et al. 2020), increasing or prolonging effects on water resources. An increase in the occurrence of drought could make these compounded effects more common in the future.

Evidence from Hurricane Harvey in 2017 in and around urban areas in Texas showed that the immediate effects on streamflow and other hydrologic characteristics were extreme, causing large amounts of runoff, high sediment loads, and disrupted salinity, but many of these measures returned to prehurricane levels within days (Patrick et al. 2020). A large amount of rainfall occurred during and preceding the storm (Yao et al. 2020). As a result, soils became saturated so quickly that peak flows from urban areas were only slightly greater than peak flows from areas with other land uses, thus urbanization had a minimal effect on flows in that case (Li, X. et al. 2020).

Taken together, existing research suggests that the signal of forest recovery from hurricanes can be found in streamflow and other hydrologic characteristics for years following a hurricane and may be affected by forest characteristics as well as other disturbances. However, the most acute effects of hurricanes on water quantity and quality may be short lived and vary less with land use. If more hurricanes are accompanied by extreme rainfall in the future, land-use type and forest conditions may become less important for determining hurricane-related impacts, because all areas may experience the same extreme impacts.

## Nonnative invasive plants

Nonnative invasive plant species have long-term effects on forest ecosystems, including changes in species composition, alterations to forest succession, and concomitant effects on water quality and quantity (Liebhold et al. 2017, Mack et al. 2000, Martin et al. 2009, Poland et al. 2021). Damage caused by these species, and efforts to control them, are costly (Pimentel et al. 2005). Nonnative invasive plants are relatively common

in southern forests, and forests with the highest rates of invasion occur in the Piedmont, Mississippi Alluvial Valley, and along portions of the western areas of the region (Costanza et al. 2023a, Oswalt et al. 2021). Nonnative invasive plant distribution is linked to human land uses, roads, and forest fragmentation (Riitters et al. 2017).

Aside from direct effects on forests, invasive grasses contribute to forest ecosystem change via the grass-fire cycle, in which invasive species increase fire frequency and intensity, and as a result, suppress regeneration of native species and promote survival and persistence of invaders (Fusco et al. 2019, Kerns et al. 2020). A grass-fire cycle in the South has been documented due to invasion of Coastal Plain pine forests by cogongrass (*Iperata cylindrica*), and invasion of deciduous forests by Japanese stiltgrass (*Microstegium vimineum*) (Flory et al. 2015, NeSmith et al. 2018). These invasions can result in altered forest tree species composition or even forest loss, with potential to alter stream runoff, soil chemistry, and thus resulting water quality and quantity—though more empirical research is needed to fully quantify these potential effects.

Determining the likelihood of future invasion by one or more invasive species is complex. Generally, invasive species are likely to expand their ranges as developed, crop, and pasture land uses expand, and thus become more prevalent in forests across the South. Invasive species generally disperse well and have high growth rates; thus, they can cope well with rapidly changing conditions under climate change. As a result, changing climate is likely to result in higher rates of forest invasion (Finch et al. 2021). In addition, increasing frequencies of large disturbances such as hurricanes will stimulate encroachment of some invasives such as Chinese tallow (*Triadica sebifera*), which can quickly dominate coastal forests once overstories are opened by storm disturbance (Henkel et al. 2016). The observed effects of invasive plants on water quality and quantity are thus likely to become exacerbated with global change. Though evidence for increased water use and resulting decreases in streamflow due to the direct effects of invasive plants is accumulating, there is limited research on water quality impacts in southern forests (Miniat et al. 2021). However, there are examples of indirect effects such as increased stream nitrogen following tree mortality from invasive insects. For

example, in the central and southern Appalachians, large-scale mortality of eastern hemlock from the invasive hemlock woolly adelgid (*Adelges tsugae*) resulted in increased groundwater (Riscassi and Scanlon 2009) and streamwater nitrate (Miniat et al. 2021). More research on the specific water quality and quantity effects of invasive plants in the South is needed to determine the precise effects on water resources in the future.

## Insects and diseases

Insects and disease agents can cause local to widespread defoliation and mortality in forests. They can even cause extinction of tree species, as was the case from chestnut blight (*Cryphonectria parasitica*) in the early 20<sup>th</sup> century. Some insects that have affected southern forests in the last 2 or more decades include southern pine beetle, which can cause rapid and widespread pine mortality, and also leads to removal of trees to prevent spread; hemlock woolly adelgid, which has caused widespread mortality of hemlock in the southern Appalachians; and spongy (formerly gypsy) moth (*Lymantria dispar dispar*), which can cause widespread defoliation, sometimes leading to mortality, in some years. As an example of how invasive insects can impact hydrologic processes, large-scale hemlock mortality from hemlock woolly adelgid resulted in decreased growing season evapotranspiration (by 24 to 37 percent) and increased water yield (by 15 percent) in New England forests with a high proportion of hemlock in the watershed, because the dying trees used less water (Kim et al. 2017). In the southern Appalachians where hemlock comprises a smaller proportion of the watershed, responses to hemlock woolly adelgid-caused mortality (representing a 6-percent decrease in stand basal area) indicated a decrease in water yield (by 8 percent), which was likely a result of species with higher evapotranspiration increasing in canopy dominance after hemlocks were killed by hemlock woolly adelgid (Brantley et al. 2014). Over the long term, water yield could decrease after hemlock decline even in watersheds formerly dominated by hemlock if co-occurring species with higher water use such as red maple (*Acer rubrum*), birch (*Betula lenta*), or tulip poplar (*Liriodendron tulipifera*), replace hemlock (Ellison et al. 2018).

Research on southern pine beetle impacts on the hydrologic cycle has been more limited than it has on hemlock woolly adelgid. Literature suggests that where southern pine beetle outbreaks lead to widespread mortality, salvage harvest, or large-scale removal of trees to prevent spread, the impacts would be similar to those of harvesting itself, generally decreasing evapotranspiration and increasing water yield immediately following tree canopy removal (Tchakerian and Coulson 2011). However, activity of southern pine beetle has been relatively low across the south over the past 2 decades, with the exception of recent outbreaks in Mississippi. Reasons for the low activity are unclear, but intensive silviculture practices and increased forest fragmentation in the South are two suggested hypotheses (Asaro et al. 2017), though increased beetle mortality from warmer summer temperatures does not appear to be a factor (Lombardo et al. 2022).

Knowledge of how insect- and disease-caused disturbances will change in the future is important for informing land management decisions and priorities. Predicting the precise future impacts on forests and water resources is difficult because of the uncertainties associated with climate and land-use change, the distributions of host trees and forest conditions, and the potential for future introductions of new insects and pathogens. However, two main factors influencing future insect or disease dynamics in forests are the effects of climate and land-use change on the insect or disease agent, and changes to the susceptibility or occurrence of host trees (Simler-Williamson et al. 2019). The scientific literature generally points to the potential for increased damage from forest pests in a warmer world, yet in reality pest responses may be more complex than that (Jactel et al. 2019, Kolb et al. 2016). On one hand, warmer climate could generally increase pest activity in forests due to shorter generation times (Weed et al. 2013) or range expansion of native pests into regions with newly suitable climate conditions (e.g., Lesk et al. 2017). On the other hand, warmer or more variable temperatures can lead to lower forest pest survival or reduced ability to disperse, limiting pest activity (Weed et al. 2013). Although the recent reduction in activity of southern pine beetle is not attributed to warming (Lombardo et al. 2022), further research is needed to determine whether future temperatures and heat waves could



be severe enough to lead to mortality in southern pine beetle or other forest pests in the South. Likewise, though drought may benefit some insects and tree pathogens, especially bark beetles and their symbionts, drought does not appear to lead

to increased southern pine beetle activity (Kolb et al. 2016). For example, recent widespread drought in Texas did not lead to outbreaks of southern pine beetle (Asaro et al. 2017) or other insects or pathogens.

### (3) What Are the Likely Implications of These Changes on Water Supply, Flood Protection, and Other Water-Related Ecosystem Services?

The natural capital embodied in forests of the Southern United States provides an array of ecosystem service benefits that, when properly managed, can sustain and enhance human well-being. Among the most critical linkages between healthy forest ecosystems and ecosystem benefits are the impacts that changes in land use have on water quantity and quality. Changes in forest extent and composition alter the supply of water-based ecosystem services through multiple channels including influences on streamflow availability, drinking water supply and purification, and provision of recreational opportunities (Holmes et al. 2017). Forests also help to store water during extreme precipitation events, thereby helping to mitigate the risk of flood- and landslide-related damages to individuals and communities. Accounting for the economic impacts of changes in ecosystem service flows resulting from alterations in forest cover such as drinking water treatment, water supply and demand, flooding, and landslides can help policymakers evaluate tradeoffs inherent in forest conservation efforts.

#### **Forest land use and water treatment costs**

Forest watershed protection is receiving increased attention as a possible means for improving water quality at drinking water treatment plants (DWTPs). Contaminants in drinking water supplies pose health risks for water consumers while also increasing the costs of water treatment. The Safe Drinking Water Act (1974) was amended in 1996 and mandated that States assess DWTP source waters for possible contaminants and authorized funding mechanisms for source water protection (Tiemann 2017). Dozens of forest conservation programs have been initiated across the country since then with the goal of protecting drinking water supplies (Gartner et al. 2017).

The rationale for these programs is supported by a suite of economic studies beginning in the 1980s that estimated statistical cost functions (e.g., Holmes 1988) demonstrating that improvements in source water quality (such as reductions in turbidity or total organic carbon [TOC]) lead to reductions in DWTP costs (reviewed by Price and Heberling 2018). A few studies have taken a further step by using economic models to investigate linkages between forest cover and DWTP costs. For example, research in France (Abildtrup et al. 2013) found that a 1-percent increase in regional forest cover (with an equivalent reduction in agricultural land) reduced average water supply costs by about 1.3 percent. A similar study in Malaysia found that reducing conversion of unlogged forest to nonforest use reduced DWTP costs by 0.47 percent while reducing conversion of logged forest to nonforest use reduced treatment costs by 0.31 percent (Vincent et al. 2016). Within the United States, a sample of 37 utilities found that DWTP costs increased with both TOC and turbidity and that both TOC and turbidity decreased with the amount of forest cover within the source water supply area (Warziniack et al. 2017).

Research indicates that the cost effectiveness of source water protection programs are highly dependent upon watershed characteristics as well as the technology used by DWTPs (Heberling et al. 2015). Further, watershed protection programs that enhance the value of ecosystem services (such as swimming, boating, fishing, and nature viewing) enhance the well-being of a variety of beneficiaries (Keeler et al. 2012). In these cases, the cost effectiveness of source water protection programs may be improved by distributing costs across multiple stakeholders (Gartner et al. 2017, Price et al. 2018).

Concern among members of the American Water Works Association (AWWA) regarding future availability of drinking water supplies and source water protection can be seen in responses of AWWA members to surveys that are annually conducted by this industry organization. Source water protection is a proactive approach to safeguard, maintain, or improve the quality or quantity or both of drinking water sources and their contributing areas (AWWA 2018). Survey questions use a 5-level Likert scale (unimportant, slightly important, important, very important, critical) to gauge the level of concern expressed by DWTPs regarding specific issues facing the industry. Among the top 10 issues facing the industry nationwide in 2020, long-term water supply availability was ranked number 3 and watershed/source water protection was ranked number 5 (AWWA 2020). Surveys conducted in 2021 reported that, nationwide, 79 percent of respondents ( $n = 1,095$ ) indicated that their utility has fully implemented or is in the process of implementing a source water protection program (AWWA 2021). Data from the 2021 AWWA survey for the 13 Southern States were compiled to determine the proportion of utilities responding that were very or critically concerned about long-term water supply ( $n = 742$ ) or source water protection ( $n = 718$ ). These responses indicate broadscale concern with long-term water supply (ranging from 76 to 100 percent of DWTPs, by State) and source water protection (ranging from 52 to 86 percent of DWTPs, by State) (fig. 17).

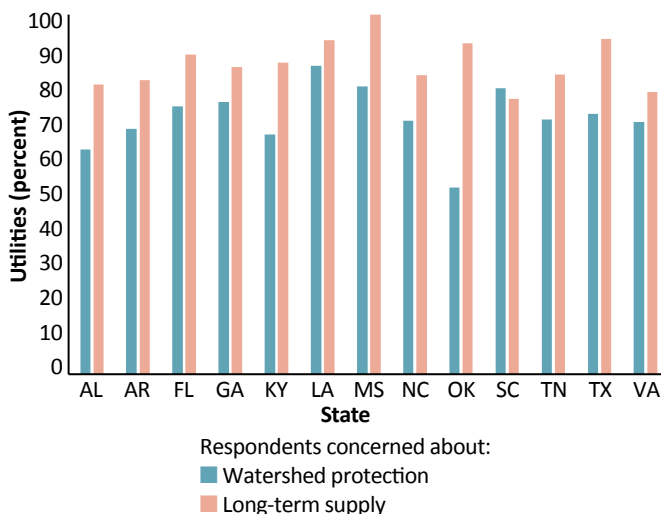


Figure 17—Percentage of utilities in the Southern United States responding that they were very or critically concerned about long-term water supply or source water protection, 2021. Data provided by the American Water Works Association.

Organizations such as the AWWA, Source Water Collaborative, U.S. Endowment for Forestry and Communities, Southeastern Partnership for Forests and Water, Keeping Forests, State forestry agencies, the Forest Service, and others are advancing linkages between forests and water supply, and building partnerships with water utilities to protect forested watersheds (Gartner et al. 2017). These partnerships are critical to watershed and source water protection, and can help forest landowners and managers and water utilities achieve their long-term management, economic, and water supply goals. Several resources provide guidance in developing source water protection plans and engaging partners (e.g., AWWA 2018, Ozment et al. 2016, TFPL 2013). In their Source Water Protection Justification Toolkit, AWWA (2018) outlines **six components of a successful source water protection program**:

- (1) **Protection vision**  
Water systems engage with stakeholders to develop vision for watershed protection.
- (2) **Characterization and assessment**  
Identify the most critical water quality and contamination concerns and inform program focus and priorities.
- (3) **Program goals and objectives**  
Establish goals and objectives to guide program. Goals should be specific and measurable, prioritized, and can capture current and future priorities and concerns.
- (4) **Action plan**  
Develop action plan identifying the roadmap for implementation including specific activities and priorities, and metrics for evaluating success.
- (5) **Implementation**  
Implement the action plan.
- (6) **Evaluation**  
Periodically evaluate the plan based on the metrics established and modify as needed.

Though the scope and complexity of these source water protection program components can vary greatly across water systems, the overall framework can apply to public water systems large and small, providing a structured approach to achieve mutual goals in watershed management. In addition, shared experience of challenges and opportunities learned from existing source water protection program efforts can help accelerate the development of new programs in other watersheds.

For example, based on examination of 13 source water protection case studies in the United States, Ozment et al. (2016) identified **10 key lessons for successful watershed investment programs:**

- (1) Identify risks** (wildfire, drought, etc.) and seize opportunities to rally support.
- (2) Build partnerships** to fill essential roles and responsibilities.
- (3) Articulate a clear vision of success.**
- (4) Cultivate champions and advocates** to build support (from water utilities, local government, nongovernmental organizations, landowners, etc.).
- (5) Develop a scientifically informed watershed plan.**
- (6) Evaluate the business case for investment.**
- (7) Identify investors** (water utilities, companies, foundations, etc.) and financing mechanisms for initial and long-term funding.
- (8) Engage landowners and public managers** to conserve, restore, and sustainably manage natural infrastructure.
- (9) Define roles and plans for program administration.**
- (10) Monitor and evaluate performance** (area of forest land protected, area treated for fire risk reduction, amount of sediment avoided from filling waterways, etc.).

Engaging partners and identifying stakeholders will contribute to the success of any source water protection program. All 13 case studies described in Ozment et al. (2016) began as a collaborative partnership across diverse organizations, which allowed for leveraging of unique skills, resources, and connections among partners. Essential partners for source water protection programs identified in Ozment et al. (2016) include investors (e.g., governments, utilities, water-dependent industries), land conservation and restoration suppliers (e.g., private forest landowners or public land managers), government regulatory agencies, scientific and technical experts, and public outreach groups.

There are multiple examples of successful source water protection programs linking stakeholders with water utilities in the South (SEPFW 2021). Among these is the collaboration among Central Arkansas Water (CAW), forest landowners and managers, and others that serves as a helpful case study to illustrate how a successful source water protection program can be implemented through partnership. CAW provides drinking water to

more than 450,000 people in greater Little Rock, AR, with water sourced from Lake Winona and Lake Maumelle, which are impoundments on the Alum Fork of the Saline and Big Maumelle Rivers, respectively.

Concern about water quality degradation in Lake Maumelle arose in the mid-1980s upon publication of projected development pressures in the watershed (Ozment et al. 2016). By 2004, two major subdivisions were proposed near the CAW intake on the Lake in high-risk areas for erosion and drinking water contamination (SEPFW 2021). This led CAW to develop a comprehensive watershed management program in 2007 in collaboration with many stakeholders to protect critical watershed land from development (CAW 2007). In 2009, the CAW Board of Commissioners approved a \$0.45 per month surcharge on each customer's drinking water bill for forested watershed management (SEPFW 2021). The fund was initially capped at \$3 million but in 2012 the cap was removed at the urging of community members (Ozment et al. 2016). These watershed protection fees and other funds leveraged from a Forest Service forest legacy grant, State appropriations, and wildlife management leases enabled CAW to acquire over 1050 ha (2,600 acres) and place 119 ha (295 acres) under conservation easements (Ozment et al. 2016). In total, CAW owns and manages more than 4000 ha (10,000 acres) of the 35 612 ha (88,000 acre) Lake Maumelle watershed. In 2020, voters in the city of Little Rock passed a \$30.6 million first-of-its-kind certified green bond to protect drinking water quality. About two-thirds of the bond in addition to other funding sources will be used for source watershed protection (SEPFW 2021). In addition to acquiring forest lands, CAW manages forests in their water supply watershed through prescribed fire and thinning (fig. 18). These management prescriptions are intended to reduce TOC loads to the reservoir that can result in the formation of regulated disinfection byproducts, and to make their forests more resilient to drought, pests, and disease (Lawson 2022).

The CAW case study shows how successful source water protection programs can be implemented through collaboration among stakeholders. Key lessons shared by Raven Lawson, Watershed Protection Manager-CAW in SEPFW (2021) include:





Figure 18—Prescribed burn at Lake Maumelle, AR. Source: Lawson (2022). (Photo courtesy of Bryan Rupar, Central Arkansas Water)

- Don't need to spend a lot of money and own land outright to be successful
- JUST DO IT!! Get some citizen champions to help you and build a good case
- Keep it simple to start—get it passed, then amend it later once the program is proven

## Water supply and demand

The economic and social significance of changes in streamflow (quantity and quality) must be assessed in the context of changing water demand. In the United States, water demand is driven primarily by three factors: population, irrigated area, and electricity water use (Warziniack and Brown 2019). At broad scales, the United States has experienced an overall decrease in water demand over the past few decades as a result of changing policies and increased efficiencies (Brown et al. 2019). However, at local scales, rapid population growth can overwhelm social (e.g., voluntary changes in individual water use), economic (e.g., price controls on water use), and technological (e.g., more efficient irrigation) tools intended to decrease water consumption (Gorelick et al. 2020). The implication is that the economic and social consequences of

changing water supply will necessarily require a decrease in demand, identification of alternative water supplies, or both. However, in areas where water demand cannot be meaningfully reduced, mitigating water stress may require changing water supply through interbasin transfers, shifting water sources from surface to groundwater (or vice versa) or increased reservoir storage or both (Brown et al. 2019). For example, Dallas, TX and neighboring municipalities are considering plans for new reservoirs and connection of additional existing reservoirs to their water supply infrastructure to meet municipal water needs by 2070 (Region C Planning Group 2020). While increasing water availability locally, new reservoirs will also have impacts on aquatic ecosystems and flow availability for downstream users. In some areas of the Southern United States, projections of increased precipitation and greater impervious cover that accompanies urbanization may help offset increased demand if the extra water can be captured and utilized (Gorelick et al. 2020). However, these changes reduce the proportion of flow from forests and increase the proportion of flow from impervious surfaces (figs. 14, 15) and thus can decrease water quality that may impact water treatment costs and reservoir storage



capacity (Holmes et al. 2017). A major consideration in shifting to increased reservoir storage is the very high cost associated with building new reservoirs. Depending on size and other characteristics, costs may exceed several hundred million dollars—and extend well into the billions of dollars range (Friedman 2019, Rogers 2020). Lower cost, greenspace alternatives may succeed under some circumstances as evidenced by the New York City Watershed Management example (Kenny 2006).

## Flooding and landslides

Increases in the magnitude and frequency of floods associated with increased precipitation intensity, land-use change, and sea-level rise could result in damage to property and water infrastructure, as well as alteration of forest landscapes. The relationship between forest cover and flood risk is receiving increased research attention because a warming climate is expected to increase extreme precipitation events (Donat et al. 2016, Laurance 2007). Flooding can impact a variety of economic sectors including private households, industry, agriculture, and infrastructure (Merz et al. 2010). For example, increased magnitude of high flows could cause damage to existing stormwater management infrastructure (e.g., culverts, detention ponds, combined sewer systems) and other infrastructure in or near flowing water (e.g., dams, bridges) where design storm magnitudes no longer reflect extreme flows.

Forest BMPs to mitigate impacts and risks are well developed and tested; however, the frequency and magnitude of extreme high flow events may require modifications of current BMPs. Because forests play an important role in mitigating high flows and thus reduce flood risks (Eisenbies et al. 2007), keeping forests on the landscape is important for reducing flood risks (Bathurst et al. 2020). In particular, maintaining (or restoring) floodplain forests is especially impactful because of functions such as storing flood waters, decreasing flow velocities, and strengthening streambanks (Mitsch and Gosselink 2015). However, floodplain forests are specifically adapted to historical flood regimes, thus changes to flood regimes will alter their condition, structure, composition, and function on the landscape (Fischer et al. 2021, Kozlowski 2002).

Several Federal agencies (e.g., Federal Emergency Management Agency [FEMA], EPA, Department of Homeland Security, Army Corp of Engineers) have focused on improving methods for identifying properties at risk of future flooding as well as developing a better understanding of the economic impacts of owning homes in alternative flood risk zones. Property-level flood risk data provide decisionmakers with information that can be used to develop flood risk adaptation and mitigation measures. Although data provided by FEMA are often used to assess flood risk, recent state-of-the-art modeling providing nationwide, high-resolution analyses of current and future flood risks to private property, and reflecting anticipated effects from climate change, suggest that FEMA analyses greatly undercount the number of properties at risk (FSF 2020). Data from this report indicate that substantial flood risk is anticipated to increase for all Southern States, except Oklahoma, between 2020 and 2050 (fig. 19), largely due to sea-level rise and changes in precipitation patterns (AWWA 2018, FSF 2020: 11). Although the number of properties facing substantial flood risk in the southern Appalachian Mountains is currently somewhat greater than in other noncoastal areas (FSF 2020: 10), little change in flood risk is anticipated for properties located in the physiographic region (FSF 2020: 11).

Increasing flood risks are likely to economically impact homeowners because of associated changes in property values. A recent meta-analysis of data from 37 studies, providing 349 estimates of the impact of flood risk on actual property values, showed that the discount for noncoastal properties located within a 100-year flood risk zone were 2.9

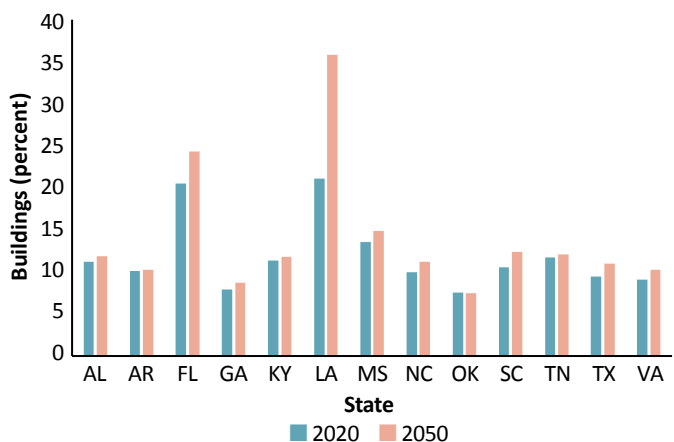


Figure 19—Percentage of buildings with substantial flood risk in 2020 and 2050, by State. Data source: First Street Foundation (2020).

percent, increasing up to 6.9 percent after a recent flood in the zone (Beltrán et al. 2018). Accounting for the fact that there is a significant recovery in property values following a flood, the authors conclude that inland properties located within designated 100-year floodplains are discounted at a rate of 4.6 percent. Given that >5 million properties throughout the Southern United States are located within 100-year floodplains (NYU 2017), the economic gains in home values that might result from mitigating existing and future flood risk through various policy measures are considerable and deserve future study. In addition, the human consequences of these changes require understanding the socioeconomic context in which the physical changes are occurring (Emrich and Cutter 2011). For example, Saia et al. (2020) used a risk-based approach to evaluate changes in flood risks (i.e., extreme high flows) from climate and land-use change in the Yadkin-PeeDee watershed in central North Carolina. Saia et al. (2020) combined projected changes in extreme flows with socioeconomic indices and found that the potential consequences of climate and land-use change driven increases in extreme flows are greater as the overall socioeconomic status in a watershed decreases. In other words, less socially vulnerable areas (e.g., more affluent) are better buffered against (and can recover faster from) extreme hydrologic events than more socially vulnerable (e.g., less affluent) areas. Wing et al. (2022) projected flood risk by 2050 across the United States, finding that relative flood risk may increase most along the Atlantic and Gulf coasts in the South, and disproportionately in black communities.

In addition, the authors found that projected population change could increase future flood risk four times that due to climate change alone.

Increased magnitude and frequency of precipitation could also result in greater landslide or debris flow occurrence (especially in steep terrain), imposing higher economic costs and risks to people. For example, Haque et al. (2019) estimated that 3,976 landslides between 1995 and 2014 caused a total of 163,658 deaths globally, and that the number of deadly landslides associated with extreme rainfall events increased during this period. FEMA identified several counties in the southern Appalachian Mountains with high landslide risk (using an index that includes the likelihood of a landslide occurring and the potential impacts on infrastructure and people), and some of these counties are projected to have large increases in population density in the future (Wear and Prestemon 2019). Underlying factors that determine the likelihood of a landslide occurring are complex and include physical (e.g., steepness, soil type and depth, land-use history, and more) (Wooten et al. 2016) and biological factors (e.g., vegetative cover and rooting depth) (Hales et al. 2009). Forest cover is an important stabilizing factor on steep upland hillslopes through precipitation interception, evapotranspiration, and root biomass (e.g., root reinforcement and soil cohesion). Human activity on forested mountain slopes, such as vegetation removal and road construction (e.g., cut slopes and fill slopes) can create initiation points that increase the risk of landslides (Wooten et al. 2016).

## (4) What Management Actions Best Minimize Impacts of and Increase Resiliency to Climate Change and Human Population Increase?

Forest harvesting (e.g., selection cutting, thinning, clearcutting) is one of the most widespread forms of “managed disturbance” in southern forests (Oswalt et al. 2019, Schleeweis et al. 2020). The effects of forest harvest on water quantity and quality in the Southern United States are well known (Jackson et al. 2004, Sun et al. 2004); however, the applicability of historical observations and the ability for current forestry BMPs to remain effective under future conditions is less certain. In this section, we

focus on how forest management, and the BMPs required to minimize adverse effects, may need to be modified in anticipation of climate change and other co-occurring disturbances and stressors.

### Water quality

Numerous studies clearly and uniformly suggest that current State recommended BMPs, such as Streamside Management Zones (SMZs), are effective at controlling nonpoint source pollutants

to levels compatible with water quality goals when properly implemented and maintained (Anderson and Lockaby 2011, Cristan et al. 2016, Croke and Hairsine 2006, Shepard 2006). In their review of 81 studies from across the United States, Cristan et al. (2016) found almost total agreement among studies regarding the effectiveness of BMPs, across variations in BMP specifications (e.g., different SMZ widths), when properly applied and maintained. Despite the widespread evidence of BMP effectiveness, several scenarios where effectiveness may become diminished are also apparent. For example, logging roads (Anderson and Lockaby 2011, Bilby et al. 1989, Jackson et al. 2004), stream crossings (Lang 2016, Lang et al. 2022), and installation of culverts (Aust et al. 2011, Morris et al. 2015) remain potential sources of erosion, particularly in steeper terrain and on finer textured soils. Furthermore, Rivenbark and Jackson (2004) showed that, although SMZs can be effective, SMZ integrity is a critical factor. The authors found an average of one SMZ breakthrough (i.e., a location where the SMZ is not effective) by surface runoff per 8 ha (20 acres) on a 1527 ha (3,773 acre) clearcut in the Georgia piedmont. These were principally associated with reactivated legacy agricultural swales and gullies with little vegetative ground cover on steeper slopes. Similarly, Litschert and MacDonald (2009) focused on sediment delivery pathways following harvests on national forest lands, and found rills and gullies to be prevalent conduits for sediment transport in spite of SMZ implementation. In the Virginia piedmont, Lang et al. (2015) reported that these breakthrough areas were mostly associated with stream crossings followed by legacy agricultural gullies and harvest-induced disturbances.

In addition to legacy agricultural gullies, legacy sediment in stream channels from historical forest harvesting and agricultural land can be a major source of sediment, even in forested watersheds where BMPs are implemented. For example, Anderson and Lockaby (2011) observed that increased water yield following harvests could resuspend legacy sediment in stream channels in regions such as the southeastern Piedmont where 19<sup>th</sup> and early 20<sup>th</sup> century agriculture resulted in massive export and deposition of topsoil into stream beds (Trimble 1974). Similarly, a postharvest increase in suspended sediment in the North Carolina piedmont, despite a 15 m wide

buffer, was attributed to resuspension of legacy sediment caused by increased discharge following harvest (Boggs et al. 2016). Some might speculate that increased rainfall could flush legacy sediment from stream beds entirely and reduce the potential for resuspension. However, in some cases, it has been estimated that thousands of years would be required to achieve this because of the massive amounts of historical sediment that have been deposited (Jackson et al. 2005).

Though previous studies suggest that BMPs and SMZs developed and tested under historical climatic conditions and disturbance regimes are generally effective when properly implemented, their applicability to future conditions with climate change driven shifts in precipitation regimes (e.g., drought and heavy rainfall events), temperature, and associated disturbances are less certain. Given the many uncertainties associated with future climate scenarios, quantitatively predicting the effects of future precipitation and temperature regimes on overland flow, erosion, and sediment export following forest harvests is challenging and beyond the scope of this report. However, it is likely that the effectiveness of existing BMPs will be challenged under future conditions as a result of projected increases in the frequency and magnitude of extreme precipitation events (see [question 1](#)).

In an effort to evaluate the response of forest soils to projected climate conditions, Hooimeier et al. (2016) estimated the rainfall erosivity factor in the Revised Universal Soil Loss Equation under three climate change scenarios for watersheds in eastern Tennessee. Projected rainfall erosivities increased by 10 to 50 percent by 2099 with concurrent projections for increases in sediment export of the same magnitude. However, some coal mining was present in these watersheds and it is unclear how much of the projected sediment increases were associated with that land use. Ciampalini et al. (2020) emphasized that potential increases in rainfall erosivity might be offset by concurrent increases in temperature through positive effects on net primary productivity. The offset would be manifested through greater leaf interception (reduction in rain drop energy) and improved infiltration (Ciampalini et al. 2020). Furthermore, revegetation of sites is well established as the most important biogeochemical stabilization and recovery mechanism following

forest disturbances and, consequently, enhanced net primary productivity could play a critical role in protecting soils in harvest zones. Dunkerley (2019) suggests that considering the possibility of increased frequencies of rainfall intensity bursts under future climate scenarios would be beneficial in fully assessing risks of increased erosion. Short-lived bursts of high intensity could become highly erosive through reduced infiltration and dislodgement of particles on particular soils. This is especially true on roads, decks, skid trails, and other areas where bare soil is created from logging operations.

Given the large-scale nature of climate change influences, combatting the effects of higher amounts and intensities of rainfall will require new strategies to cost effectively mitigate the cumulative, landscape-scale effects of new rainfall patterns. For example, the identification, prioritization, and management of critical source areas (CSAs) within watersheds or basins (e.g., 12- to 14-digit HUCs) could be effective if sufficient data are available to support the application of CSA assessment techniques (Dressing 2018). CSAs are defined as “areas of the landscape that contribute disproportionately high levels of nonpoint source pollution relative to other areas” (Vermont Water Quality 2022). Once CSAs are identified, management based on CSA identification can be highly cost effective, because resources are then allocated in proportion to the magnitude of nonpoint source water quality contributions within the watershed. The CSA concept focuses on identification of specific goals (e.g., maintenance of water quality within a basin), key sediment pathways from potential sources to streams, the most critical source areas (i.e., those with the most potential to influence goals), and management protocols to employ within the most critical CSAs (Dressing 2018). The CSA concept is versatile and may also be used to maximize water yields from CSAs in basins with marginal water supplies in tandem with expanding populations (Arteaga et al. 2020), a scenario that could become increasingly relevant in the Southeastern United States. Though CSA assessment has been widely used in State and regional water planning (e.g., Saraswat et al. 2013, Wilson et al. 2012), its application can be limited given that CSAs usually do not align with property or political boundaries (Kalin 2021).

A key management goal that requires more attention will be reduction of channelized flow and sediment transport through rills and gullies within CSAs. While noting the problems associated with these avenues of sediment transport, Lakel et al. (2010) also reported that SMZ widths of 7.6, 15.2, and 30.4 m were not significantly different in terms of sediment filtration, and averaged 97 percent effectiveness. Consequently, widening SMZs alone may not remedy increased erosion potential from higher precipitation amounts and intensities. Greater attention within CSAs at sealing SMZ breakthroughs using logging slash or turnout ditches or both may be required to address rising precipitation stresses.

Roads and stream crossings are the major concerns for water quality in forest watersheds and represent another major CSA (Aust et al. 2011, Boggs et al. 2018). It is estimated that the Forest Service manages approximately 600,000 km of roads and at least 40,000 stream crossings (i.e., road leadoff structures, fords, culverts, and bridges) (Heredia et al. 2016) that are mostly located in headwater watersheds with small drainage areas (Mannering and Kilareski 1998). The U.S. Department of Transportation (USDOT) (2018) recognized that most road crossings do not adequately accommodate bankfull flow conditions that occur about every 1 or 2 years. State BMP guidelines are available to minimize soil erosion from improper designs of harvesting access roads, skid trails, and stream crossings. A review study (Cristan et al. 2016) suggests that forestry BMPs protect water quality when constructed correctly and in adequate numbers. However, with increasing frequency and intensity of extreme storms, the size of permanent culverts or bridges and other hydraulic infrastructure designed with historic or offsite rainfall data or traditional rainfall-runoff models might not be sufficient to accommodate increased peakflow rate and stormflow volume (Amatya et al. 2021b). Existing hydrologic models need to be tested for forested watersheds under various topographical and climate conditions. Recent efforts focused on developing geospatial support tools for designing and assessing vulnerability of road culverts and stream crossings (Panda et al. 2022). Hydrological models incorporate design rainfall intensities and land morphologic features for identifying erosion hazards and vulnerability risks to road culverts



and stream crossings. These models are developed in an ArcGIS ModelBuilder platform (ESRI, Redlands, CA) and can assess streambank erosion and estimate potential erosion and streambank vulnerability for identifying locations of culverts and stream crossings that are morphologically vulnerable to erosion and scouring.

Anticipated increases in precipitation amounts and intensities imply that there is value in reassessing the balance between using water quality services from forested watersheds versus green infrastructure in urban areas. The oldest application of this concept in the United States are the two large, forested 10 522 ha (26,000 acre) watersheds that are owned by Greenville, SC in the mountains above the city where water management is the highest priority (The Nature Conservancy 2014). Other examples include the City of Austin Water Quality Protection Lands (Austin Water 2024) with 11 331 ha (28,000 acres) preserved, and the San Antonio Water System (San Antonio Water System 2024), which are both charged with protecting the Edwards Aquifer Recharge Zone (Edwards Aquifer Authority 2024). Management of woody vegetation through thinning and prescribed fire is considered effective for protecting drinking water quality while also improving groundwater recharge to aquifers, especially when paired with good coverage of native grasses and forbs. Therefore, in contrast to more mesic headwater source protection projects like Greenville, SC and the Catskills of New York (NYC Environmental Protection 2017) that focus on native forest cover, semiarid projects in Texas seek a more open savanna landscape for optimal water quality services. In all cases, practices that minimize erosion also tend to facilitate rapid stormwater infiltration and maximize aquifer recharge.

In summary, current standards for BMPs are generally effective under the historical climate regimes for which they were developed. It is likely, given general but common projections from many climate futures, that hydrologic stresses on forest BMPs—particularly on SMZs—will increase and, consequently, SMZ integrity will be tested. Higher rates of sediment export and concurrent decreases in water quality and site productivity are also possible in the future. Identification of critical source areas could be a useful approach

for identifying areas where current BMPs remain appropriate versus areas where BMPs may need to be revised.

## Water quantity

Decades of observations from paired watershed studies (e.g., Boggs et al. 2016, Ford et al. 2011a), and more recent observations from eddy covariance towers and sapflow studies provide a solid understanding of water yield and evapotranspiration responses to forest management (Aguilos et al. 2021, Oishi et al. 2010). In general terms, high and low flows usually increase if forest removal occurs in more than 12–25 percent of the watershed (Stednick 1996, Wei et al. 2021). However, the magnitude and duration of water yield increase depends on numerous biophysical factors, such as edaphic conditions, rate and composition of regrowth, imperviousness, and short-term climate conditions (Ford et al. 2011a). Longer term effects of forest change on water quantity depend on the structure and species composition of the postharvest forest (Elliott et al. 2017). What is less certain, however, is how hydrologic processes will change with forest management under future climate and disturbance regimes, and whether management practices should be modified to minimize negative effects on water yield (Vose et al. 2011). Because forests are often managed for multiple uses, managing forests to ensure reliable water supplies may conflict with other objectives such as maximizing carbon sequestration (Brantley et al. 2017, King et al. 2013).

Managing forests for water supply has been studied for decades. More than 40 years ago, Douglass (1983) synthesized decades of research from paired watershed studies in the Eastern United States and developed an empirical model of expected changes in water yield with forest cutting. Douglass (1983) concluded that water yield increases from large-scale forest cutting could meet the anticipated demands for drinking water at that time; however, the scale of cutting required would not be practical or acceptable to the public. One of the challenges for managing forests for water yield and mitigating the impacts of extreme events is the scale of management required to have meaningful broadscale impacts. This is especially challenging in fragmented forest landscapes that are largely privately owned, yet recent studies (Li, C. et al. 2020, Martin et al. 2017, Saia et al. 2020, Suttles et

al. 2018) suggest that these areas will show the greatest hydrologic responses (e.g., high flows) to climate change. The nexus of increasing demand for freshwater from a growing population, climate change, and associated disturbances and extreme events, has increased attention on the concept of managing forests for water yield (e.g., McNulty et al. 2021). These more recent syntheses have focused on less intensive options that may be more likely to be practical and acceptable to the public and landowners. In general, managing for water quantity focuses on two focal areas: managing species and managing stand structure.

### Managing species

In the Southern United States, many studies have shown that managed pine forest plantations use more water (i.e., greater evapotranspiration) than nearby unmanaged stands (Aguilos et al. 2021, Stoy et al. 2006, Swank and Douglass 1974). Part of the difference can be attributed to higher growth, but also structural differences that contribute to higher interception and wintertime transpiration in conifer forests. Water use can vary within genera as well. For example, at both the tree and stand level, longleaf pine generally has lower overall transpiration than loblolly pine or slash pine (*Pinus eliottii*) (Boggs et al. 2021, Brantley et al. 2017, Gonzalez-Benecke et al. 2011, McLaughlin et al. 2013), with differences attributed primarily to lower leaf area. A recent modeling study suggests that longleaf pine restoration (i.e., increasing the amount of longleaf relative to loblolly pine and cropland) has the potential to increase streamflow in the Flint River Basin (Qi et al. 2022). Sapflow studies have provided insights into water demands at the species level in stands as well (Cooper et al. 2018, Ford et al. 2011b). These studies suggest wide variation in sapflow among species driven at least in part by xylem anatomy, with diffuse porous species having the greatest sapflow (e.g., plant water use for transpiration) (Ford et al. 2011a, b; Guillen et al. 2022). Long-term paired watershed studies suggest that these species-level observations translate to the watershed scale, with streamflow declining as the proportion of diffuse porous species increases over time or after disturbance (Caldwell et al. 2016, Elliott et al. 2015, Jackson et al. 2004). Taken together, these results suggest that managing species composition through planting or other silvicultural treatments (e.g.,

thinning, prescribed fire) to favor low water-use species may have positive benefits on water yield (Ford et al. 2011a, Qi et al. 2022).

Though managing species composition has potential to enhance water availability, there are tradeoffs to consider. For example, the Southern United States is a strong carbon sink because of high net ecosystem exchange rates, where net ecosystem exchange represents the net balance between carbon gained from photosynthesis and carbon lost from respiration (Novick et al. 2015). In general, fast growing trees require more soil water to support higher productivity. As a result, higher water use (i.e., transpiration) can decrease soil water and reduce streamflow and groundwater recharge (Aguilos et al. 2021, Caldwell et al. 2016). Where water is abundant, efforts to grow forests for maximum productivity and carbon sequestration may have minimal impacts on water resources; however, in drier areas, high water use may have unintended negative consequences on surface and groundwater resources. Examples include the effects of afforestation in China (Sun et al. 2006) and planting fast growing Eucalyptus and pines in south Africa (Scott and Prinsloo 2008). In contrast, low water-use species may also have slower growth rates and hence lower carbon accumulation. In these cases, selecting (or breeding) and managing species with high water-use efficiency (i.e., carbon gain per unit of water used) may help optimize the tradeoff between water and carbon. In addition, the concept of “water zoning” provides another option to maximize tradeoffs. In water zoning, the fastest growing forests (that likely use more water) are managed preferentially in areas with the greatest water availability and discouraged in drier areas (McNulty et al. 2021).

### Managing stand structure

Stand structure refers to the physical characteristics of a forest stand that determine the amount of leaf area (i.e., LAI) and surface area of branches and stems. Transpiration is highly correlated with LAI and interception is correlated with total surface area of leaves, stems, and branches (Sun et al. 2011a). Leaf area, branch surface area, and stem surface area increase over time after clearcutting or other disturbances as the forest regrows, typically reaching a maximum LAI at canopy closure, although stem and branch area

may continue to accumulate (Brantley et al. 2019, Vose and Swank 1990). One option for reducing stand water use is to maintain forests at lower densities (and hence, lower leaf area) (Acharya et al. 2022, Pisarello et al. 2022). Paired watershed studies suggest that water yield increases are proportional to stand basal area, with a threshold of a 12 to 25 percent reduction in stand basal area required to observe measurable increases in water yield in the first year after cutting (Stednick 1996, Wei et al. 2021). Sun et al. (2015) used the WaSSI model to evaluate the potential of reducing LAI to offset climate change and found that simulated forest LAI reductions of 20, 50, and 80 percent resulted in increased annual water yields of 3, 8, and 13 percent, respectively. The rate of forest regrowth has an important role in regulating how long water yield increases last. For example, Liu et al. (2018) found that a 50-percent reduction in stand basal area in a loblolly pine plantation in the North Carolina coastal plain had a negligible effect on evapotranspiration due to rapid vegetation recovery at a wet site. This suggests that longer term benefits will require sustained management actions to maintain lower stand density and LAI (Pisarello et al. 2022). For example, in longleaf pine stands, lower evapotranspiration can be maintained by frequent prescribed fire that sustains an open forest canopy and limits hardwood encroachment (Qi et al. 2022). Less is known about fire impacts on evapotranspiration and water yield in hardwood forests and mixed species stands; however, a recent study observed increased water yield following a high severity fire in hardwood forests that lowered stand density and disproportionately killed diffuse porous species (e.g., tulip poplar, red maple, birch) (Caldwell et al. 2020a). In mixed loblolly pine and oak stands in Texas, impacts on evapotranspiration varied by fire severity, with evapotranspiration being higher for pines after low and moderate severity fires and higher for oaks after high severity fires (Cooper et al. 2019). Higher oak evapotranspiration after high severity fires was attributed to high pine mortality, whereas oak sprouted rapidly postfire (Cooper et al. 2018). Whether similar responses would be observed with frequent prescribed fire is not known. The expansion and increased density of fire intolerant mesophytic species (i.e., high water use, diffuse porous) in eastern hardwood forests is speculated to be driven, at least in part,

by fire exclusion (Nowacki and Abrams 2008). Vose and Elliott (2016) postulated that increased fire (prescribed or wildfire) in eastern forests could reverse this trend and increase water yield; however, some studies suggest that litter characteristics created by mesophytic species including rapid decomposition that reduces fuel loads, higher water content, and flat leaf shape that reduces oxygen circulation around the leaf litter, could limit prescribed fire application and effectiveness (Arthur et al. 2012).

The previous discussion focused on management practices to increase or maintain water yield, but hydrologic extremes (e.g., drought and heavy rainfall) present additional management challenges. Though trees are inherently resilient to drought (Vose et al. 2016), drought resiliency is not unlimited as evidenced by recent widespread forest drought-related mortality events in the Western United States (Vose et al. 2018). Where drought becomes more frequent and severe, managing to create or favor more drought resilient forests may be required. Many of these management actions are similar to those required to increase water yield (i.e., reduce density, favor low water-use species) so cobenefits are likely. Alternatively, the concept of “watering forests for the trees” (Grant et al. 2013) suggests that management activities such as changes in road design to route water away from ditches and into the forest, reintroduction of the American beaver (*Castor canadensis*) to increase water storage in localized wetlands, and expanded riparian zones could enhance forest resilience in the face of increasing drought—sometimes at the expense of increased water yield (Vose et al. 2019).

## **Urban BMPs and climate projections**

Future climate changes may be even more problematic for urban areas compared to their rural counterparts. Based on outputs from GCMs selected by the EPA (EPA 2018, Faizullahoy et al. 2018), rainfall increases of approximately 12 percent, and 1.5-fold increases in intensity were modeled for a site in Maryland with 65 percent impervious surface. Results indicate that runoff and maximum hourly peak flow would increase by 55 and 61 percent, respectively, with conventional infrastructure. However, mixing conventional with green infrastructure reduced those percentages by 69 and 16 percent, respectively. The increases

in runoff and peak flow with conventional infrastructure were associated with 75, 58, and 6 percent increases in sediment, total nitrogen, and total phosphorus loads, respectively. As was the case with runoff, mixing conventional with green infrastructure reduced those loads considerably.

In a small 0.8 ha (2 acre) Atlanta, GA site with 90 percent impervious surface, models suggest that significant increases in gray and green BMP footprints would be required to accommodate runoff, max hourly peak flow, and sediment and total nitrogen export responses to 1.1-fold increases in annual rainfall and 1.2-fold increases in max hourly precipitation volumes (EPA 2018). The necessary gray BMP footprint enhancement (e.g., concrete and steel components of urban drainage systems such as gutters and pipes) represented 84 and 24 percent footprint increases in underground sand filters and underground dry detention basins, respectively. Green BMPs (i.e., BMPs that approximate the unaltered water cycle, such as urban forests and rain gardens) with gray infrastructure required from no increase to as high as a 40-percent increase for bioretention areas with underdrains, underground dry detention basins, green roofs, and permeable pavement (EPA 2018). However, installing adaptation measures for both conventional and mixed conventional/green infrastructure in Atlanta, GA (90 percent impervious) is expensive—approximately \$2.96 million per impervious hectare (\$1.2 million per impervious acre) for conventional versus \$3.46 million per impervious hectare (\$1.4 million per impervious acre) for mixed conventional/green (Faizullabhoy et al. 2018). Much of the price with conventional infrastructure represents adaptation costs (approximately \$450,000), though only about \$200,000 is required for adapting mixed infrastructure (but initial installation costs are higher for the latter).

Although the modeling scenarios were driven by some of the more extreme climate model projections (EPA 2018), there is strong potential in both examples for higher runoff and rain intensities (along with the concurrent increases in sediment and nutrient loads as well as flooding) to occur to some degree in urban and suburban areas.

Similarly, significant costs will be associated with efforts to avoid or mitigate climate change impacts through infrastructure adaptations.

Traditional urban BMPs focus on maximizing soil infiltration and storm water storage to reduce surface runoff and thus reduce the speed of flows. Studies suggest that large water use from tree transpiration and canopy rainfall interception can play a significant role in reducing urban stormflow during the growing season (Boggs and Sun 2011; Li, X. et al. 2020), even during hurricanes. For example, under the influence of Hurricane Charley on August 14, 2004, the peak flow reached about 30 mm per day in an urban watershed, while a nearby forest dominated watershed had a small response (<1.0 mm per day) (fig. 20). Forest soils are drier due to higher evapotranspiration compared to urban lands and thus have larger water storage capacity to store runoff during extreme events. Following nature-based solution principles, urban development can seek to maintain less intensive land uses such as forests, grasslands, and wetlands as much as possible to not only reduce soil compaction but also retain the power of evapotranspiration to pump water to the atmosphere. Such water vapor returned to the air has additional benefits to mitigate urban heat and dry islands (Hao et al. 2023, Huang et al. 2022), and stormflow impacts.

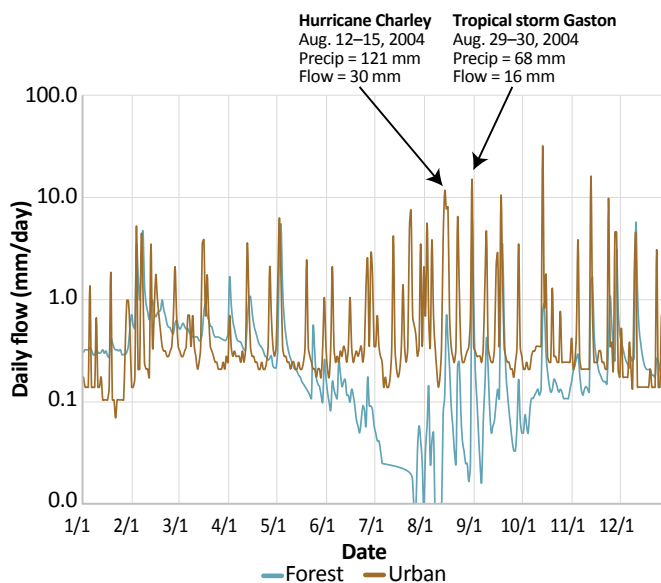


Figure 20—Hydrographs of forested and urban watersheds in the North Carolina piedmont ecoregion (Sun et al. 2021).



# Conclusions

The importance of forests for providing high quality, dependable freshwater has long been recognized. As a result, there is substantial information on (and understanding of) the relationships between forests and water in both managed and unmanaged systems. Despite this understanding, new challenges require updated assessments of risks and management options to reduce those risks. Since the SFFP, new knowledge and science-based tools are available to help provide the necessary information to keep watersheds functioning and capable of providing and sustaining ecosystem services into the future; however, integrating new science findings and tools into planning and management actions presents a challenge for both researchers and natural resource managers.

We used a combination of hydrologic modeling and synthesis of existing literature to answer questions that were identified as important to stakeholders. Our results suggest that changes in climate, land use, and forest conditions have and will continue to alter streamflow regimes and water quality through the 21<sup>st</sup> century, with consequences for drinking water treatment costs, flood protection, and other ecosystem services. Furthermore, these projected changes in hydrologic conditions will likely have complex feedbacks and interactions with other forest disturbances such as wildfire, drought, sea-level rise, hurricanes, nonnative invasive plants, and insect and disease outbreaks. In some cases, these interactions can exacerbate hydrologic changes from climate and land-use change alone. Hence, it will be important to minimize disturbance frequency, severity, and spatial extent—especially those disturbances that exacerbate undesirable hydrologic changes—through forest management and protection.

Water quality BMPs are well established and effective; however, it is not known whether they will remain effective under future climate regimes, especially higher intensity storm events.

As such, it will be beneficial to identify areas (i.e., CSAs) that contribute most to nonpoint source pollution to prioritize implementation of BMPs and evaluate their effectiveness under more extreme precipitation events. The combination of greater demand for freshwater and decreased water availability (e.g., either from drought or greater tree water use) also suggests value in considering management practices that decrease drought vulnerability and reduce evapotranspiration. For example, many of the mitigation and adaptation management actions used today will likely reduce drought vulnerability and benefit water quantity, including prescribed burning, thinning, and eradication of invasive species. On the other hand, forest evapotranspiration reduces soil water and thus enhances soil water storage capacity, which can help mitigate flood risk associated with increases in the magnitude and frequency of extreme events in the future.

Future forest management will continue to balance tradeoffs among management strategies that optimize water supply, flood protection, carbon sequestration, and other ecosystem services. Whether managing to protect water quality or maintain (or enhance) water quantity, a key determinant of success will be ensuring that the most vulnerable areas are identified and that management actions are implemented at a large enough scale and for a long enough duration to ensure both local and downstream benefits. Furthermore, forest water management strategies will require an adaptive approach (Golladay et al. 2016) with local knowledge given the inherent uncertainty of future water resources under a changing environment. Fortunately, increasing evidence suggests that in many cases, management actions can be effective at helping forests adapt to climate and land-use change, which can thus ensure the availability and quality of water resources in the future.

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# List of Acronyms

|       |                                                |
|-------|------------------------------------------------|
| AIC   | —Akaike information criterion                  |
| AWWA  | —American Water Works Association              |
| BMP   | —best management practice                      |
| CAW   | —Central Arkansas Water                        |
| CSA   | —critical source area                          |
| DWTP  | —drinking water treatment plant                |
| EPA   | —Environmental Protection Agency               |
| FAO   | —Food and Agriculture Organization             |
| FDM   | —forest dynamics model                         |
| FEMA  | —Federal Emergency Management Agency           |
| FIA   | —Forest Inventory and Analysis                 |
| GCM   | —general circulation model                     |
| GDP   | —gross domestic product                        |
| HUC12 | —hydrologic unit code, 12-digit                |
| IPCC  | —Intergovernmental Panel on Climate Change     |
| LAI   | —leaf area index                               |
| MODIS | —Moderate Resolution Imaging Spectroradiometer |
| RCP   | —Representative Concentration Pathway          |
| RPA   | —Resources Planning Act                        |
| SFFP  | —Southern Forest Futures Project               |
| SFO   | —Southern Forest Outlook                       |
| SMZ   | —streamside management zone                    |
| SSP   | —Shared Socioeconomic Pathway                  |
| TOC   | —total organic carbon                          |
| WaSSI | —water supply stress index model               |

## Appendix

**Table A1—Projected percent change in annual water yield in 2041–2070 compared to 1981–2010 across all scenario-climate futures by State**

| Scenario/GCM      | AL    | AR    | FL    | GA    | KY    | LA    | MS    | NC    | OK    | SC    | TN    | TX    | VA    | Southwide |
|-------------------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-------|-----------|
| -----percent----- |       |       |       |       |       |       |       |       |       |       |       |       |       |           |
| HL-dry            | -31.9 | -44.0 | -3.2  | -19.9 | -14.9 | -41.5 | -41.3 | -7.3  | -47.3 | -10.6 | -25.6 | -51.2 | -3.2  | -29.6     |
| HM-dry            | -31.7 | -43.9 | -2.5  | -19.6 | -14.7 | -41.2 | -41.1 | -7.0  | -47.0 | -10.4 | -25.3 | -50.6 | -3.0  | -29.3     |
| HH-dry            | -31.3 | -43.5 | -1.5  | -19.1 | -14.2 | -40.7 | -40.9 | -6.6  | -46.3 | -9.9  | -24.7 | -49.7 | -2.6  | -28.7     |
| LM-dry            | -24.8 | -19.3 | 1.0   | -20.1 | -7.5  | -28.6 | -25.6 | -10.5 | -25.5 | -13.9 | -15.6 | -33.8 | -10.3 | -19.8     |
| HL-hot            | -15.3 | -19.7 | -20.1 | -14.9 | -22.8 | -12.9 | -18.5 | -4.5  | -26.7 | -10.3 | -22.1 | -13.4 | -6.1  | -16.1     |
| HM-hot            | -15.1 | -19.6 | -19.5 | -14.6 | -22.6 | -12.5 | -18.3 | -4.3  | -26.4 | -10.1 | -21.8 | -12.6 | -5.9  | -15.7     |
| HH-hot            | -14.7 | -19.2 | -18.5 | -14.2 | -22.2 | -11.9 | -18.0 | -3.9  | -25.8 | -9.7  | -21.2 | -11.3 | -5.6  | -15.1     |
| HL-middle         | -2.5  | -11.3 | -11.4 | 3.2   | -4.9  | -13.6 | -8.2  | 6.7   | -3.8  | 7.9   | -2.9  | -14.0 | 2.9   | -5.7      |
| LM-middle         | -3.5  | -20.7 | 5.0   | 10.3  | -12.4 | -12.3 | -12.8 | 14.0  | -12.4 | 17.1  | -11.2 | -14.7 | 8.2   | -5.5      |
| HM-middle         | -2.3  | -11.1 | -10.8 | 3.6   | -4.7  | -13.3 | -8.1  | 7.0   | -3.4  | 8.1   | -2.5  | -13.3 | 3.2   | -5.3      |
| HH-middle         | -1.9  | -10.7 | -9.8  | 4.1   | -4.3  | -12.7 | -7.8  | 7.5   | -2.7  | 8.6   | -1.9  | -12.0 | 3.5   | -4.7      |
| LM-hot            | 7.3   | 3.7   | -3.2  | -1.6  | -4.2  | -2.5  | -3.1  | 10.8  | 2.0   | 5.8   | -3.9  | -3.3  | 5.9   | 0.4       |
| HL-wet            | 1.0   | 3.2   | 11.5  | 0.8   | 4.7   | -0.1  | 7.3   | 4.2   | -2.8  | 3.5   | 7.6   | -3.6  | 8.8   | 3.1       |
| HM-wet            | 1.2   | 3.4   | 12.2  | 1.1   | 4.9   | 0.2   | 7.4   | 4.5   | -2.4  | 3.7   | 7.9   | -2.8  | 9.0   | 3.5       |
| HH-wet            | 1.6   | 3.8   | 13.2  | 1.5   | 5.3   | 0.9   | 7.7   | 4.9   | -1.6  | 4.2   | 8.5   | -1.4  | 9.4   | 4.1       |
| LM-wet            | 2.4   | -3.5  | 11.7  | 5.0   | 8.1   | -1.1  | 2.7   | 14.8  | -4.9  | 15.5  | 7.5   | 2.9   | 14.0  | 4.9       |
| LM-least warm     | -0.7  | 8.4   | 16.1  | 7.1   | 0.0   | 8.6   | 3.5   | 3.2   | 3.3   | 4.5   | -0.6  | 10.4  | 6.2   | 6.0       |
| HL-least warm     | 7.2   | 12.5  | 16.9  | 18.0  | -2.1  | 2.5   | 4.4   | 16.1  | 4.6   | 23.4  | 0.8   | 1.4   | 15.6  | 8.3       |
| HM-least warm     | 7.4   | 12.7  | 17.7  | 18.4  | -1.9  | 2.9   | 4.5   | 16.4  | 5.0   | 23.6  | 1.2   | 2.2   | 15.8  | 8.7       |
| HH-least warm     | 7.8   | 13.2  | 18.8  | 18.9  | -1.5  | 3.5   | 4.8   | 16.9  | 5.8   | 24.1  | 1.8   | 3.6   | 16.2  | 9.4       |

Scenario-climate futures are sorted according to their rank for the entire South (driest to wettest), and colored according to the scenario-climate futures rank for each State; browns are decreases and blues are increases in projected water yield. Climate future scenario details can be found in tables 1 and 2.

GCM = general circulation model.



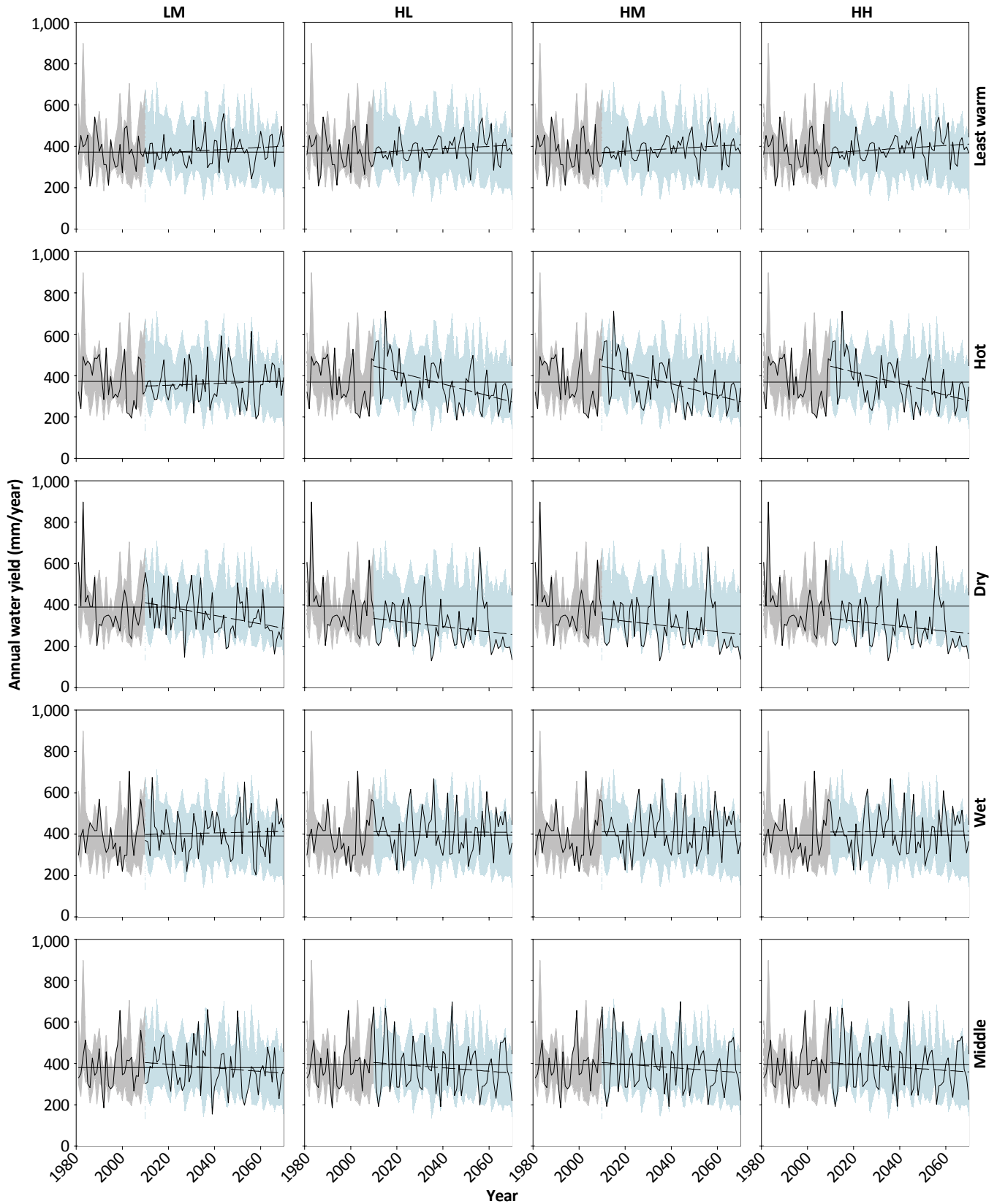


Figure A1—Southwide annual water yield (solid lines) by scenario climate future (tables 1 and 2) and range in projected annual water yield across scenario-climate futures (shaded area). Solid horizontal line is the 1981–2010 mean annual water yield, and dashed line is the future trend in 2011–2070 mean annual water yield for each scenario climate future.

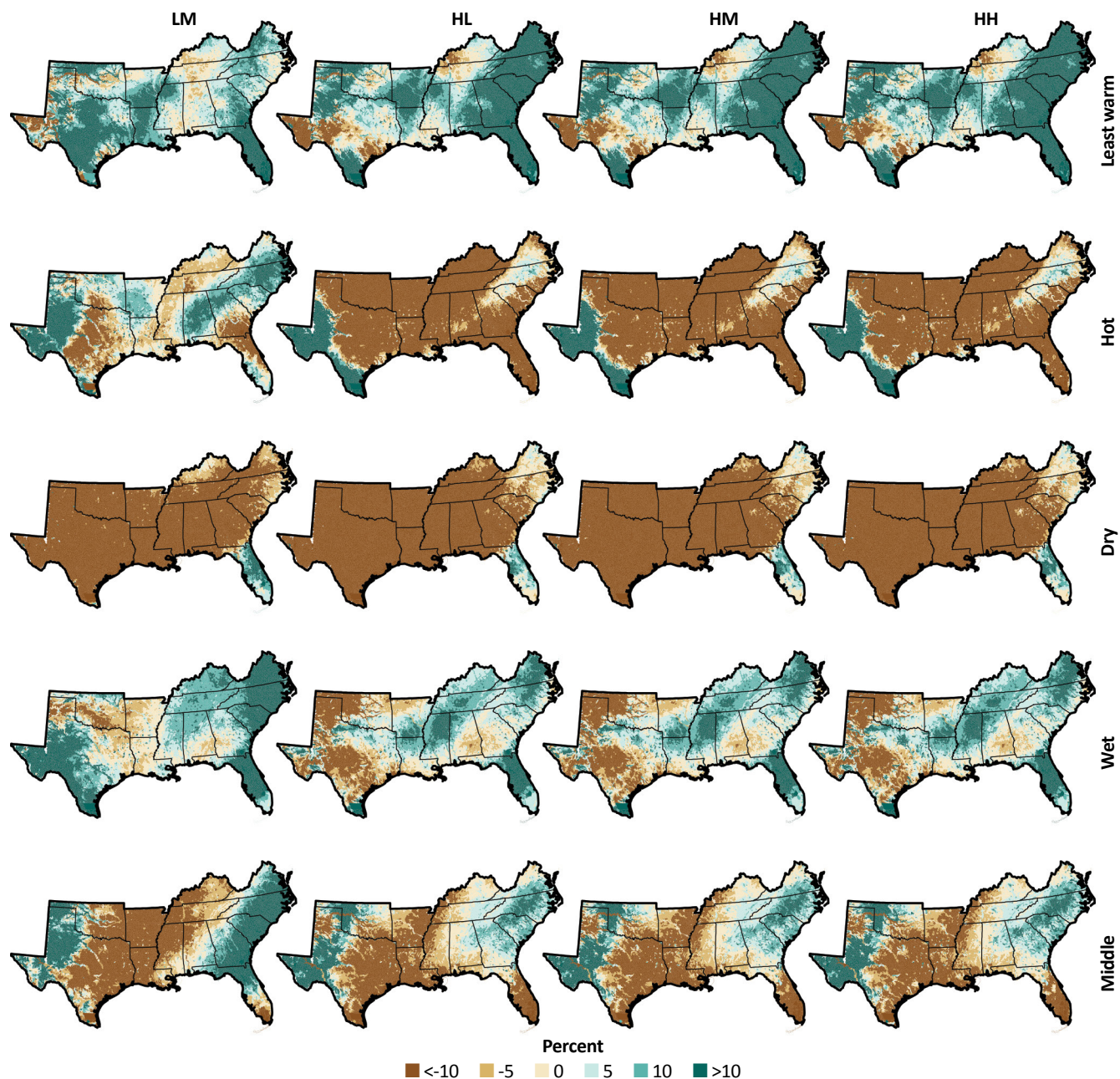


Figure A2—Percent change in mean annual flow between 1981–2010 and 2041–2070 by scenario climate future (tables 1 and 2).

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The linkages among forests, water, and people have long been recognized by natural resources professionals and policymakers. This report, in support of the U.S. Department of Agriculture, Forest Service, Southern Research Station's *Southern Forest Outlook*, provides a contemporary assessment of current and future forest and water resource issues in the Southern United States, a region encompassing the 13 Southern States from Virginia to Texas. This report uses a combination of literature review, hydrologic modeling, and data synthesis to address pressing research questions, leveraging scenarios of future climate, land use, and forest conditions developed as part of the 2020 Resources Planning Act Assessment. Results show that climate and land-use change have and will continue to alter streamflow regimes and water quality through the 21<sup>st</sup> century, with consequences for drinking water treatment costs, flood protection, and other ecosystem services. These impacts will present broad-scale and complex challenges for land and water resources managers. Fortunately, increasing evidence suggests that in many cases, management actions can be effective at helping forests adapt to climate and land-use change, which can ensure the availability and quality of water resources in the future.

**Keywords:** Climate change, water yield, water supply, ecosystem services, land-use change, disturbance interactions, management actions.





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