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# 8 Productivity and Carbon Sequestration of Forests in the Southern United States

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Sixty percent of the Southern United States landscape is forested (Wear 2002). Forest types vary greatly among the five subregions of the South, which include the Coastal Plain, Piedmont, Appalachian-Cumberland, Mid-South, and the Mississippi Alluvial Valley. Current inventory data show upland hardwood forests being the predominant forest type in the South (>30 million ha) followed by planted pine (>15 million ha), natural pine and bottomland hardwoods (~13 million ha), and oak-pine (>3 million ha) forest types (Huggett et al. in press). These forest ecosystems provide a multitude of ecosystem goods and services including clean water and air, wildlife habitat, recreation and aesthetics, timber and fiber production, and CO<sub>2</sub> sequestration. Southern forests play an important role in meeting the current and future timber and fiber needs across the United States, as harvesting has substantially decreased, in other regions of the country. As a whole, the South's forest sector produces approximately 60% of the total U.S. wood production, more wood than any other single nation (Prestemon and Abt 2002).

The future of forests and forest management in the Southern United States face many uncertainties. Land-use change, population growth, urbanization, changing public values, unstable timber markets, and a changing climate are some of the factors that will influence forests and forest management in the South (Wear and Greis in press). Although forest management is unable to address the anticipated changes in forestland availability due to shifts in demography (e.g., increasing urbanization, land-use change, etc.), it can be used to increase the adaptability of forests to changing climate conditions. Proactive forest management that focuses on increasing resistance (i.e., reducing a stand's susceptibility to disturbance) and resilience (i.e., reducing the negative effects of disturbance and/or reducing recovery time following disturbance) may ameliorate some of the anticipated changes in forest structure, function, and productivity anticipated under a changing climate. Climate change is likely to affect forest structure and function through a variety of pathways, such as changing disturbance patterns, shifting species composition, and altering growth and productivity. Because the South produces the vast majority of wood-related raw materials, long-term trends in forest productivity are especially important in assessing whether southern forests can continue to meet the nation's timber and fiber demands in the context of a changing climate.

For large-scale planning purposes, models of forest growth and yield and/or forest stand dynamics have been traditionally used to project the growth and productivity of forest stands into the future. These models are a valuable tool in our conceptual understanding of how forests grow and of the products they can produce. However, existing growth and yield models or ecological process models essentially operate in a twentieth-century environment, because the long-term studies on which these models were based have largely been conducted, and algorithms developed from them, prior to the turn of the twenty-first century. Empirical forest growth models have been developed using common static measures of site productivity (i.e., site index [SI]) without direct consideration of fluctuating climatic variables. Consequently, parameter estimates in these models will not automatically adjust in response to the projected changes in climate, leaving the expected increases or decreases in forest productivity unaccounted for in model predictions. Current growth models, including empirical, process, and/or hybrid models (see Medlyn et al. 2011 for a review of models), should, therefore, be examined to determine how the interactions between climatic and nonclimatic factors limit stand-level productivity.

Since the 1950s forest productivity, as defined as the change in growing stock volume over time, in the South has steadily increased (Smith et al. 2007). During this period, net volume per hectare increased 95% in the Southern United States (Smith et al. 2007). In 2010, productivity of southern forests was estimated at 8.3 billion cubic meters, with softwood and hardwood growing stock approximating 3.4 and 4.8 billion cubic meters, respectively (Huggett et al. in press). Past increases

in forest productivity can be attributed to a variety of factors, including a reduction in harvesting on National Forest System lands, afforestation activities, intensive forest management, an increase in the area converted from natural forest conditions (e.g., upland hardwood and/or natural pine forest types) to planted pine, and the age class distribution of forest stands across the region (Hicke et al. 2002).

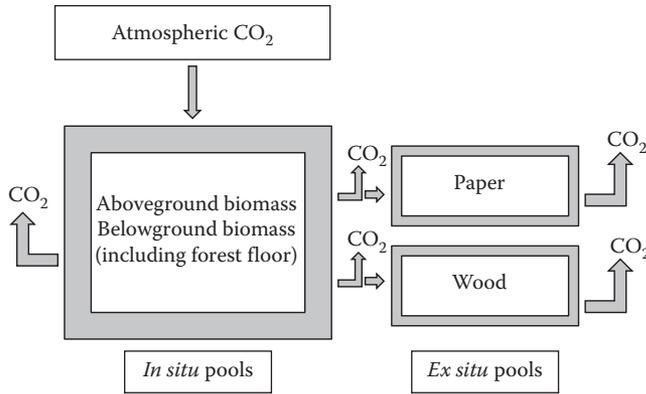
The increase in forest productivity observed over the last four decades in the South has coincided with an identifiable change in climate. Since the 1960s, the average annual temperature across the Southern United States has increased (McNulty et al. in press). Although no significant trend in average annual precipitation has accompanied the increase in temperatures (McNulty et al. in press), patterns of precipitation have been altered, with areas experiencing moderate to severe spring and summer drought increasing across the South (Karl et al. 2009). Under future emission scenarios outlined in the Southern Forest Future's Project (SFFP), average annual temperatures in the Southern United States are expected to increase by an additional 2.5–3.5°C by 2060 (Wear and Greis 2012). Models predicting precipitation patterns are less consistent and show a high degree of spatial variability across the South's 13 states. Of the four climate scenarios reported on utilized in the SFFP, the MIROC3.2 + A1B scenario forecasts the most drastic decrease in precipitation, with up to a 24% reduction, relative to 2010 levels, in average annual precipitation forecasted by 2040 (McNulty et al. in press). The remaining climate change scenarios (CSIROMK3.5 + A1B, CISROMK2 + B2, HadCM3 + B2) predict that by 2040, average annual precipitation across the South may decrease (but to a lesser degree than the MIROC3.2 + A1B scenario), remain similar, or even increase relative to 2010 levels, depending on subregion (McNulty et al. in press). In addition to forecasted changes in total annual precipitation, the seasonality of precipitation may be altered, depending on climate change scenario and subregion.

Climate variables, including temperature and precipitation, exert a strong influence over site productivity. The expected trend of increased temperatures and decreased growing season precipitation under various climate change scenarios has implications for the future productivity of southern forestlands. Although factors controlling forest productivity in terms of net primary productivity (NPP) are numerous and include biotic factors such as leaf area, the efficiency of the foliage to absorb solar radiation [i.e., light use efficiency ( $\epsilon$ )], soil nutrient availability, and species composition, abiotic factors that will be altered under a changing climate, including the amount of solar radiation, temperature, and available water (Churkina and Running 1998; Hicke et al. 2002; Running et al. 2004), have the greatest control over productivity. Global analyses indicate that NPP in the common forest types of the South is limited by available water and temperature (Churkina and Running 1998; Running et al. 2004). Solar radiation only represents a limiting factor in the tropics due to cloud cover (Running et al. 2004).

Therefore, climate change clearly has the potential to impact forest productivity and thus carbon (C) sequestration. Land occupied by southern forests represents 30% of the total forestland in the United States (Han et al. 2007). Owing to their extent and high productivity, southern forests have also been estimated to account for 36% of the C sequestered in the conterminous United States (Turner et al. 1995). Han et al. (2007) estimated forests in the South sequester 13% of regional greenhouse emissions.

Carbon can be sequestered via southern forests by two main routes: *in situ* and *ex situ* C sequestration (Figure 8.1, Marland and Marland 1992; Johnsen et al. 2001a, Gonzalez-Benecke et al. 2010). Carbon sequestered *in situ* is C that is tied up in aboveground and belowground biomass and necromass. Carbon is sequestered *ex situ* via incorporation into wood products that store C away from the atmosphere for different durations depending on the forest product type. As described below, gross primary productivity (GPP) is a critical component of *in situ* ecosystem C sequestration. GPP is the component of ecosystem C sequestration most amenable to influence by forest managers even in the event of climate change. Thus, forest managers can directly influence the rate of C sequestration (Johnsen et al. 2001a; Ryan et al. 2010; McKinley et al. 2011).

In this chapter, we review how forest productivity and C sequestration are related, discuss the impacts of selected press and pulse effects (Chapter 2 and briefly described below) on productivity



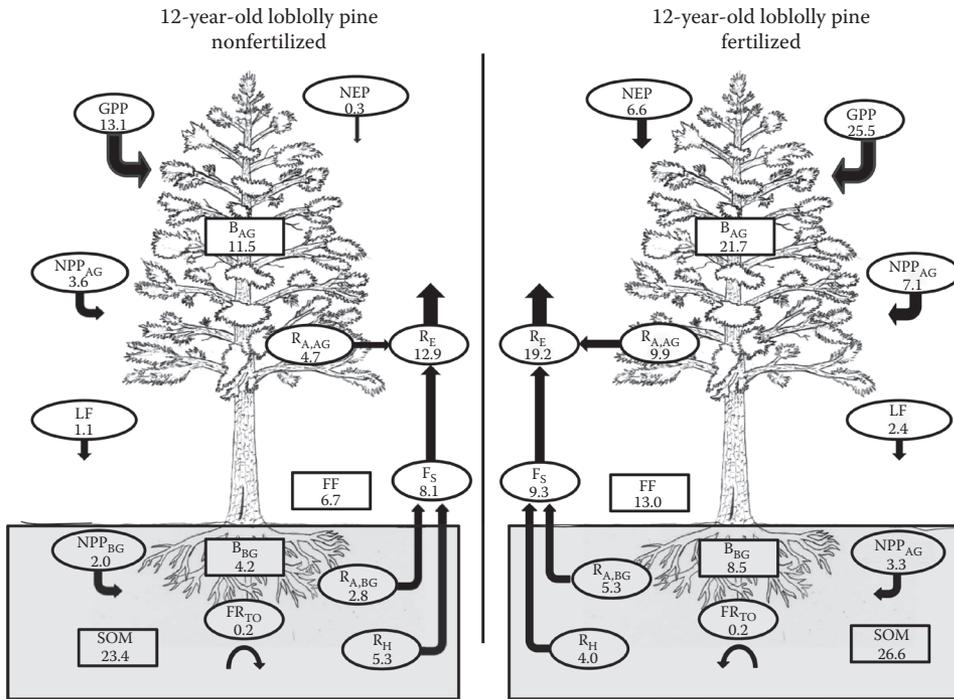
**FIGURE 8.1** Conceptual model of carbon sequestration via southern forests. (Adapted from Johnsen, K.H. et al. 2001a. *Journal of Forestry* 99:14–21.)

and/or C sequestration, model the potential effects of climate change on forest productivity in the Southern United States, examine several case studies that highlight the potential impact of climate change and/or the impact of forest management on forest productivity and/or C sequestration, and, finally, consider general management options, including the potential of bioenergy production as they impact productivity and C sequestration options. We concentrate on biological climate change effects on forest productivity and C sequestration, excluding important impacts of land-use change, so that we can address strategies that land-use managers can use to ameliorate the impacts of climate change and/or maintain or increase C sequestration at the stand level. Impacts associated with land-use change including issues of leakage, permanence and disturbances are reviewed by McKinley et al. (2011).

## IN SITU CARBON SEQUESTRATION

### THE FOREST CARBON CYCLE: THE RELATIONSHIP BETWEEN NET PRIMARY PRODUCTION, GROSS PRIMARY PRODUCTION, NET ECOSYSTEM PRODUCTION, AND ECOSYSTEM RESPIRATION

Southern forests contain about 30% of the nation's C stock (Mickler et al. 2004) and play a prominent role in the regional and global C cycle (Turner et al. 1995). Forests exchange large amounts of C, as CO<sub>2</sub>, with the atmosphere and store C in living plant biomass, detritus, and soil (Figure 8.2). Ecosystem C accumulation is essentially the balance between C gain and loss through photosynthesis and respiration. Carbon gain or GPP is the sum of individual leaf photosynthesis and represents the total C input into the ecosystem (Figure 8.2). GPP is integrated over space and time—typically one year—and is often expressed in terms of C (e.g., Mg C ha<sup>-1</sup> year<sup>-1</sup>). Annual Forest GPP is a function of leaf area index (LAI, leaf surface area per unit ground area) and the duration of display (Chapin et al. 2002). The biophysical mechanisms (e.g., light, temperature, and moisture) regulating annual GPP are well understood and can be modeled with reasonable accuracy (e.g., Landsberg and Waring 1997). Conversely, the mechanisms controlling C allocation to the growth and maintenance of different stand components are less well understood. About half of annual GPP is lost through autotrophic respiration ( $R_A$ ), which is the total release of C, as CO<sub>2</sub>, from all living primary producers, and represents the energy required for tissue growth, nutrient uptake and transport, and tissue maintenance. The balance between GPP and  $R_A$  is NPP ( $NPP = GPP - R_A$ ); the net flux of C from the atmosphere into organic matter (i.e., foliage, branches, stems, reproductive organs, and roots). Forest NPP is a fundamental ecological variable because it measures the amount of energy input that drives ecosystem metabolism (Chapin et al. 2002). NPP is usually measured as the increment of new biomass or C equivalent (e.g., Mg C ha<sup>-1</sup> year<sup>-1</sup>), but also includes C loss in root exudates, herbivory,



**FIGURE 8.2** Major carbon pools (Mg C ha<sup>-1</sup>; boxes: B<sub>AG</sub>—aboveground biomass, B<sub>BG</sub>—belowground biomass, FF—forest floor, SOM—soil organic matter) and fluxes (Mg C ha<sup>-1</sup> year<sup>-1</sup>; oval: NEP—net ecosystem productivity, GPP—gross primary productivity, NPP<sub>AG</sub>—aboveground net primary productivity, NPP<sub>BG</sub>—belowground net primary productivity, LF—litterfall, R<sub>A,AG</sub>—aboveground autotrophic respiration, R<sub>A,BG</sub>—belowground autotrophic respiration, R<sub>H</sub>—heterotrophic respiration, F<sub>S</sub>—soil CO<sub>2</sub> efflux, R<sub>E</sub>—ecosystem respiration, FR<sub>TO</sub>—fine root turnover) for a 12-year-old loblolly pine plantation (Maier et al. 2004). F<sub>S</sub> is the combined respiration of heterotrophic soil microbes and plant roots. R<sub>H</sub> was estimated as the difference between F<sub>S</sub> and root respiration (R<sub>H</sub> = F<sub>S</sub> − R<sub>A,BG</sub>) (Maier and Kress 2000). Estimate of the carbon flux in annual FR<sub>TO</sub> (0.22 year<sup>-1</sup>, Johnsen et al. 2005) are not included in NPP<sub>BG</sub> pool.

and volatile emissions (Chapin et al. 2006). Annually these additional C losses are relatively small (<5%), but may be a substantial component of the C budget when assessed over many years or after large episodic disturbances (e.g., insect outbreaks). As the products of NPP die, dead organic matter or detritus accumulates on the forest floor and soil. Decomposition of detritus releases C to the atmosphere through heterotrophic respiration (R<sub>H</sub>) and represents the second largest C flux from the ecosystem (Figure 8.2). Carbon not consumed in R<sub>H</sub> accumulates in the soil, retained in chemically bound organomineral aggregates resistant to decomposition. This recalcitrant C pool can persist for thousands of years and accounts for most of the ecosystem carbon storage in some forest ecosystems.

Allocation of NPP into various ecosystem components (e.g., foliage, stems, and roots) determines ecosystem structure and can have a strong effect on other ecosystem processes such as biogeochemical cycling, water use, and C sequestration. The physiological and biochemical mechanisms that control allocation, particularly belowground, is only partially understood (Litton et al. 2007). Carbon allocated belowground supports root production and maintenance as well as rhizosphere food webs or mycorrhizae, and so provides the majority of soil detrital C. Plants allocate C to minimize water and nutrient limitations (Friedlingstein et al. 1999), shifting allocation belowground to support root growth and resource acquisition at the expense of aboveground growth as soil resources become limiting (Haynes and Gower 1995; Albaugh et al. 1998). In general, as site GPP increases,

the proportion of GPP allocated belowground decreases (Litton et al 2007). Because 20–80% of GPP is allocated belowground (Litton et al. 2007), small changes in this percentage in response to climate change may greatly alter ecosystem productivity and soil C storage.

The C use efficiency (CUE) or the ratio of NPP to GPP reflects the ecosystems C storage potential. CUE is relatively conservative at the individual plant level because physiologically respiration is linked tightly to photosynthesis (Amthor 1994). Gifford (2003) suggested that a constant CUE also operates at the ecosystem scale. Waring et al. (1998) found that CUE was relatively constant ( $\approx 0.47$ ) across a range of forest types and environmental conditions, suggesting that the ecophysiological controls on NPP and GPP are the same in most forest ecosystems. A constant CUE greatly simplifies modeling NPP and GPP because it obviates the need to measure component  $R_A$ , a notoriously difficult process to measure (Landsberg and Sands 2011). However, the constant CUE reported in Waring et al. (1998) may be an artifact of estimating both GPP and  $R_A$  from NPP (Medlyn and Dewar 1999). DeLucia et al. (2007) analyzed a large number of studies where GPP was estimated independent of NPP, and found that CUE varied widely between forest types (0.2–0.8) and that it decreased with stand age. A better understanding of the ecophysiology  $R_A$  is needed to reliably predict how climate change effects forest CUE (DeLucia et al. 2007).

## HETEROTROPHIC RESPIRATION AND NET ECOSYSTEM PRODUCTIVITY

As the products of NPP die, dead organic matter or detritus accumulates on the forest floor and in the soil. Decomposition of dead organic matter releases C to the atmosphere through heterotrophic respiration ( $R_H$ ) and is a major component of ecosystem metabolism (Figure 8.2). Generally, factors that favor high NPP also contribute to high  $R_H$  (Lambers et al. 2006). Detrital C not used in  $R_H$  accumulates in the soil and is retained in chemically bound organomineral aggregates resistant to decomposition, which can persist for hundreds to thousands of years. Soil C is an important C pool and accounts for most of the ecosystem C in some forest ecosystems.

Net ecosystem productivity (NEP) is the rate that C accumulates in living biomass, detritus, and soil, and defines the ecosystems ability to sequester C. In most forest ecosystems, NEP is the difference between GPP and ecosystem respiration ( $R_E$ ), where  $R_E$  is the sum of  $R_A$  and  $R_H$  (Figure 8.2). As defined here, NEP does not include C movement through leaching or lateral transfer of dissolved inorganic and organic C or loss through emissions of volatile organic compounds, methane, or CO, which may represent important C fluxes over large spatial or temporal scales or in certain ecosystems (e.g., forest wetlands) (Chapin et al. 2006). NEP is relatively small compared to its component fluxes of GPP and  $R_E$ . If GPP exceeds  $R_E$ , then NEP is positive and the system is capturing C (i.e., C sink); however, if  $R_E$  exceeds GPP, then NEP is negative and the system is losing C (i.e., C source). NEP can vary from year to year owing to different environmental responses of GPP,  $R_A$ , and  $R_H$ . Changes in temperature, precipitation, length of growing season, or CO<sub>2</sub> will directly affect GPP, resulting in either increased or decreased NEP. However, the interannual variability of  $R_E$  is more complicated because of the disparate response of  $R_A$  and  $R_H$  to temperature, moisture (Falge et al. 2002), and substrate availability (Johnsen et al. 2007).

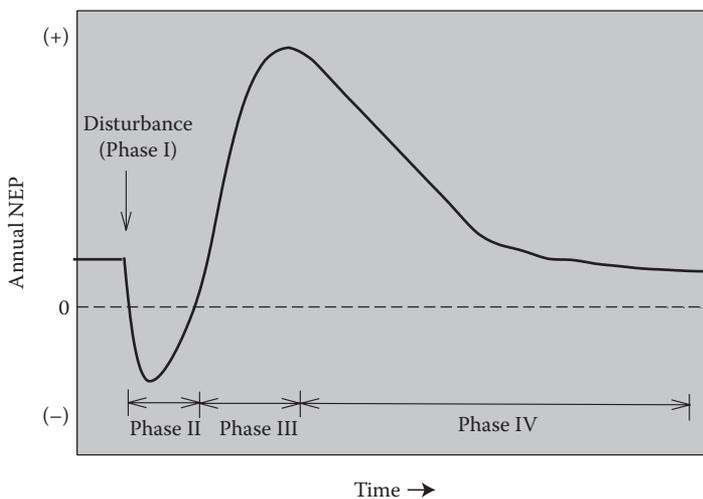
Quantifying NEP, GPP, and  $R_E$  is a formidable task even for a single site. The ecological “bottom-up” approach uses biometric measurements of NPP and chamber-based measurements of  $R_A$  and  $R_H$  to estimate C flux of component C pools and then sums these values over space and time to estimate NEP (Figure 8.2). This approach allows for determining how ecosystem components contribute to NEP and how these components respond to the environment. The meteorological “top-down” approach utilizes eddy-covariance measurements of energy and mass exchange between the forest canopy and the atmosphere to estimate net ecosystem CO<sub>2</sub> exchange (NEE) (Landsberg and Sands 2011). It differs from the ecological approach in that it integrates NEE over large areas and thus provides direct ecosystem level estimates of GPP and  $R_E$ . Conceptually, NEP estimated from ecological analysis and NEE is the same because both comprise the difference between GPP and  $R_E$ ; however, each method measures these components at different spatial and temporal scales

and thus do not always agree (Baldocchi 2003). NEE also differs from NEP in that it does not account for nongaseous transfer of C into and out of the system via leaching, lateral transfer, herbivory, and harvest (Chapin et al. 2006). In addition, annual estimates of NEE and NEP can diverge because of a lag between C fixation and biomass growth; however, estimates between the two methods converge when averaged over multiple years (Baldocchi 2003; Gough et al. 2008). In the scientific literature, NEE and NEP are often expressed in opposite sign. This is because atmospheric scientists define NEE as the net C flux from the ecosystem to the atmosphere, whereas ecologists define NEP as the net C flux from the atmosphere into the ecosystem (Chapin et al. 2006). While the eddy-covariance technique has become a standard method for estimating NEE, in most studies, it is combined with the ecological method to estimate NEP and component processes (Baldocchi 2003). This complementary approach has provided robust estimates of NEP and component processes for a number of southern forest ecosystems (Ehman et al. 2002; Curtis et al. 2002; Lai et al. 2002; Clark et al. 2004; Noormets et al. 2010; Goulden et al. 2011).

### DISTURBANCE AND NET ECOSYSTEM PRODUCTIVITY

Forests in the Southern United States are characterized by frequent disturbances from natural (e.g., fire, wind and ice storms, drought, insects, and disease) and human-induced (e.g., harvesting) causes (Dale et al. 2001). The type of disturbance (consumptive or nonconsumptive) and management intensity will determine the amount and type of detrital C and the trajectory or recovery of NEP over time (Sprugel 1985). Nonconsumptive disturbances such as windstorms or disease transfer carbon directly from living biomass to forest floor and soil detrital pools, leaving almost all biomass on site. Consumptive disturbances (e.g., fire), remove large amounts of carbon in live biomass, forest floor detritus, and potentially soil C, transferring it directly to the atmosphere. Similarly, forest harvesting, a consumptive disturbance, removes large amounts of biomass; however, in contrast to fire, the forest floor and soil are usually left intact. The C dynamics following disturbance have been intensively studied in southern pine plantations (Gholtz and Fisher 1982; Maier et al. 2004; Noormets et al. 2010; McCarthy et al. 2010; Maier et al. 2012); however, less is known about natural pine ecosystems (Powell et al. 2008) or mixed deciduous forests (Ehman et al. 2002; Curtis et al. 2002).

Changes in NEP following a disturbance can be described in four phases (Figure 8.3). For example, tree harvesting of a southern pine plantation transfers some live residual biomass C to detrital C



**FIGURE 8.3** Conceptual pattern of annual net ecosystem productivity (NEP,  $\text{Mg C ha}^{-1} \text{ year}^{-1}$ ) following a stand disturbance.

pools (phase I). Residual forest floor biomass following harvest of pine plantations can range from 15 to 50 Mg C ha<sup>-1</sup> (Eisenbies et al. 2009; Maier et al. 2012), depending on whether the site was stem-only or whole-tree harvested. The second phase is a period of negative NEP, where R<sub>E</sub> exceeds GPP. The magnitude and duration of this phase is strongly influenced by disturbance intensity, soil characteristics, and NPP of the regenerating stand (Maier et al. 2004). During this phase, increased R<sub>H</sub> from accelerated decomposition of forest floor and soil organic matter dominates R<sub>E</sub>. The post disturbance pulse of R<sub>H</sub> can be quite high (>20 Mg C ha<sup>-1</sup> year<sup>-1</sup>) and is influenced heavily by the type of site preparation (e.g., burning, disking, bedding, weed control) (Gough et al. 2005). The recovery period or the point of time to the transition between negative to positive NEP can take <3 years on highly productive sites (Gholtz and Fisher 1982; Clark et al. 2004) and as long as 15 years on poor sites (Thornton et al. 2002; Sampson et al. 2006). In phase III, NEP transitions to an extended period of rapid C accumulation as tree growth moves into the exponential growth phase and reaches a peak near crown closure as canopy biomass and leaf area stabilize. In young, rapidly growing pine plantations, NEP can reach 5–7.5 Mg C ha<sup>-1</sup> year<sup>-1</sup> (Clark et al. 1999; Hamilton et al. 2002; Maier et al. 2004). During this phase, NEP is strongly linked to NPP and can be manipulated by management. For example, 4 years of fertilization in a loblolly pine plantation growing on a poor site shifted NEP at age 12 from C neutral (nonfertilized; NEP ≈ 0 Mg C ha<sup>-1</sup> year<sup>-1</sup>) to a strong C sink (fertilized: NEP = 6.4 Mg C ha<sup>-1</sup> year<sup>-1</sup>) (Maier et al. 2004) (Figure 8.2). Fertilization doubled GPP, but only increased R<sub>E</sub> by 48%. Increased ecosystem C storage occurred mainly in perennial woody biomass. Phase IV is a period characterized by a gradual decline in NEP (Chapin et al. 2002). The ecophysiological mechanisms responsible for reduced NEP in phase IV are unclear, but are probably a function of reduced NPP following canopy closure (Gower et al. 1996). Age-related decline in NPP has been linked to increased respiration costs as the wood:foliage biomass ratio increases (Hunt et al. 1999; Goulden et al. 2011), decreasing GPP due to nutrient and/or water limitations (Ryan et al. 1997; Hubbard et al. 1999), or some combination of the two. Ryan et al. (2008) concluded that the decline in aboveground NPP was due primarily to reduced GPP and secondarily to a shift in partitioning of GPP to foliage respiration and belowground allocation. Alternatively, reduced NPP could be due to changes in stand structure where competition-induced mortality and lower individual tree resource-use efficiencies reduce stand growth (Binkley et al. 2002; Ryan et al. 2008). Regardless of the mechanisms for declining NEP, it is likely that management designed to increase NPP will result in increased NEP. NEP of intensively managed pine plantations can range between 5 and 7 Mg C ha<sup>-1</sup> year<sup>-1</sup>, much higher than naturally regenerated stands (<2 Mg C ha<sup>-1</sup> year<sup>-1</sup>, Powell et al. 2008). A more complete understanding of the ecophysiological mechanisms controlling NEP and its component processes will aid in the development of strategies for actively managing carbon.

## EX SITU CARBON SEQUESTRATION

Carbon is stored in forest products that are currently in use and those in landfills. The effectiveness of *ex situ* C sequestration is dependent on the fate of the wood fiber; that is, whether it is converted into products where the C is confined from the atmosphere for relatively long or short durations of time (Skog and Nicholson 1998; Skog 2008). The amount of C per unit of wood from southern forests has been estimated to be 270.7 and 317.5 kg m<sup>-3</sup> (Skog and Nicholson 1998) for softwood and hardwood, respectively—the highest values estimated for the United States. The half-life (time until it ends its initial use) of wood used to build single-family homes before 1939 was 78 years and has increased since then (Skog and Nicholson 2008). In contrast, paper has an estimated half-life of only 2.5 years (Skog 2008). Wood products used to be burned in dumps. Now, wood products are disposed of in landfills, where due to low oxygen they decompose slowly at rates ranging from 3% for solid wood up to 38% for office paper, over a 50-year period (Micales and Skog 1997). It should be noted, however, that landfills do produce methane, which is 25 times more effective than CO<sub>2</sub> as a greenhouse gas (Ryan et al. 2010). Lastly, waste wood (i.e., sawdust) is used as a fuel in wood

manufacturing plants, pulp mills, and paper processing plants and so acts as a direct replacement for fossil fuels, making the production of these products more energy efficient. Case Study 3 on the impact of silviculture on C sequestration provides an example of an analysis that takes both C *in situ* and *ex situ* sequestration pools into account.

## **PULSE AND PRESS DISTURBANCES, FOREST PRODUCTIVITY, AND CARBON SEQUESTRATION**

It is likely the effects of climate change will occur as exogenous or endogenous disturbance events of as-yet unknown duration, frequency, or intensity in southern forest stands and landscapes. These disturbance events will act as either “press” or “pulse” disturbances (Chapter 2). Press, or persistent, disturbances include fundamental changes associated with climate change itself, such as regional and localized temperature increases, altered precipitation patterns and increased atmospheric CO<sub>2</sub>. Changes in structure and function associated with press events may not be immediately evident, but manifest over time and include changes in species distribution, concomitant shifts in natural ranges of species, and long-term changes in forest growth and productivity. Unlike press events, pulse events occur as discrete and distinct disturbance events in time and space. Pulse disturbances associated with climate change are expected to increase in frequency and/or intensity, and include an increase in the frequency and intensity of wind- or storm-related events, wildfire, insect and disease outbreaks, and flooding. Although some pulse events have only a short-term effect on structure and function, other events may perturb the system in such a way that long-term forest productivity is altered. These climate-change related pulse and press disturbance events will likely have both positive and negative consequences on the production and sustainability of the Southern United States timber and fiber supply, as well as the C storage potential of affected forest stands.

## **EXAMPLES OF PRESS DISTURBANCES**

### **INCREASED TEMPERATURE AND/OR DECREASED PRECIPITATION ON PRODUCTIVITY**

The control over individual tree, stand-, and forest-level growth and productivity is defined by complex interactions among climate, edaphic conditions, genetics, endogenous and exogenous disturbance patterns, and competition for water, nutrients, light, and other resources. The immediate effects of climate change on tree growth will be caused by changes in regional and localized patterns of precipitation and temperature, which directly affect the phenology (e.g., Vitasse et al. 2009) and physiological processes that control C uptake and assimilation (Kozlowski et al. 1991).

Of the abiotic factors that control photosynthesis (e.g., water, nutrients, light, temperature, CO<sub>2</sub>), it is temperature and water availability that most limit photosynthetic activity (Salisbury and Ross 1992) and, hence, tree growth and productivity. A recent study by Way and Oren (2010) suggests that temperate tree species are currently growing at temperatures below their maximum thresholds, and that the increased temperatures associated with climate change may stimulate tree growth. While this generalization regarding the positive relationship between temperature and tree growth is supported by provenance trials for a variety of temperate tree species (Schmidting 1994; Carter 1996; McLane et al. 2011) and retrospective dendrochronology studies (Friend and Hafley 1989; Pichler and Oberhuber 2007; White et al. 2011), the amount by which growth is increased appears to be dependent upon genetics (McKeand et al. 1997; Sonesson and Eriksson 2000), species (Nedlo et al. 2009), and functional group, with deciduous tree species generally experiencing a greater response to increased temperature than evergreen tree species (Way and Oren 2010).

In addition to possible temperature-related increases in growth due to increased photosynthetic activity (Way and Oren 2010), changes in growing season length and associated changes in phenology have the potential to substantially impact not only tree growth and C assimilation (White et al. 1999; Rötzer et al. 2004), but also the timing of flowering and reproductive success of many species

(Walkovsky 1998; Beaubien and Freeland 2000; Peñuelas and Filella 2001). Despite the inherent temporal and spatial variation in growing season length (White et al. 1999), a correlation between temperature and length of growing season has been documented. In the Eastern United States, for example, an increase in growing season length of five days for every 1°C increase in average annual temperature has been observed (White et al. 1999). This increase in growing season length has been corroborated by other studies across both North America (Schwartz and Reiter 2000) and Europe (Menzel 2000; Chmielewski and Rötzer 2001). In North America, the onset of biological spring was advanced by an average of 5 to 6 days between 1900 and 1997 (Schwartz and Reiter 2000), while in Europe, the average annual growing season has lengthened between 8 (Chmielewski and Rötzer 2001) and 11 days since the 1960s (Menzel 2000). The effect of increased growing season length on tree growth and productivity is less understood than the effect of temperature on growing season length itself. Models that incorporate the timing of bud burst into growth simulations suggest that the timing of bud burst and, consequently, length of growing season, have a positive effect on tree growth (Menzel and Fabian 1999), although the responsiveness of bud burst and leaf flushing to temperature is species specific (Vitasse et al. 2009).

Although increased temperatures are predicted to stimulate tree growth, patterns of precipitation in a changing climate may greatly alter or even negate the theoretical increase in tree growth expected with increased temperatures (Way and Oren 2010). Holding edaphoclimatic factors constant, increased temperatures increase evapotranspiration rates. In forest stands with low soil water-holding capacity in particular, this increase in evapotranspiration and soil moisture deficits eventually leads to decreased C assimilation due to stomatal closure (Kozłowski et al. 1991), altered tree allometry and biomass partitioning (Callaway et al. 1994; McDowell et al. 2006; Landsberg and Sands 2011), and, ultimately, decreased cell expansion and tree growth (Kozłowski and Pallardy 1997; Henderson and Grissino-Mayer 2009; White et al. 2011). Provided that the increase in temperature predicted to occur over the next 100 years is concomitant with a decrease in precipitation, substantial reductions in tree growth and productivity could occur despite the positive effects that increased temperatures may have on tree growth and productivity.

Current models suggest that average annual precipitation in some of the sub-regions in the South may decrease as much as 24% over the next 40 years (McNulty et al. 2010), and at a regional scale the frequency and severity of episodic drought events may increase (Dale et al. 2001). If, as the data suggest, precipitation is a strong determinant of annual rates of tree growth for many tree species in the Southern United States (e.g., Orwig and Abrams 1997; Pan et al. 1997; Henderson and Grissino-Mayer 2009; Speer et al. 2009), tree growth and productivity should be expected to decrease as temperature increases and precipitation, and subsequent soil moisture availability, decrease. However, dendrochronology studies have demonstrated that the temporal distribution of precipitation throughout any given year is as important, if not more important, than total annual precipitation. For example, studies have shown that oak species (*Quercus*) are most prone to reduced growth when soil water balance is reduced early in the current growing season (Tardif et al. 2006; Speer et al. 2009) or when the water balance of a stand is low late in the growing season of the year prior to ring formation (i.e., preconditioning) (Jacobi and Tainter 1988; White et al. 2011). In contrast, loblolly (*Pinus taeda*) and longleaf pine (*Pinus palustris*) do not demonstrate any preconditioning requirements and generally experience a decrease in annual growth when precipitation during the spring and summer months is reduced (Jordon and Lockaby 1990; Henderson and Grisso-Mayer 2009).

Retrospective dendrochronology studies have proven useful in quantifying climate–growth relations. However, the response of tree growth to future climate will vary among and within tree species, making broad-scale generalizations regarding the future impact of climate change on tree growth based on dendrochronology studies difficult. The sensitivity of annual growth to future climate will depend greatly upon species and edaphic conditions as well as local temperature and precipitation patterns, including the amount and temporal distribution of precipitation within and among years. Neither precipitation nor temperature alone determines growth dynamics. Rather,

tree growth is heavily influenced by a given site's water balance (Kozlowski and Pallardy 1997; Littell et al. 2008), which is influenced by temperature, precipitation, and edaphic conditions (e.g., soil moisture-holding capacity). Relatively few dendrochronology studies have quantified the variation in climate–growth relationships across environmental gradients, with most concluding that climate–growth relationships for any given species are not uniform across the landscape (Tardiff and Bergeron 1997; Orwig and Abrams 1997; Case and Peterson 2005; Griesbauer and Green 2010; White et al. 2011). For example, the decrease in growth that an individual tree growing on a low-quality site (e.g., a site with low water-holding capacity) experiences during periods of high evapotranspirative demand (i.e., drought) is generally greater than that of an individual tree of similar size and species found on a highly productive site (e.g., a site with high water-holding capacity) (Orwig and Abrams 1997; Case and Peterson 2005; White et al. 2011), suggesting an individual tree's resistance to reduced water availability is greater on high- versus low-quality sites. However, recovery to predrought growth levels (i.e., resilience to drought) may occur more quickly on the lower quality sites due to an individual's acclimation to local growing conditions (Orwig and Abrams 1997) suggesting greater resilience in sites of low versus high productivity. This differential response of tree growth to a climate across ecological gradients emphasizes the variation in resistance and resilience of species to changing climate across a diverse and complex landscape.

### FOREST SOIL C AND CLIMATE CHANGE

Over 2700 Gt of C are stored in soils globally, more than double the combined amount of C contained in the atmosphere (780 Gt) and stored in biomass (575 Gt) (Lal 2008a). The vast pool size and slow turnover of soil C make it central to global C cycling, but also make it difficult to study or manipulate experimentally (Hungate et al. 2009). Across an array of biomes, land–air temperature is rising and the rate is expected to increase in the coming decades (IPCC 2007a). In the scientific community there is a general consensus that this warming (currently ~0.3 C per decade) is due to greenhouse gas forcing by anthropogenic increases of greenhouse gases (e.g., CO<sub>2</sub>, CH<sub>4</sub>) in the atmosphere and it is very unlikely that this warming could be produced by natural causes (IPCC 2007a). Attention is being focused on the role forests play in sequestering some of the anthropogenic C inputs to the atmosphere in biomass and soils, while conserving existing C stocks through informed resource management. Predicted warming of 3–6°C over the next century (IPCC 2007a) may cause changes in forest productivity, nutrient availability, additions to belowground C pools from autotrophs and accelerated decomposition of soil C. There remains a great deal of uncertainty in forecasting whether future forests will be more productive in a warmer world and store more C in soil or release more C into the atmosphere via higher rates of soil organic matter decomposition accelerating warming, that is, “positive feedback” (Luo 2007). The effect of gradual warming on long-term soil C storage will be determined by the balance of autotrophic inputs and losses of stored C soil. A universal theory on the temperature sensitivity of soil C decomposition is desirable, but the myriad of differences in climate, existing soil C stocks, nutrient availability, vegetation, soil mineralogy, soil microorganisms, and land-use practices add complexity and uncertainty requiring regional assessments.

Conversion of forest and other natural systems to agriculture leads to declines in soil organic C (SOC); in temperate regions it is estimated that 30–50% of SOC is lost in 50–100 years after conversion (Lal 2008b). In much of the Southern United States, soil C and soil fertility have already been depleted through past agricultural practices (Giddens and Garman 1941; Jackson et al. 2005). Conventional wisdom in managed southern forests has been that limiting erosion, maintaining canopy cover, and good silvicultural practices would lead to C accumulation in soil organic matter and eventually in the mineral fraction. Given the level of degradation, it seemed there was little additional soil C that could be lost on upland sites. In wet or seasonally inundated forests, where SOC is protected from oxidation by anoxic conditions and has accumulated for centuries, maintaining hydrology is key to continued C retention. Atmospheric C is fixed via photosynthesis and enters the

soil via decomposition of biomass or other more ephemeral photosynthate products such as root exudates. Relatively labile C in biomass is then retained via humification (breakdown by microbes into more resistant forms), aggregation (formation organo-mineral complexes, especially with clay), and formation of biochar (Lal 2008b). Biochar or black C (BC) formed by incomplete combustion of biomass is resistant to decay and has the potential to persist in soils for thousands of years (Schmidt and Noack 2000). The dynamics of BC in soil are still poorly understood, but BC could be an important sink in fire-adapted ecosystems. Frequent prescribed fires (2- to 5-year return) are being used to restore longleaf pine ecosystems in the Coastal Plain and Piedmont regions. The rate of BC accumulation and the importance of this pool are unknown in these systems, but it is the focus of active research. Incorporation of logging debris directly into the soil after harvest has been explored as a means to accelerate humification and long-term C storage in managed forests (Buford and Stokes 2000). However, 8 years after establishment, a forest slash incorporation study on both mineral and organic soils did not significantly SOC or soil chemistry (Sanchez et al. 2009).

Seasonal variation in field measurements of soil respiration is frequently described as a response to changes in soil temperature and moisture. When moisture is not limiting, the response of soil respiration to soil temperature is exponential (e.g., Palmroth et al. 2005); this widely observed phenomenon likely contributed to speculation that runaway C releases from soil were possible in a warmer climate. To avoid some of the experimental artifacts of laboratory incubations with disturbed soils, a variety of soil-warming experiments were established in forest (Rustad and Fernandez 1998; Melillo et al. 2002), prairie (Luo et al. 2001), and agricultural systems to evaluate the fate of soil C and plant productivity.

The soil warming experiment at the Harvard Forest in Massachusetts was established in 1991, using buried heating cables to create a 5°C elevation in soil temperature in a mature hardwood forest. Soil respiration and soil organic matter decomposition were higher in heated plots for a few years, but the response declined until there was no difference between the treatments (Melillo et al. 2002). Warming increased nitrogen (N) mineralization, and Melillo et al. (2002) speculated that this could stimulate C storage in biomass in N-limited environments. The results of this study call into question projections of large long-term releases of C from soils with limited labile soil C. An earlier study in a low elevation spruce-fir forest showed that soil respiration rates had not acclimated to heated soils after 3 years and were 25–40% higher than controls (Rustad and Fernandez 1998). It is unclear whether the 3-year experiment duration was long enough to deplete labile C reserves. There is evidence that soil C decay acclimates to higher soil temperatures after several years of warming (Luo et al. 2001; Melillo et al. 2002; Bradford et al. 2008), and that decomposition of organic C in mineral soil does not vary with temperature (Giardina and Ryan 2000), which would weaken any potential positive feedback between soil C decay and the climate system. However, the debate is far from settled with others asserting that nonlabile SOC is more sensitive to temperature than labile SOC and the duration of current warming studies is not long enough to detect these changes (Knorr et al. 2005). In this scenario, the positive feedback between SOC decay and climate would be greater than predicted. In the eroded soils of the southeast, soil C stocks are generally low on upland sites and soil respiration and SOC decay would be expected to follow the acclimation scenario described by Melillo et al. (2002) and Lou et al. (2001). However, the effects of warming in forests on nutrient-poor clay soils in the southeastern United States, subtropics, and tropics are understudied. Newly installed warming studies in oak forests in the Piedmont region of Georgia and North Carolina will provide experimental results in the coming years (Machmuller et al. 2011).

Lacking a grand, manipulative warming experiment in the Southern United States to guide our understanding, it is necessary to make inferences from the literature on the effect of forest management and disturbance on soil C. In general, harvesting forests has little or no effect on soil C or N, especially when whole tree harvesting is avoided (Johnson and Curtis 2001). In a meta-analysis by Johnson and Curtis (2001), on average, harvesting conifers led to an increase of soil C and N by approximately 25% and harvesting hardwood stands led to small declines in soil C (–8%), and small increases in soil N. In the course of a rotation, trees accumulate biomass and contribute leaf

litter to the forest floor, the forest floor is shaded and a small quantity of C may be stabilized in the mineral soil. After harvest, the forest floor, no longer shaded, heats to temperatures commensurate to the increase predicted a century from now; for example, soil in a clear cut was +3°C warmer than an adjacent 27-year-old slash pine stand (Castro et al. 2000). Over the course of a rotation, the soil is exposed to sudden warming followed by gradual cooling as trees shade the forest floor and soil C still accumulates, more for conifers, less for hardwoods (Johnson and Curtis 2001).

Protecting existing soil C is important in the Southern United States, but the more relevant question is how management practices and climate change will affect additional C inputs. Worldwide, forests occupy approximately 50% of the land surface area, but tend to be located in temperature- and light-limited areas. In general, productivity of forests will be positively affected by increasing atmospheric CO<sub>2</sub>, temperature, and precipitation, positively affected by N deposition (at least in the short term, where not at toxic levels), and negatively affected by pollutants, for example, O<sub>3</sub> under current climate change predictions (Boisvenue and Running 2006; Hungate et al. 2009). Atmospheric deposition of N varies annually based on weather patterns and anthropogenic emissions; in the Southern United States, it currently is less than one-third of deposition rates in the Midwest or Northeast (National Atmospheric Deposition Program, N deposition maps, nadp.sws.uiuc.edu). Forest productivity and soil C accumulation under various climate change scenarios can only be maintained if there is a minimum level of available N (Hungate et al. 2009), and this is a concern in southern forests where N limitations are common. Based on available information and models, southern forest soil C stocks will likely be retained and slowly accumulate as long as forest productivity is maintained or increases, and forestry practices that enhance productivity and soil fertility are used.

## ELEVATED ATMOSPHERIC CO<sub>2</sub>

Owing primarily to fossil fuel burning and deforestation, atmospheric CO<sub>2</sub> levels have increased nearly 35% since preindustrial times, from ~280 ppm to ~380 ppm (IPCC 2007b). Depending on the growth and emissions scenario used, atmospheric CO<sub>2</sub> may rise as high as 550–850 ppm by 2100 (IPCC 2007b). While CO<sub>2</sub> is the primary driver of anthropogenic climate change, it is also the basis of plant photosynthesis. Given that plant photosynthesis is not saturated at current CO<sub>2</sub> levels, anthropogenic increases in CO<sub>2</sub> will almost certainly lead to higher photosynthesis. However, greater photosynthesis may not translate to significantly greater forest productivity and plant C storage, and gains in productivity may not be sustainable over the long term. There are a number of questions that must be answered in order to assess the likely effect of elevated CO<sub>2</sub> on future forest productivity. These questions include: How much can elevated CO<sub>2</sub> increase forest productivity? How variable is the response across and within forests? How much of the CO<sub>2</sub> enhanced productivity will be allocated to woody biomass versus other tissues such as foliage and fine roots? Finally, will elevated CO<sub>2</sub> actually increase forest productivity in the long term, or will responses be constrained by the availability of other growth resources?

Effects of elevated CO<sub>2</sub> on plant growth have been studied extensively for more than 30 years. These investigations began with greenhouse or potted plant seedling studies, which often demonstrated large responses of photosynthesis, leaf area, and growth. However, the use of young plants in the exponential growth phase made it difficult to separate acceleration of development from direct CO<sub>2</sub> effects (see reviews of Drake and González-Meler 1997; Curtis and Wang 1998). Subsequently, open top chamber experiments allowed for the study of larger vegetation in field conditions. However, it was still difficult to predict the effects of increased CO<sub>2</sub> on entire ecosystems, especially those in intermediate or mature stages where canopy closure has occurred and plants are competing for resources. With the development of free air CO<sub>2</sub> enrichment (FACE) technology, it became possible to study the response of intact forest ecosystems to elevated CO<sub>2</sub> (Hendrey et al. 1999). These experiments (described in detail below) form the basis for our current understanding of the effect of increased CO<sub>2</sub> levels on forest productivity and C storage.

FACE technology allows for elevation of CO<sub>2</sub> concentrations in unenclosed patches of forest, with minimal alteration of microclimate and other stand conditions (Hendrey et al. 1999). Four FACE experiments have been conducted in established or establishing forest stands. These FACE sites contain 6–12 circular plots (22–30 m in diameter), where half of the plots are exposed to “ambient” CO<sub>2</sub> concentrations and half are exposed to “elevated” CO<sub>2</sub> concentrations. In “elevated” plots, pure CO<sub>2</sub> is mixed with ambient air to create air with a CO<sub>2</sub> concentration of ~550 ppm (~200 ppm greater than the current level), which is then released from vertical pipes surrounding the circular plot (Hendrey et al. 1999). “Ambient” CO<sub>2</sub> plots have a similar infrastructure, but receive air with current CO<sub>2</sub> concentrations (~376 ppm across time and sites; Norby et al. 2005). Two FACE experiments are located in southern forests, one at Duke Forest (Chapel Hill, NC) in a planted loblolly pine forest, and the other in a planted sweetgum forest in Oak Ridge, TN. In these forests, the CO<sub>2</sub> elevation experiments were conducted when the forests were 13–27 (Duke FACE) and 10–21 (ORNL FACE) years old. Two other forest FACE experiments were conducted on newly planted forests: a mixed aspen/birch/maple forest in Rhineland, WI (operated for 12 years), and two rotations of poplar species in Viterbo, Italy (6 years total; i.e., the stand grew for 3 years, was coppiced and monitored for an additional 3 years). Detailed descriptions of the experimental setups are provided for Duke FACE (Hendrey et al. 1999), ORNL FACE (Norby et al. 2001), AspenFACE (Karnosky et al. 1999, 2005), and POP-EUROFACE (Miglietta et al. 2001).

The average enhancement of NPP observed in forest FACE experiments (after 2–6 years of CO<sub>2</sub> exposure) was surprisingly consistent at 23% (Norby et al. 2005). In forests with low native LAI, such as the coniferous Duke FACE, the increase in productivity was driven almost entirely by increased light capture due to higher LAI (e.g., ~16% higher LAI at Duke FACE; McCarthy et al. 2007) under elevated CO<sub>2</sub> (Norby et al. 2005; McCarthy et al. 2006). In contrast, for forests with high native LAI (such as the broadleaf ORNL site), elevated CO<sub>2</sub> did not increase LAI and light interception (Gielen et al. 2003; Norby et al. 2003, 2005), and the bulk of the enhancement of productivity resulted from increased photosynthetic efficiency (Norby et al. 2005; McCarthy et al. 2006).

Despite the appearance of a predictable, constant enhancement of NPP with elevated CO<sub>2</sub>, analysis of within-site data reveals a great deal of variability in the growth response of forests to increased CO<sub>2</sub>. Much attention has been focused in the Duke FACE experiment on quantifying the interaction of elevated CO<sub>2</sub> enhancement with other growth resources, particularly nitrogen availability and water (Oren et al. 2001; Finzi et al. 2002; McCarthy et al. 2006, 2010). At Duke FACE, NPP enhancement has been closely correlated with soil N availability. The greater the available N, the larger the NPP enhancement with elevated CO<sub>2</sub> (McCarthy et al. 2010). Conversely, the NPP response to elevated CO<sub>2</sub> decreased with decreasing soil N availability (McCarthy et al. 2010), corresponding with studies in which forests with very low availability of nutrients (e.g., in *Pinus taeda* on sandy soils and *Picea abies* on sandy glacial till), had no detectable response to elevated CO<sub>2</sub> (Oren et al. 2001; Ward et al. 2008). In both of these studies, significant CO<sub>2</sub> responses were observed in fertilized trees, demonstrating that the lack of response was related to low N availability. Additionally, water availability also influences productivity response to elevated CO<sub>2</sub>. Interannual variability in both basal area (Moore et al. 2006) and NPP (Finzi et al. 2006; McCarthy et al. 2010) at Duke FACE were correlated with water availability (assessed as growing season precipitation or growing season precipitation minus potential evapotranspiration; P-PET), with NPP enhancement under elevated CO<sub>2</sub> greatest when P-PET was highest (McCarthy et al. 2010). However, N availability was a stronger driver of variability in CO<sub>2</sub> response than water availability, as NPP under elevated CO<sub>2</sub> was ~130 g C m<sup>-2</sup> greater at the highest versus lowest N availability, and only ~30 g C m<sup>-2</sup> greater at the highest versus lowest P-PET (where both factors spanned the full range of possible values; McCarthy et al. 2010). Furthermore, results from AspenFACE, where half of the ambient and elevated CO<sub>2</sub> plots were also exposed to elevated (1.5 times ambient) ozone (O<sub>3</sub>), suggest that O<sub>3</sub> pollution may completely offset (i.e., negate) CO<sub>2</sub> induced growth enhancements (Karnosky et al. 2003; King et al. 2005). Overall, these findings demonstrate that there can be great spatial

variability in how forest productivity is affected by elevated CO<sub>2</sub>, and that some forests may exhibit no productivity enhancement at all.

To fully understand the impact of elevated CO<sub>2</sub> on future forests it is necessary to understand not only how elevated CO<sub>2</sub> may affect overall stand productivity, but to account for how the additional NPP may be partitioned to different tree biomass components. The implications of elevated CO<sub>2</sub> for timber and fiber production or C storage will be much different if the majority of additional NPP gained under elevated CO<sub>2</sub> concentrations is invested in nonwoody biomass such as foliage and fine roots versus wood biomass. In the extreme case, stimulation of NPP by elevated CO<sub>2</sub> could do little to increase C storage or wood production. Although allocation is frequently poorly understood, even under current environmental conditions (Litton et al. 2007), the traditional view of allocation is that plants allocate their resources (C and nutrients) in order to optimize their gain of further resources (including water, e.g., Thornley 1972; Dewar 1993; McConnaughay and Coleman 1999). Therefore, an optimal allocation strategy for trees growing under elevated atmospheric CO<sub>2</sub> would be to allocate proportionally more C to root formation, in order to more fully exploit soil resources (i.e., increase water and nutrient uptake). The magnitude of this shift should be driven by soil resource availability, with more nutrient- or water-limited systems showing a greater increase in fine root allocation (Palmroth et al. 2006; Litton et al. 2007). In practice, results from forest FACE sites have been mixed regarding whether elevated CO<sub>2</sub> causes shifts in the proportion of C allocated to different plant pools. The sweetgum plantation at ORNL FACE showed a dramatic shift in C partitioning, with up to 80% of the extra CO<sub>2</sub>-enhanced NPP being partitioned to wood during the first 2 years of the experiment, but only 25% to wood after 3 years (Norby et al. 2002, 2004). The remainder of NPP was allocated largely to short-lived, nonwoody biomass, doing little to increase standing biomass (Norby et al. 2004) but ultimately increasing soil C storage (Iversen et al. 2012). To a lesser degree, at the end of the first rotation in the POP-EUROFACE experiment on *Populus* species, root pools were increased relatively more under elevated CO<sub>2</sub> than aboveground woody components (Gielen et al. 2005). However, root-to-shoot ratio was unchanged, and the fraction of NPP allocated to woody aboveground biomass was high, ranging among species from 53% to 67% (Calfapietra et al. 2003; Gielen et al. 2005). Unlike the first rotation in which relative accumulation of biomass in stems and roots did not change, during the second (coppice) rotation of *Populus* species at POP-EUROFACE, elevated CO<sub>2</sub> resulted in greater C accumulation in branches and lesser accumulation stems as compared to ambient CO<sub>2</sub> trees; the ratio of above- and belowground biomass remained the same (Liberloo et al. 2006). On the other hand, being exposed to elevated CO<sub>2</sub> since planting, *Populus tremuloides* exhibited no changes to the fraction of standing biomass in various pools, nor the partitioning of NPP (King et al. 2005). These differing outcomes suggest that elevated CO<sub>2</sub> does not have a uniform effect on biomass allocation, and should be considered together with other site factors, for example, LAI (Palmroth et al. 2006).

In the context of forests managed for timber, an important distinction is whether forests under elevated CO<sub>2</sub> will accumulate more tree biomass in the long term or whether elevated CO<sub>2</sub> will simply accelerate the process of stand development, allowing canopies to close and trees to reach their maximum sizes more quickly (Körner 2006). Results from natural CO<sub>2</sub> springs, showing decline in the enhancement of stem growth with age, suggest that the elevated CO<sub>2</sub> may not increase steady-state stem biomass pools (Hättenschwiler et al. 1997). However, little information is available to address this issue as elevated CO<sub>2</sub> experiments have been shorter than the life of a forest stand. At the Duke FACE site, after 14 years of CO<sub>2</sub>, mortality under ambient and elevated CO<sub>2</sub> was similar (~2.5% for pines and ~1% for understory hardwoods), and analysis of average tree biomass versus stand density does not suggest that elevated CO<sub>2</sub> modified the expected relationship of tree size and density (H. McCarthy, unpublished data). Thus far, there is little evidence that elevated CO<sub>2</sub> drastically increases site carrying capacity. However, more rapid accumulation of biomass could allow for shorter rotation lengths and more rapid timber and fiber production.

Finally, it is necessary to consider whether productivity gains resulting from elevated CO<sub>2</sub> are sustainable over the long term. Elevated CO<sub>2</sub> experiments, by necessity, induce a step-change in CO<sub>2</sub>

concentrations. Thus, there is the possibility that productivity changes observed in such experiments represent only temporary responses, resulting from disequilibrium of C and N pools in the ecosystem. Higher production under elevated CO<sub>2</sub> must be supported by some combination of increased uptake of N and increased efficiency of N use. Given that most forest ecosystems are nitrogen limited, many models and long-term simulations that directly evaluate CO<sub>2</sub> effects (unlike most global scale models, which do not link CO<sub>2</sub> stimulation with N availability) predict that the growth enhancement observed in relatively short (2–15 years) FACE experiments cannot be sustained over the long term (Luo et al. 2004). The leading hypothesis that predicts a decline in CO<sub>2</sub> stimulation is progressive N limitation (PNL). In this scenario, NPP initially increases, litter production increases while litter quality changes, decomposition decreases, and N mineralization and thus availability is reduced, ultimately feeding back to a lower NPP (Luo et al. 2004). In short, much of the ecosystem N gets tied up in plant biomass, and there is reduced N availability for future biomass production. This process is commonly observed in developing stands (Richter et al. 2000), but elevated CO<sub>2</sub> may further accelerate this process. Many studies have examined N cycling in forest FACE sites, in order to look for evidence of the onset of PNL, and to answer questions regarding how forests are able to acquire the extra N necessary to support CO<sub>2</sub> enhanced growth. Across the four forest FACE sites, three sites have been found to support increased NPP through increases in N uptake, despite these forests being considered N limited (Finzi et al. 2007). Only the N unlimited (due to previous agricultural land use) POP-EUROFACE demonstrated an increase in nutrient use efficiency. Thus far, there have been conflicting findings regarding PNL in different forest types. At the Duke FACE, initial results from the FACE prototype plot (a plot established prior to the main experiment to test the FACE approach) showed a loss of NPP stimulation after 3 years of elevated CO<sub>2</sub> (Oren et al. 2001). This was attributed to N limitation, as fertilized plots showed no such reduction in CO<sub>2</sub> enhancement (Oren et al. 2001). However, over the longer term in the Duke FACE experiment, the average NPP enhancement has been maintained with no decline over time (McCarthy et al. 2010; H. McCarthy, unpublished data). The longevity of this NPP stimulation has been attributed to greater allocation of C belowground (to roots, mycorrhizal symbionts, and labile C exudates), which in turn makes N more available and accessible (Drake et al. 2011). In contrast, PNL has clearly been observed in the sweetgum forest at ORNL FACE (Norby et al. 2010). Six years into the experiment, NPP (of both ambient and elevated plots) began declining, and NPP in elevated CO<sub>2</sub> plots was no longer significantly greater than NPP in ambient plots (Norby et al. 2010). This loss of CO<sub>2</sub> enhancement was correlated with declining N availability, where N availability declined more rapidly with elevated CO<sub>2</sub> (Norby et al. 2010).

Based on current knowledge, elevated CO<sub>2</sub> may increase short-term productivity and C storage in southern forests. However, the response is likely to be quite variable, depending on the availability of N and water in different regions and sites, with some forests exhibiting little or no CO<sub>2</sub> response due to very low resource availability. Furthermore, increases in productivity may not translate into increases in wood production or C storage, as some forests may allocate much of their extra CO<sub>2</sub>-induced biomass into nonwoody biomass. Finally, there is evidence to suggest that productivity gains may not be sustainable in the long term, and that NPP under future elevated CO<sub>2</sub> concentrations may not be much greater than current levels, due to the inability of forests to acquire the N necessary to support increased growth.

## EXAMPLE OF A PULSE DISTURBANCE

### HURRICANES

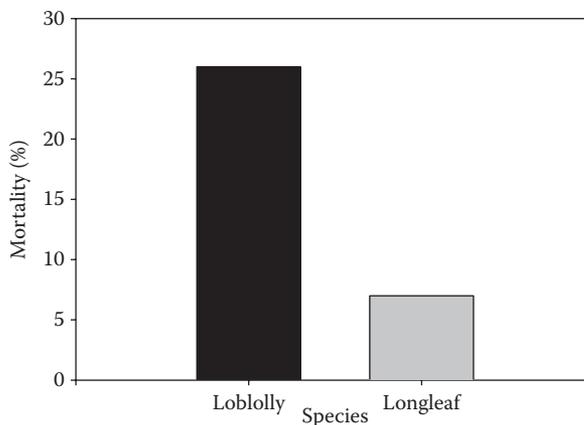
Hurricanes (i.e., tropical cyclones with sustained winds  $\geq 119$  km/h<sup>-1</sup>) can cause substantial economic damage to forests. In 2005, Hurricane Katrina resulted in massive damage to forests along the Louisiana and Mississippi gulf coasts (Chambers et al. 2007; Kupfer et al. 2007; Stanturf et al. 2007). McNulty (2002) estimated that a single hurricane can obviate the equivalent of 10% of the

annual C sequestered in the United States. Owing to its size, intensity and trajectory, Hurricane Katrina may have had 6–14 times that impact (Chambers et al. 2007). In 2005, winds from Hurricane Katrina damaged 22 million m<sup>3</sup> of timber estimated to be valued between \$1.4 and \$2.4 billion. Impacts are not limited to loss of wood volume and quality; ecosystem services provided by these forests can also be impaired. Subsequent decomposition of dead biomass has been estimated to be reducing C sequestration capacity of Gulf Coast forests by an amount equal to the total U.S. net annual forest C sink (Chambers et al. 2007).

Although not necessarily linked to climate change, hurricane activity has increased since the mid-1990s and this higher activity has been projected to last for the next 10–40 years (Goldenberg et al. 2001). Four main factors are related to the extent and intensity of wind damage on forests: climate, soils, topography, and stand conditions (Wilson 2004). Hurricanes obviously represent an extreme climatic event. Sites with soil conditions that restrict root growth and depth are consistently more prone to uprooting. Variation in windthrow along topographical gradients is more complicated and often confused with species and soil variation. There are many stand attributes that help determine the susceptibility of stands to windthrow. These include height-to-diameter ratios, height, spacing, recent thinning, and impacts of previous disturbance on creating exposed edges that contain trees more vulnerable to windthrow. Species composition may also impact the degree of damage from hurricanes and represents a stand attribute that can be manipulated by forest managers.

Some evidence suggests that longleaf pine might be more tolerant of high winds than either slash pine or loblolly pine. In a study of the Hobcaw Forest in coastal South Carolina after Hurricane Hugo, Gresham et al. (1991) reported that longleaf pine suffered less damage than loblolly pine. It was noted that species native to the coastal plain are possibly better adapted to the disturbance regimes found there; for example, longleaf pine, baldcypress (*Taxodium distichum*), and live oak (*Quercus virginiana*) suffered less damage than forest species with broader distribution ranges.

Johnsen et al. (2009) studied wind damage of these pine species in a common garden experiment in southeast Mississippi following Hurricane Katrina, which directly impacted the stand in August 2005. The experiment, a factorial arrangement of silvicultural treatments established in 1960, included one hundred twenty 100-tree plots covering about 22 ha. Following the hurricane, diameter at breast height (DBH) was measured on all trees and each tree was rated with respect to mortality from wind damage. Longleaf pine suffered lower mortality (7%) than loblolly pine (26%) (Figure 8.4). Longleaf pine lost significantly fewer stems ha<sup>-1</sup> and less basal area than loblolly pine. Differences in mortality between species were not a function of mean plot tree height or plot density.



**FIGURE 8.4** Mortality of loblolly and longleaf pine from a Mississippi experiment hit directly from Hurricane Katrina in 2005. Species effects were statistically significant at  $\alpha = 0.05$ . (Adapted from Johnsen, K.H. et al. 2009. *Southern Journal of Applied Forestry* 33:178–181.)

It is not possible to unequivocally state that longleaf pine has adapted to be more tolerant to wind damage than loblolly pine. Wind damage increases with tree size, but the frequency and severity varies with species, site, wind parameters, and stand characteristics (i.e., canopy evenness and age distribution), making blanket statements regarding species fitness an oversimplification (Gresham et al. 1991; Peterson 2007). As per the risk map shown in Stanturf et al. (2007), the southern coastal plain of the United States (the center of the historical range of longleaf pine) is highly prone to hurricane events. Intense hurricanes occur two out of every 3 years across the Eastern United States (McNulty 2002). Similar to historical natural fire regimes, the selection pressure of frequent high-velocity winds appears to have been high. This, and the results above, support the supposition that longleaf pine has evolved to have higher resistance to wind damage than loblolly pine.

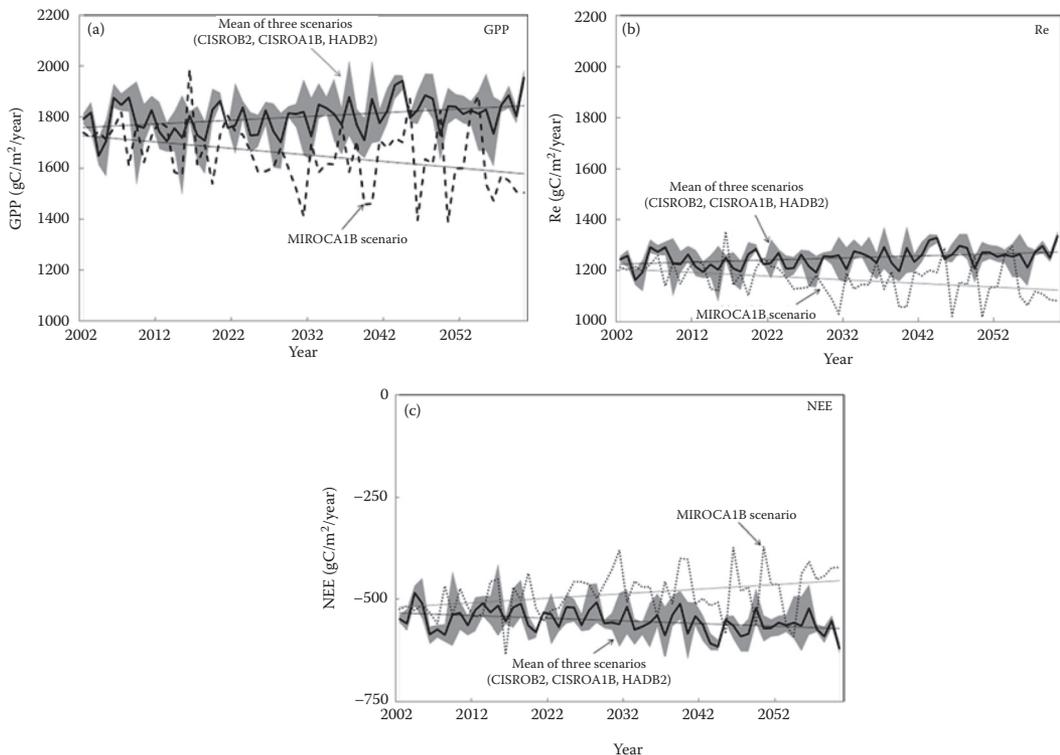
## REGIONAL ESTIMATES OF FOREST PRODUCTIVITY AND RESPONSES TO CLIMATE CHANGE

The effect of and responses to future climate change on forest productivity at regional scales can only be estimated via forest productivity models. We used the WaSSI-C model (Sun et al. 2011a) to examine the annual gross ecosystem productivity (GEP), ecosystem respiration ( $R_E$ ), and net ecosystem exchange (NEE) of the five ecoregions of the Southern United States. The WaSSI-C model is a water-centric model that simulates the monthly water balances (precipitation, evapotranspiration [ET], water yield, and soil moisture storage) of 2103 basins (or called 8-digit Hydrologic Unit Code; HUC) across the lower 48 states including the 674 watersheds in the 13 southern states. The average size of the basin with mixed land use and land cover is 3662 km<sup>2</sup>, ranging from 184 km<sup>2</sup> to 20,515 km<sup>2</sup>. The key component of watershed water balance is ET, which accounts for as much as 85% of precipitation input at the annual scale in the Southern United States. Monthly ET of a watershed with mixed land-covers is modeled as a function of potential ET (calculated based on a temperature-driven model), precipitation, and LAI, that is,  $ET = f(PET, P, LAI)$  (Sun et al., 2011b). Ecosystem C fluxes of each land cover within a watershed were modeled as a series of linear functions of ET. Mathematically,  $GEP = a * ET$ ,  $R_E = b + c * GEP$ , and  $NEE = R_E - GEP$ . The parameters, a, b, c, are derived from global eddy flux data (Sun et al. 2001a). An ecosystem with a negative NEE is considered a C sink; otherwise, the ecosystem is a C source. At the watershed scale, all modeled variables (ET, GEP,  $R_E$ , NEE) were calculated by land-cover type, and then averaged by land-cover proportion to generate the mean for each watershed. The WaSSI-C model has been validated with both USGS runoff and MODIS ET and GEP products at the continental scale (Sun et al. 2011a). Four cornerstone climate change scenarios (CSIROA1B, CSIROB2, HADB2, and MIROCA1B) were applied to simulate monthly and annual ET, GEP, and NEE. Historic LAI data were derived from MODIS remote sensing products for the period of 2000–2006. Given the complexity and uncertainty of climate–vegetation–biogeochemical interactions at the regional scale, we focused on impacts of the physical climate on water availability and its influences on C balances. We assumed LAI for each land-cover type does not change over the simulation time frame (2002–2060) and CO<sub>2</sub> fertilization effects on water use efficiency and plant growth are not considered (see Elevate Atmospheric CO<sub>2</sub> section above for rationale). WaSSI-C modeling results for the first three future climate scenarios were averaged to represent the mean response to climate change among these scenarios. We did not average the climate, but rather averaged the simulated response variables. It appears that the MIROCB2 scenario represents extreme hot and dry conditions (precipitation decreased 8% and PET increased 20% nationally) and differs significantly from other three scenarios. Thus, for this analysis, we amalgamated the first three scenarios to represent the “mean” conditions, and used the MIROCB2 as the “worst-case” scenario. We define “2060 Results” as the mean between 2051 and 2060, while the baseline was considered the years 2002–2010. The “change” was estimated as the difference between results projected for 2060 and 2010.

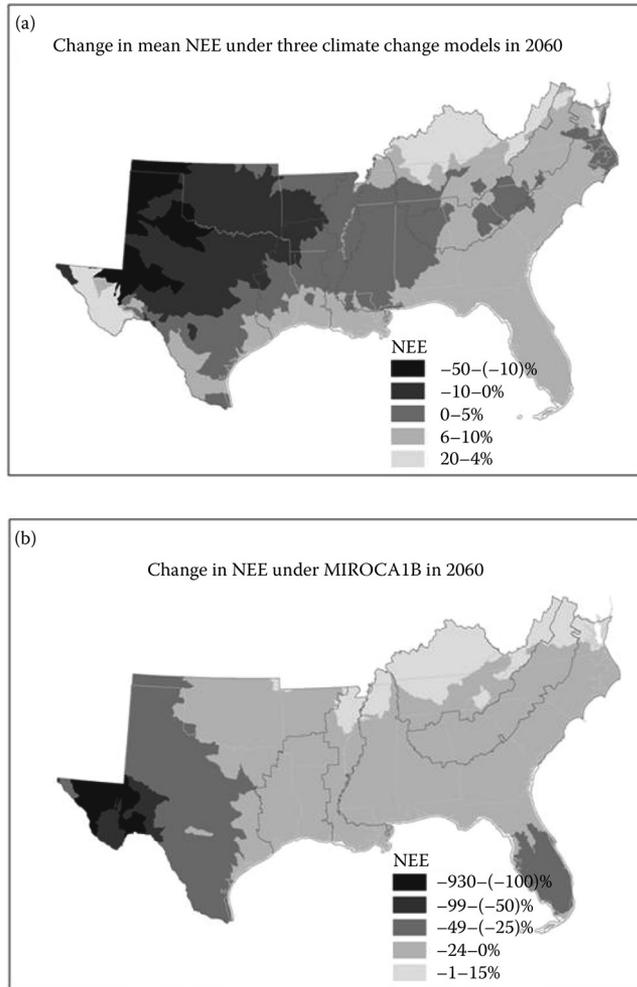
As a whole, under the amalgamated ensemble “mean” condition, the Southern United States was projected to increase in ecosystem productivity and C sequestration strength due to a warming

climate that will result in increased potential ET (PET), actual water loss (ET), photosynthesis (GEP), and ecosystem respiration ( $R_E$ ) (Figure 8.5). At the southern regional scale, GEP,  $R_E$ , and NEE were projected to increase by 2.6%, 2.1%, and 3.8% respectively, in 2060, however, there was large spatial variability. Dry regions were expected to show a decrease in ecosystem productivity, presumably due to water stress caused by reduced precipitation and increased PET. The causal effects between productivity and water stress are more pronounced under the MIROCA1B scenario. Under this scenario, because of reduced precipitation and elevated PET, large water stress was predicted to occur. Ecosystem productivity (GEP) and C sequestration (NEE) were projected to decrease greatly under the MIROCA1B scenario due to reduced ET and water stress (Figure 8.6). At the regional scale, GEP,  $R_E$ , and NEE were expected to decrease by 6.7%, 5.4%, and 9.8% in 2060. Similarly, the dry arid Mid-South had the highest negative impacts from this climate change scenario (Figures 8.6 and 8.7).

Simulation results show that the Mid-South had the lowest GEP and NEE due to low water availability under a dry climate and the impacts on this subregion were most pronounced under all climate change scenarios (Figures 8.6 and 8.7). The Coastal Plain and Mississippi Alluvial Valley subregions had the highest GEP and NEE due to a warm and wet climate (Figures 8.6 and 8.7). These scenario modeling exercises demonstrate that future regional forest productivity patterns are generally controlled by the combination of changes in air temperature and precipitation. Precipitation is a key factor that should be examined carefully, especially in the traditionally climate transition zones where current precipitation levels just barely support forests. Our analyses also indicate that because the existing climate models do not agree on the future trends of precipitation, the projections of future change in forest productivity have large uncertainty.



**FIGURE 8.5** Predicted mean annual (a) Gross Ecosystem Productivity (GEP), (b) Ecosystem Respiration ( $R_E$ ), and (c) Net Ecosystem Exchange (NEE) across 674 watersheds in the Southern United States under four climate change scenarios.



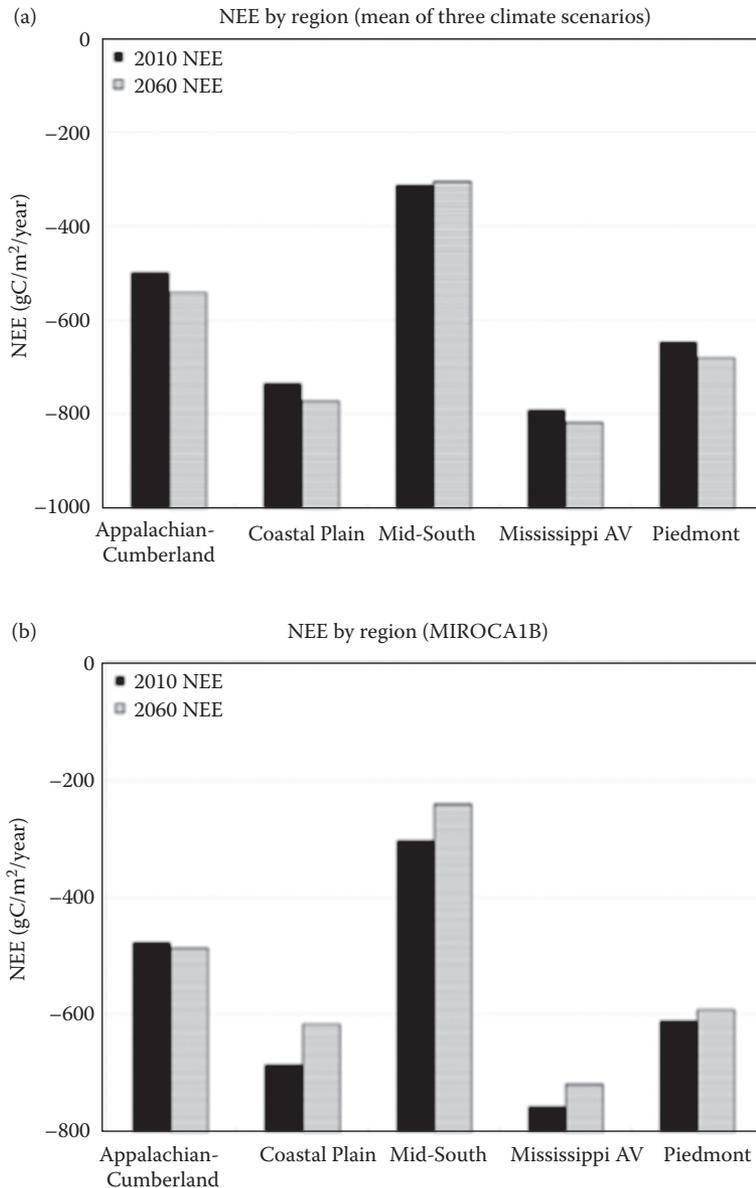
**FIGURE 8.6** Predicted mean response of annual net ecosystem exchange (NEE) across 674 watersheds in the Southern United States to: (a) three climate change scenarios (CSIROA1B, CSIROB2, HADB2), and (b) MIROCA1B climate change scenario.

## CASE STUDIES

The following case studies are presented to provide tangible examples of ways forest management can impact forest productivity and C sequestration. The first case study is a retrospective analysis of growth and aboveground *in situ* C sequestration comparing loblolly and longleaf pine planted in Mississippi in 1961. Using a combination of long-term plot data and modeling, the second case study examines the impact of thinning and future climate scenarios on aboveground productivity of yellow poplar. The third case study presents a modeling analysis on the impacts of rotation length, thinning, and planting density on *in situ* and *ex situ* C sequestration in loblolly and slash pine plantations.

### CASE STUDY 1—RELATIVE GROWTH, STOCKING, AND CARBON ACCUMULATION OF LOBLOLLY AND LONGLEAF PINES IN THE MISSISSIPPI GULF COAST

In the United States Gulf Coast region, climate change is anticipated to alter not only temperature and precipitation, but also the frequency and severity of tropical storms and hurricanes. Comprehensive



**FIGURE 8.7** Summary of change in net ecosystem exchange (NEE by) subregion (a) under three climate scenarios (CSIROA1B, CSIROB2, HADB2), and (b) under MIROCA1B.

management of forests for wood products, fuel load reduction, C sequestration, and ability to withstand frequent disturbance is highly desirable, but the outcomes of prior management decisions are difficult to assess decades after they are made. Long-term studies on experimental forests can be quite valuable to objectively assess the impacts of management as forests mature. In this case history of the Species by Management Intensity study at the Harrison Experimental Forest (Smith and Schmidting 1970), we highlight the impact of pine species, site preparation, and fertilization on aboveground productivity. While this study was originally intended to examine pine species' response to management and the heritability and expression of desirable wood properties in 1960, by inventorying the experiment over the years it has proven valuable for assessing species-specific

growth and yield, susceptibility to wind damage, and the cumulative impact of frequent storms on stocking and C accumulation. In this case history, loblolly pine (*Pinus taeda*), a fast growing, easy to propagate pine that is very responsive to intensive forest management is compared with longleaf pine (*Pinus palustris*), which was historically more dominant in the Gulf Coast, but was passed over due to difficulty with propagation and slow early growth in favor of other pine species. This comparison will show the importance of selecting the most appropriate genetic material (in this case, species) suited to the current and future site conditions and microclimates to meet management goals.

## Materials and Methods

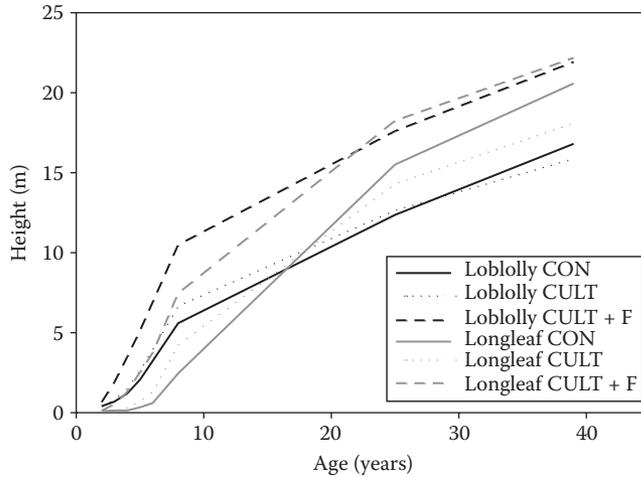
*Site and experimental design:* The site (30.65N, 89.04W, elevation 50 m) is located 32 km north of Gulfport, Mississippi. The soils are variable, but best described by the Poarch series (coarse-loamy, siliceous, semiactive, thermic Plinthic Paleudults) and the Saucier-Susquehanna complex with well-drained upland, fine sandy loams and slopes from 1% to 4%. This case history uses a subset of the treatments and species from an experimental planting established in 1960 after a second rotation longleaf pine stand was clear cut, as originally described by Schmidting (1973). Using a split-plot design with four blocks, the whole plots were randomly assigned one of two species (loblolly and longleaf pine) and the split plots were cultural treatments of varying intensity. The three split plots were: (1) no cultivation or fertilization (CON); (2) cultivated with no fertilization (CULT); (3) cultivated with a single application of 112, 224, or 448 kg ha<sup>-1</sup> of NPK fertilizer (10-5-5) (CULT + F). Cultivated plots were cleared of all stumps and slash, plowed, and then disked prior to planting. They were then disked three times each season for 3 years to reduce woody competition and then mowed in years 4 and 5. Fertilizer was applied 1 year after planting.

*Tree growth and aboveground C determination:* In February and March 1961, 100 1-year-old bare root seedlings were bar-planted with 3.05 m spacing in each square measurement plot enclosed by an additional two rows of buffer trees. Growth through 25 years (fall 1984) has been reported by Smith and Schmidting (1970), Schmidting (1973), and Schmidting (1987). This case history uses height data collected at ages 2, 3, 4, 5, 8, 9, 25, and 39, and diameter at breast height at ages 8, 9, 25, and 39. Longleaf pine aboveground biomass (AGB) was determined by fitting an equation that estimates ln(AGB) as a function of ln(D<sup>2</sup>) using data from Garbett (1977). Loblolly pine AGB was estimated from equations reported by Jokela and Martin (2000). Aboveground biomass was converted to aboveground C by multiplying by 0.5.

## Results

*Height and diameter:* In both loblolly and longleaf pines, mean tree height increased with intensity of cultural treatments in the first 15 years. Trees were smaller in the CON and increased from CULT to CULT + F (Figure 8.8). By age 39, trees of both species were still taller in the CULT + F treatment, but mean tree height in CON was larger than in CULT. Despite earlier gains in height by loblolly pine, by age 25, longleaf pine was taller than loblolly pine within the same cultural treatment (Figure 8.8). At age 9, DBH increased with cultural intensity and loblolly was greater than longleaf (Figure 8.9). Through age 39, loblolly in the highest intensity of management (CULT + F) maintained the largest diameter, closely followed by longleaf pine. By this age, diameters of unfertilized loblolly and cultivated longleaf were 27% lower than fertilized loblolly and longleaf. Diameters of longleaf in CON were only 9% lower than fertilized treatments by the end of the study, though it is likely that early mortality led to fewer larger trees, confounding direct comparisons.

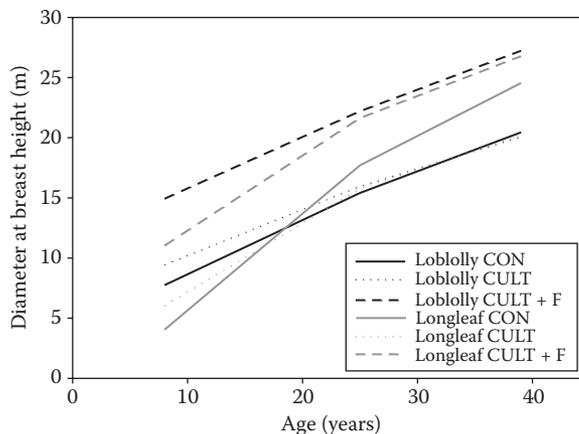
*Stocking and aboveground carbon accumulation:* While mean tree statistics describe the individual trees in a plot, differences in survivorship between species and treatments led to differences in stocking, affecting mean tree parameters. Throughout the experiment loblolly pine stocking was very consistent and there were no substantive differences between treatments (Figure 8.10). Longleaf pine suffered mortality in the first several years of the experiment. Survivorship in CON was particularly poor and stocking remained much lower than the other treatments for the rest of the experiment (Figure 8.10). Without fire, cultivation, or herbicide for weed control, longleaf pine can



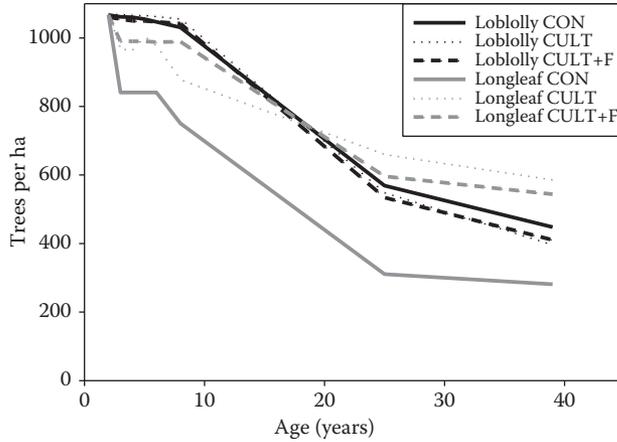
**FIGURE 8.8** Height of loblolly and longleaf pines at the Species by Management Intensity study at the Harrison Experimental Forest, located in Saucier, Mississippi, through age 39. Abbreviations in legend are as in text.

be quickly outcompeted. By age 25, longleaf pine stocking in CULT and CULT + F was higher than any of the loblolly treatments. Total aboveground biomass converted to  $\text{Mg C ha}^{-1}$  is a useful metric that combines individual tree parameters with stocking data, giving the sum total C accumulated in an experimental plot. Loblolly pine in CULT + F accumulated 36% less C than longleaf CULT + F, while loblolly CULT was 45% less than longleaf CULT; there was virtually no difference between the species in CON (Figure 8.11). Clearly, if stocking can be maintained through management activities in early years, longleaf pine will accumulate more C than loblolly pine on this Mississippi Gulf Coast site, albeit given the genetic sources used in this study.

*Effects of management intensity and species on soil C and N:* In addition, despite large differences in aboveground C, soil C and N were not influenced by species selection at age 49 (Butnor et al. 2011). The treatments did result in long-term differences in soil C and N. In the upper 10-cm of soil, CULT had 24% less C than the average C content of CON and CULT + F, which were not



**FIGURE 8.9** Diameter at breast height of loblolly and longleaf pines at the Species by Management Intensity study located at the Harrison Experimental Forest, located in Saucier, Mississippi, through age 39. Abbreviations in legend are as in text.

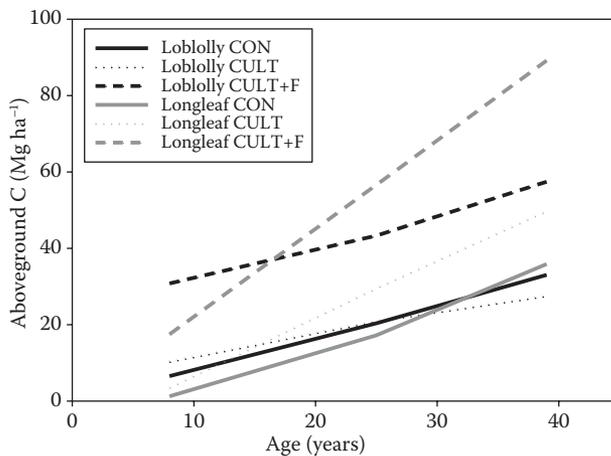


**FIGURE 8.10** Stocking of loblolly and longleaf pines at the Species by Management Intensity study located at the Harrison Experimental Forest, located in Saucier, Mississippi, through age 39. Abbreviations in legend are as in text.

significantly different; in the 10- to 20-cm depth interval, that difference grew to 32%. Soil N content in CULT was 38% lower than the average C content of CON and CULT + F, which were not significantly different (Butnor et al. 2011).

**Discussion**

This case history comparing loblolly and longleaf pine demonstrates the importance of selecting appropriate genetic material (in this case species) suited to the current and future site conditions and microclimates to meet management goals. On productive soils or with the addition of fertilizer, loblolly pine will have greater productivity than longleaf pine in most of its range. This is especially true if longleaf pine seedling survival is poor. Over the long term, in disturbance prone environments, the advantage shifts to longleaf pine. Being more resistant to wind-related mortality loblolly pine (Johnsen et al. 2009), it is better able to survive in regions with frequent storm intervals. Given the rapid height growth of loblolly early in the rotation (Figure 8.8), there has been widespread



**FIGURE 8.11** Cumulative aboveground C accumulation in loblolly and longleaf pines through age 39. Abbreviations in legend are as in text.

planting of loblolly in the range historically dominated by longleaf. If storm frequency and intensity increase as predicted in coming decades, deciding to plant loblolly pine stands will entail a level of risk. The Harrison Experimental Forest is impacted by damaging winds every 5–10 years, resulting in gradual declines in stocking of planted pines. The cumulative effects are more pronounced in loblolly than longleaf as stands age. There is always the risk of catastrophic losses from direct impacts of hurricanes, but if storm frequency increases in regions where they did not occur before, growth and yield are likely to suffer. Industrial forestland owners and managers may be in a position to potentially mitigate these risks by using intensive management and short rotations, but managers of public-owned land and small private landowners may find that deploying better suited, more resilient genetic material to be most feasible.

The results of this experiment show that in the conditions found at the Harrison Experimental forest, longleaf pine will accumulate more aboveground C than loblolly if early survival is maintained. Much less is known about the relative differences in belowground C storage and the recalcitrance of roots and decaying materials between these species. After 49 years, there was no significant effect of pine species on soil C, but soil C accumulates slowly and it may require multiple rotations for the effects to become evident, if at all. Stump removal caused reductions in soil C and soil N in CULT, which was mitigated by a one-time fertilizer application in CULT + F. Thus, the importance of residual stumps for soil C sequestration and site fertility was pronounced.

## **CASE STUDY 2—INFLUENCE OF PAST AND FUTURE CLIMATE ON THE GROWTH OF YELLOW POPLAR (*LIRIODENDRON TULIPIFERA* L.) IN THE SOUTHERN APPALACHIAN MOUNTAINS**

Yellow poplar (*Liriodendron tulipifera* L.) is a mesophytic species that grows in highly productive forest stands in the eastern portion of the Central Hardwood Region. According to FIA estimates (Thompson 1998; Schweitzer 2000; Brown 2004; Rose 2007), yellow poplar constitutes ~15% of the total live-tree volume in the southern Appalachians. This shade-intolerant tree species is an aggressive competitor on mid- to high-quality sites throughout the region. Although a highly productive timber species, yellow poplar is sensitive to drought, with reduced growth during periods of reduced precipitation (Beck 1985; Kolb 1990; Elliott and Swank 1994; Orwig and Abrams 1997; Klos et al. 2009).

Although the role of climate in regulating tree growth is well established, relatively few studies have addressed the role active management may have in altering climate–growth relationships. In Finland, Mäkinen et al. (2002) observed that the growth of Norway spruce was negatively correlated with temperature and positively correlated with precipitation, and the variation in growth was similar in stands thinned to different densities. In contrast, Misson et al. (2003) found the growth of Norway spruce in heavily thinned stands was less affected by drought than trees in lightly thinned or unthinned stands.

In the southern Appalachians, minor changes in topography have a significant influence on climatic patterns, species composition, and site quality, all of which influence productivity. Yellow poplar is one of the most productive forest types in the southeastern United States. Information regarding climate–growth relationships under past climatic conditions as well as under varying management histories and site qualities may be used to inform practitioners of management activities that may increase resilience of a dominant tree species to climate change. Results presented here will help identify stands most susceptible to climate-related reductions in productivity, and provide guidelines on management activities that best offset those negative effects.

### **Methods**

*Study area:* This study uses data collected as part of a long-term study examining the growth and yield of yellow poplar throughout the southern Appalachians. Between 1961 and 1964, 1410.1 ha growth and yield plots were established in yellow poplar stands throughout the Blue Ridge and northern Ridge and Valley Provinces of the southern Appalachian Mountains. Plots were located in northern Georgia, western North Carolina, and southern Virginia. All plots were established in

**TABLE 8.1**  
**Pre-Thinning (1961) and Post-Thinning (1966) Stand Attributes ( $n = 104$ )**

Stand Attribute	Mean	Min.	Max.	Standard Deviation
Yellow poplar SI (m)	32.3	22.9	40.2	3.3
Age				
Pre-thinning	49	18	76	15
Post-thinning	54	23	81	15
Trees/ha				
Pre-thinning	245	80	700	134
Post-thinning	194	70	510	101
Dq (cm)				
Pre-thinning	34.4	15.9	51.4	8.3
Post-thinning	38.3	17.3	54.5	8.2
Basal area (m <sup>2</sup> /ha)				
Pre-thinning	19.6	8.9	35.0	6.8
Post-thinning	19.9	9.2	34.7	6.7

even-aged stands dominated by yellow poplar across a range of initial ages, SI, and structures (Table 8.1).

*Data collection:* At the time of plot establishment and prior to the thinning treatment, all trees >11.4 cm diameter at breast height (DBH; 1.37 m above ground line) within each plot were tagged. For all trees, species, DBH, and total height were recorded. Following the initial inventory, plots received a low thinning to a residual basal area (BA; m<sup>2</sup>/ha) at least one 6 m<sup>2</sup>/ha BA class less than the pre-thinning BA. After the second inventory cycle (1966–1969), 128 of the 141 permanent plots were thinned for a second time to the originally assigned residual BA. No subsequent thinnings followed. Re-measurement of all plots occurred every 5 years through 1991. During each inventory cycle, the status of all tagged trees was assessed and DBH was recorded on all live trees.

*Statistical analysis:* Statistical analysis was performed using only those plots located in the Blue Ridge Physiographic Province in Georgia and North Carolina. An additional 10 plots were removed from analysis due to harvesting and incomplete data, making 104 of the original 141 plots available for use in data analysis. Regression analysis was used to model the effects of age, stand structure (i.e., BA), SI, and climate on BA at 5-year intervals. Because weather data obtained from stationary weather stations do not correspond to precipitation observed at a given plot (Beck 1985), average

**TABLE 8.2**  
**Five-Year Average Growing Season Precipitation (May through September), Maximum Growing Season Temperature, June and July Precipitation, and the 5-Year Average Ratio of June to July Precipitation ( $Q$ ) Used in the Model Building Process**

Variable	Mean	Min.	Max.	Standard Deviation
Precipitation (mm)	725	446	1025	141
Maximum temperature (°C)	25.9	23.1	27.8	0.9
$Q$	1.16	0.74	2.29	0.26

*Source:* Daly, C. et al. 2004. Up-to-date monthly climate maps for the conterminous United States. Proceedings of 14th AMS Conference on Applied Climatology, 84th AMS Annual Meeting Combined Preprints. American Meteorological Society: Seattle, WA, January 13–16, 2004, Paper P5.1, <http://ams.confex.com/ams/pdfpapers/71444.pdf>.

*Note:* Climate data were obtained for each of the 104 plots using the climate interpolation program, PRISM.

monthly climate data between 1961 and 1999 for each plot (Table 8.2) were obtained using the PRISM climate model, which is specifically designed to interpolate climate data for mountainous terrain (Daly et al. 1994). Climate variables tested included average 5-year total growing season precipitation (mm), with growing season defined as the months of May through September, average 5-year maximum growing season temperature ( $^{\circ}\text{C}$ ), and the 5-year average ratio of June to July precipitation ( $q$ ). This  $q$  ratio was investigated because work by Beck (1985) suggests that the amount of precipitation received during the month of July in addition to the ratio of precipitation in June versus July, as opposed to total growing season precipitation, best explains annual diameter growth in yellow poplar stands.

Models were fitted using stand and site attributes SI, age, and BA along with the climate variables. Interactions between climate variables and stand and site attributes were also examined. The best model was chosen based on the model resulting in the lowest Akaike's information criterion (AIC) values. The covariance structure used to account for the autocorrelation among measurements that occurred on each independent plot was modeled using first-order autoregressive structure that allowed for heterogeneous variances. Basal area was  $\log_e$ -transformed to achieve normality and homoskedasticity. The model was fitted using Proc Mixed in SAS/STAT<sup>®</sup> software, version 9.3 (SAS Institute, Inc.) with a significance level of  $\alpha = 0.05$ .

*Effects of future climate on the productivity of yellow poplar:* To compare the productivity in terms of BA development of the 104 stands located on the Blue Ridge Providence of the Appalachian Ecoregion, the climate-sensitive BA model developed above was used to project BA at two different time intervals. The first interval was the time period between 1960 and 2010, and the second interval was for the time period between 2010 and 2060. In the first interval, PRISM data provided the requisite climate data, while stand attributes (e.g., age, SI, BA) observed following the second thinning (Table 8.1) were used as starting points in the modeling process. In the second projection interval, future climate data downscaled to the county level provided the requisite climate data (Coulson et al. 2010), while again, stand attributes observed after the second thinning were used as starting points in the modeling process. Future climate data were obtained from two IPCC climate scenarios coupled with two GCM combinations: (1) A1B/MIROC scenario, which predicts extreme future climatic conditions, and, therefore, represents the upper bound of the productivity predictions; and (2) B2/HAD scenario, which predicts a more moderated climate, and, therefore, represents the lower bound of the productivity predictions. Basal area for each plot under the 1960–2010 and two future climatic conditions were compared and contrasted at each time step (i.e., 5-year interval).

## Results

The climate data obtained from PRISM for the 1961–1999 time period coinciding with the periodic inventories of the 104 plots was variable (Table 8.2). The 5-year average ratio of June to July precipitation ( $q$ ) coinciding with the inventory cycles was the most significant predictor of BA over time relative to the other climate variables examined. When  $q$  was included in the model, the 5-year average rainfall received in July, as suggested by Beck (1985), was not significant and, therefore, not included in the model. The addition of a temperature variable increased AIC, and was therefore not included. The final climate-sensitive BA model for yellow poplar predicted BA as a function of SI, age, BA,  $q$ , along with the interactions between  $q$  and age and  $q$  and BA:

$$\ln BA_2 = b_0 - b_1 (1/SI) + b_2 (1/A_2) + b_3 ((A_1/A_2) \ln BA_1) - b_4 (q) - b_5 (q \times A_2) - b_6 (q \times \ln BA_1)$$

where SI is site index,  $A_1$  is stand age at the previous inventory,  $A_2$  is stand age at the current inventory,  $BA_1$  is stand BA at the previous inventory,  $q$  is the 5-year average ratio of June to July precipitation, and  $b_0, b_1, b_2, b_3, b_4, b_5,$  and  $b_6$  are estimated model parameters (Table 8.3).

Climate under the A1B/MIROC and B2/HAD future scenarios varied substantially from the climate normals used in the baseline scenario (Table 8.4). Although average  $q$  for the baseline and A1B/MIROC scenario were similar, the A1B/MIROC scenario had the greatest variability

**TABLE 8.3**  
**Estimated Model Parameters (Standard Error) and Associated AIC and Approximate R<sup>2</sup> Values for the Final Climate-Sensitive Basal Area (BA) Projection Model**

$b_0$ (intercept)	$b_1$ (1/SI)	$b_2$ (1/A <sub>2</sub> )	$b_3$ ((A <sub>1</sub> /A <sub>2</sub> ) * ln BA <sub>1</sub> )	$b_4^*$ ( $q$ )	$b_5$ ( $q * A_2$ )	$b_6$ ( $q * \ln BA_1$ )	AIC	R <sup>2</sup>
0.2813 (0.0671)	2.6528 (0.7496)	15.0290 (0.4148)	0.9565 (0.0181)	0.0443 (0.0385)	0.0007 (0.0001)	0.0345 (0.0118)	-2386.9	0.98

Note: The  $p$ -values for all parameters were <0.05 except for those noted.

\* Parameter estimate not significant at  $\alpha = 0.05$ .

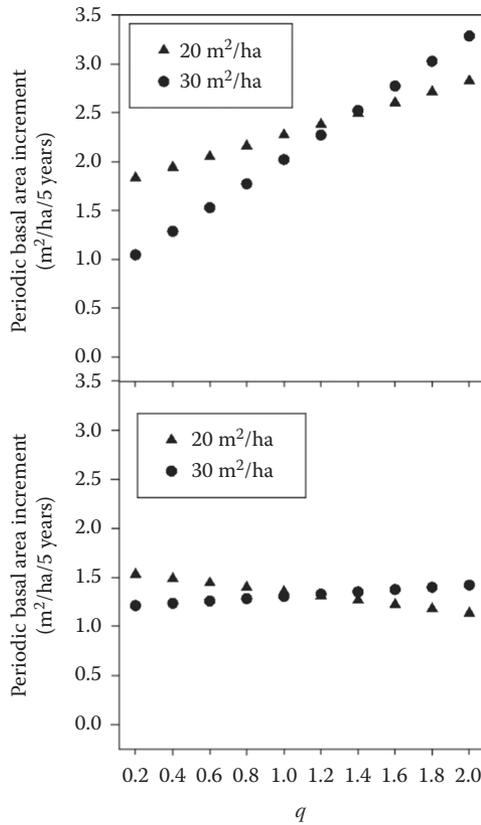
in climate over the 50-year projection period. In contrast, average  $q$  under the B2/HAD scenario was 21% lower than under the baseline, with variability surrounding  $q$  lower than in both the A1B/MIROC and baseline scenarios.

The effects of climate, precipitation in particular, were significant in the final model. However, on average there was little difference in BA of yellow poplar stands over time under the climate scenarios examined in this study. This was likely due to the high degree of variability, as opposed to a strict increase or decrease in precipitation, during the 50-year projection period in both the future and baseline scenarios. At the end of the 50-year projection period (2060), the average difference between predicted BA under the baseline scenario and the future A1B/MIROC scenario was negligible at 0.07 m<sup>2</sup>/ha while the average difference in total BA between the baseline scenario and the future B2/HAD scenario was 1.12 m<sup>2</sup>/ha. The increase in BA predicted to occur in the 104 plots between 2010 and 2060 varied between 29% and 270% for the baseline scenario, 28% and 278% for the A1B/MIROC scenario, and 27% and 253% for the B2/HAD scenario.

**TABLE 8.4**  
**Five-Year Average Growing Season Precipitation (May through September), 5-Year Average Maximum Growing Season Temperature, and Average 5-Year Ratio of June to July Precipitation ( $Q$ ) Corresponding to the Baseline (1961–2006), A1B/MYROC (2011–2060), and B2/HAD (2011–2060) Scenarios**

	Mean	Min.	Max.	Standard Deviation
<b>Baseline Scenario with Climate Normals (1961–2006)</b>				
Precipitation (mm)	665	455	1049	110
Maximum temperature (°C)	25.8	22.6	27.6	1.2
$Q$	1.16	0.69	1.75	0.27
<b>A1B/MYROC (2010–2060)</b>				
Precipitation (mm)	537	353	832	102
Maximum temperature (°C)	29.0	24.8	32.9	1.7
$Q$	1.12	0.57	2.09	0.41
<b>B2/HAD (2010–2060)</b>				
Precipitation (mm)	712	502	1036	112
Maximum temperature (°C)	28.1	23.9	30.9	1.6
$Q$	0.87	0.59	1.21	0.17

Note: Climate normals used in the baseline scenario were only available from 1961 to 2006.



**FIGURE 8.12** Predicted 5-year periodic basal area increment (BAI; m<sup>2</sup>/ha/5 years) for a 50-year-old stand (a) and 100-year-old stand (b) in response to density (20 m<sup>2</sup>/ha versus 30 m<sup>2</sup>/ha) and *q* values (as described in text).

Although the average BA for all 104 plots under the two future climate scenarios after 50 years was relatively similar to the average BA projected using past climate normals, there was substantial variability in the 5-year periodic projections. Regardless of age, lower density stands experience an increase in BA increment relative to higher density stands at low *q* values. This trend is reversed when *q* increases beyond a given value. The *q* value at which BA increment in higher density stands exceeds that in lower density stands is dependent upon stand age (Figure 8.12). The reversal in the trend in BA increment occurs at slightly higher *q* values for younger rather than older stands. Although the patterns in BA increment in young versus old stands can be described by normal stand development/production patterns, trends between BA increment and climate are visible. Young stands experience an increase in BA increment with increasing *q* values. In older stands, BA increment appears to increase with increasing *q* values in higher density stands, while BA increment decreases with increasing *q* values in higher density stands (Figure 8.12).

**Discussion**

Based on this, as well as other published studies (Pan et al. 1997; Fekedulegn et al. 2008; Klos et al. 2009), the growth of yellow poplar can be considered to be sensitive to past and future fluctuations in precipitation. Contrary to the findings of Pan et al. (1997), temperature was not found to significantly influence the accumulation of stand-level BA over time. Using BA projections based on past climate normals as a reference, the effects of altered precipitation predicted to occur under the A1B/MYROC and B2/HAD scenarios on BA accumulation were minor. Although the overall average BA accumulation differed little among the climate scenarios, substantial variation among 104 plots

was observed. Much of this variation was due to differences in the interaction between  $q$  and age and  $q$  and initial stocking among the 104 plots.

The fact that stand structure and age, two factors that can be controlled through active management, significantly interact with climate to control growth and productivity suggests there are management strategies that can be utilized to increase resiliency of yellow poplar stands to a changing climate. According to the model, as stocking increases, the sensitivity to decreased precipitation increases. Although the sensitivity of the accumulation of BA to  $q$  decreases as stands age, the general decline in stand productivity associated with stand age (Ryan et al. 1997) is evident. Aging of forest stands in the southern Appalachians coupled with extensive forest management may leave highly productive yellow poplar stands susceptible to decreased productivity over the next 50 years. Thinning may make yellow poplar less vulnerable to decreased BA production under future climate scenarios during summers when June rainfall is low relative to July rainfall (i.e., low  $q$  values). The decision to thin these stands, for a relatively small increase in production may, however, conflict with the economics surrounding thinning in what are generally extensively managed systems.

The model presented here does not include loss of production due to mortality. Consequently, any decrease in stand-level production due to drought-induced mortality is not accounted for in the model. Given that most of the yellow poplar mortality that occurred in the dataset used in this study was due to stem breakage and/or wind-related (e.g., uprooted) events, it does not appear that drought is a significant or common cause of yellow poplar mortality (Klos et al. 2009). Under past climatic conditions, precipitation throughout the Blue Ridge was equally distributed throughout the year. Diameter growth begins in June and is complete by the end of August (Beck 1985). Changes in the length of growing season, in particular earlier leaf-out due to increased spring temperatures, could alter when diameter growth commences and terminates. Consequently, the model presented here, where June and July rainfall are the only climate variables influencing BA growth, may not adequately describe future conditions.

Although yellow poplar is sensitive to altered precipitation, the effect of climate on BA accumulation is dwarfed by the influence of stand age, site quality, and stand structure. The nonexistent role of absolute temperature or precipitation in the model is likely a function of two factors: (1) the fact the model was built using 5-year average climate and stand data as opposed to annual climate and stand data; and (2) the position on the landscape where yellow poplar is predominant. Even under the most extreme future climate scenario, A1B/MIROC, average growing season precipitation approximates the minimum total annual precipitation that occurs throughout the current range of yellow poplar (Beck 1990). In the southern Appalachians, yellow poplar grows best on moist, well-drained soils with high available soil moisture common to stream bottoms, coves, and moist slopes (Beck and Della-Bianca 1981). The plots used to develop the climate-sensitive growth model are all highly productive stands where available moisture exceeds potential evapotranspiration even during periods where precipitation may be below normal (Beck 1985). Consequently, this model may not accurately describe the growth response of yellow poplar outside high-quality sites in the Blue Ridge Physiographic Province (e.g., ridge-tops, Piedmont region, southern extent of its range, etc.).

### **CASE STUDY 3—EFFECTS OF SILVICULTURE ON CARBON BALANCE OF LOBLOLLY PINE AND SLASH PINE PLANTATIONS IN SOUTHEASTERN UNITED STATES**

An important advantage of forest management approaches to CO<sub>2</sub> mitigation is that silvicultural technologies are well developed, in place, and inexpensive to apply. In managed forests, C stocks can be divided into two major pools: *in situ* C in standing biomass (above- and belowground) and soil organic matter, and *ex situ* C sequestered in products created from harvested wood (Marland and Marland 1992). Sustainable forest management has the potential to greatly influence both *in situ* and *ex situ* C pools (Johnsen et al. 2001a). Loblolly pine and slash pine play an important role in mitigation of CO<sub>2</sub> emissions due to their high productivity and extensive planting throughout

the southeastern United States. Accurate determinations of C stocks and understanding of factors controlling C dynamics in both species are essential for C offset projects and the development of sustainable management systems.

In this case study, we assessed the effects of silvicultural management on forest C stocks per unit area in loblolly pine and slash pine plantations established in the southeastern U.S. Coastal Plain. We used the models reported by Gonzalez-Benecke et al. (2010 and 2011a), to account for net C stock dynamics, and to address the following questions: (1) To what extent do extended rotations increase C sequestration in southern pine forests? (2) Is initial planting density an important factor for C storage? (3) What is the effect of thinning on net C storage for each species? (4) Under similar site quality and silvicultural treatments, which species accumulates more C? The models used in this case study were validated against published data of net ecosystem production, forest floor mass, and above- and belowground living pine biomass.

## Materials and Methods

**Models:** Allometric and biometric equations were combined with growth and yield models to estimate C stocks and dynamics for loblolly pine and slash plantations in the southeastern United States (Gonzalez-Benecke et al. 2010 and 2011a). *In situ* C stock (C stored in living pine trees + understory + forest floor + coarse woody debris + standing dead trees) was determined using growth and yield models for loblolly pine (Harrison and Borders 1996; Logan et al. 2002) and slash pine (Pienaar et al. 1996; Martin et al. 1997; Yin et al. 1998; Bailey et al. 1980), and combined with allometric and biometric equations reported for both species (for details of equations see Gonzalez-Benecke et al. 2010, 2011a). The models use quadratic mean diameter (QMD) and number of trees per hectare, estimated by the growth and yield models, as inputs for biomass equations to determine above- and belowground C stock. Projected LAI and litterfall were estimated from the model reported by Gonzalez-Benecke et al. (2012), where LAI was set to be proportional to SI and stand density index, and current year needlefall was set to be correlated with previous year mean annual projected LAI. The model also determined litterfall from needlefall using age-related needlefall-to-litterfall ratios for loblolly and slash pine. Forest floor biomass accumulation was determined as the sum of yearly litterfall inputs corrected for decay loss using the equation to estimate decay rate of the forest floor as reported by Radtke et al. (2009). Understory biomass accumulation was estimated from the equation reported by Gonzalez-Benecke et al. (2010), which predicts understory biomass as a function of stand LAI. At the time of thinning, reductions in pine LAI were set to be proportional to reductions in BA due to thinning and, therefore, needlefall, litterfall, and forest floor and understory biomass were affected due to their LAI-dependence. Standing dead trees, estimated from mortality equations of the growth and yield models, were also incorporated into the dead component of total biomass. At thinning and final harvest (clear-cut), logging slash (root and crown biomass plus stem residues) from harvested trees were also included into flux calculations and incorporated into the dead biomass pool. Stem residues were obtained by assuming a harvest efficiency of 88% and 87% of stand volume, for loblolly pine and slash pine, respectively (Bentley and Johnson 2008a, 2008b, 2009). Initial C accumulated from the previous rotation in coarse root debris, forest floor, and aboveground coarse woody debris was assumed to be  $\sim 55.4 \text{ Mg C ha}^{-1}$  (Pehl et al. 1984; Van Lear and Kapeluck 1995; Clark et al. 2004; Miller et al. 2006; Eisenbies 2009).

To estimate *ex situ* C pool (C stored in wood products), harvested groundwood estimated from thinnings or clear cuts with the growth and yield models was assigned to three main product classes, sawtimber (ST), chip-and-saw (CNS), and pulpwood (PW), using the models proposed by Harrison and Borders (1996) and Pienaar et al. (1996). Industrial conversion efficiencies of 65%, 65%, and 58% were assigned to ST, CNS, and PW, respectively (Spelter and Alderman 2005; Smith et al. 2006). In addition, all the product types were divided into four life span categories (Liski et al. 2001; Gundimeda 2001) and adapted to loblolly pine and slash pine utilization patterns in the southeastern United States (Row and Phelps 1991; Skog and Nicholson 1998).

The model also accounts for C costs of silvicultural operations. Emissions of C by silvicultural activities were determined from Markewitz (2006) and White et al. (2005). C release in transportation of raw material from the forest to the mill was estimated according to White et al. (2005), assuming an average distance of 100 km from forest to mill, load per logging truck of 24 m<sup>3</sup>, and fuel economy of diesel logging truck of 2.6 km L<sup>-1</sup>.

Net C stock (Mg C ha<sup>-1</sup>) was defined as follows: Net C stock = Total C *in situ* (C stored in living pine trees + understory + forest floor + coarse woody debris + standing dead trees) + Total C *ex situ* (C stored in wood products ST + CNS + PW) – Total C cost (silvicultural activities, including transportation of logs to mills). Carbon mass (Mg C ha<sup>-1</sup>) was calculated by using an average C content of 50% for pine and understory biomass components (Clark et al. 1999; Johnsen et al. 2004). The models did not include changes in soil C. It was assumed that C storage in soil was not affected by forest management in southern pines plantations (Gholtz and Fisher 1982; Harding and Jokela 1994; Han et al. 2007; Johnson and Curtis 2001).

*Silvicultural management scenarios:* The effects of silvicultural treatments (planting density, thinning, and rotation length) on C sequestration were analyzed by simulating C flux under different scenarios of loblolly pine and slash pine plantations established in the southeastern U.S. Lower Coastal Plain. The initial stand parameters used for simulations (set equal for both species) were: base SI = 22 m, bedding, weed control at planting and at age 1, and NP fertilization at ages 5 (135 kg ha<sup>-1</sup> N + 28 kg ha<sup>-1</sup> P) and 10 (225 kg ha<sup>-1</sup> N + 28 kg ha<sup>-1</sup> P). After site preparation, fertilization, and weed control treatments, the model estimated that base SI was increased to effective SI of 25.3 and 25.8 m, for loblolly pine and slash pine, respectively. The effect of initial stand density was evaluated by running the model under planting densities of 500, 1500, and 2500 trees ha<sup>-1</sup>. Rotation length effects on C sequestration were assessed by evaluating the model for 15, 30, and 45 years harvesting age. Both planting density and rotation length analysis were carried out under unthinned conditions. The effects of thinning (as a percentage of living trees removed) were assessed by evaluating the model under different combinations of thinning age (8, 12, and 16 years) and removal intensity (20%, 40%, and 60% of living trees removal), for stands with planting density of 1500 trees ha<sup>-1</sup> and 25 years rotation length. For all simulations, estimates of average C stock were reported as the average of all yearly values for a simulation length of 175–180 years, depending on rotation age used. In the case of scenarios with rotation lengths of 15, 25, 30, and 45 years, the number of rotation cycles used to calculate net C stock were 12, 7, 6, and 4, respectively.

## Results

*Silvicultural management effects on C sequestration:* Under conditions used in the simulations for unthinned loblolly pine stands, the average net C stock for a 180-years simulation length increased from 131 to 173 Mg C ha<sup>-1</sup>, when rotation length was increased from 15 to 30 years (Table 8.5). If the rotation length was extended to 45 years, the average net C stock increased to 181 Mg C ha<sup>-1</sup>, respectively. On unthinned slash pine stands with rotation lengths of 15, 30, and 45 years, the average net C stock was 111, 175, and 183 Mg C ha<sup>-1</sup>, respectively. For both species, across rotation lengths, *in situ* C stock accounted for between 71% and 76% of the gross C sequestration (not including silvicultural C costs). The magnitude of emissions associated with silvicultural activities (including transportation) was between 1.6% and 1.9% of the gross C stock, with larger proportion on stands with shorter rotations (Table 8.5). The relative impact of the different woody products varied depending on rotation length scenario. For example, for unthinned loblolly pine stands, ST accounted for 1%, 11%, and 15% of gross C sequestration, for rotation lengths of 15, 30, and 45 years, respectively. In contrast, CNS followed an opposite trend, accounting for 27%, 15%, and 7%, for the same rotation length scenarios, respectively. In slash pine stands the proportion of ST was lower, accounting for 1%, 9%, and 10%, while CNS accounted for 25%, 17%, and 8% of gross C sequestration for the same rotation length scenarios, respectively (Table 8.5). Across different rotation lengths, the forest floor + dead trees + coarse woody debris (FF<sub>D</sub>) components averaged 42 and 38 Mg C ha<sup>-1</sup>, for loblolly pine and slash pine stands, respectively (between 31% and 34% of

**TABLE 8.5**  
**Average Carbon Stock for Loblolly Pine (LOB) and Slash Pine (SLA) Plantations for 175–180 Years Simulation Length under Different Rotation Length Scenarios**

C Pool	15 Years		30 Years		45 Years	
	LOB	SLA	LOB	SLA	LOB	SLA
<i>In situ</i>	93.4	80.7	130.3	131.3	144.4	150.9
Living pine	50.7	44.2	87.3	90.0	102.4	110.1
Understory	1.1	1.7	1.0	1.8	0.9	1.8
Forest floor + dead trees	41.6	34.8	42.0	39.5	41.1	39.1
<i>Ex situ</i>	40.0	32.1	45.3	47.0	40.1	34.6
Sawtimber	1.5	0.4	18.9	16.3	27.2	18.6
CNS	35.8	28.3	25.6	29.6	12.6	15.5
Pulpwood	2.7	3.4	0.8	1.1	0.3	0.5
Silviculture emissions	-2.5	-2.1	-3.1	-2.8	-3.3	-3.0
Net C stock	130.9	110.6	172.5	175.4	181.3	182.5

*Note:* All units are average carbon stock ( $\text{Mg C ha}^{-1}$ ) for first five rotations. All scenarios included base site index = 22 m, planting density = 1500 trees  $\text{ha}^{-1}$ , bedding, weed control at planting and at age 1, and NP fertilization at age 5 and 10 years.

total *in situ* C stock) and the understory averaged 1.0 and 1.7  $\text{Mg C ha}^{-1}$ , for loblolly pine and slash pine stands, respectively (less than 2% of total *in situ* C stock). Between species, for 15-year rotation length, average living tree biomass for the first five rotations of loblolly pine stands was 6.5  $\text{Mg C ha}^{-1}$  larger than that of slash pine, but as rotation length was extended to 45 years, slash pine living tree biomass catches up and exceeds loblolly pine's, averaging 7.6  $\text{Mg C ha}^{-1}$  more.

The effect of planting density on average net C stock was similar for both species: planting more trees increased stand C sequestration. Loblolly pine and slash pine stands planted with 2500 trees  $\text{ha}^{-1}$  can store 43 and 36  $\text{Mg C ha}^{-1}$  more than stands planted with 500 trees  $\text{ha}^{-1}$ , respectively (Table 8.6). For both species, the effect of planting density on average net C stock was largely reflected

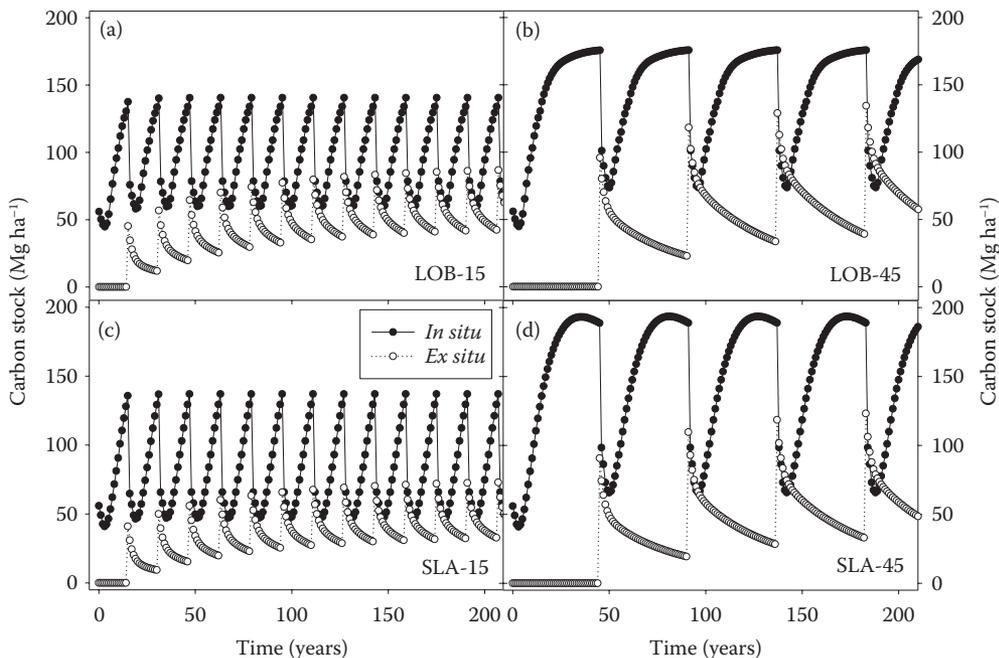
**TABLE 8.6**  
**Average Carbon Stock for Loblolly Pine (LOB) and Slash Pine (SLA) Plantations for 180 Years Simulation Length under Three Different Planting Density Scenarios**

C Pool	500 Trees $\text{ha}^{-1}$		1500 Trees $\text{ha}^{-1}$		2500 Trees $\text{ha}^{-1}$	
	LOB	SLA	LOB	SLA	LOB	SLA
<i>In situ</i>	94.7	92.8	122.2	119.2	138.6	135.9
Living pine	61.9	58.9	78.9	78.2	88.8	91.7
Understory	1.1	1.8	1.0	1.7	1.0	1.7
Forest floor + dead trees	31.7	32.2	42.4	39.2	48.9	42.5
<i>Ex situ</i>	45.8	49.8	46.2	47.4	45.7	43.5
Sawtimber	31.8	40.5	13.9	11.5	7.1	1.3
CNS	13.7	9.1	31.2	34.4	36.9	38.5
Pulpwood	0.3	0.2	1.1	1.5	1.7	3.7
Silviculture emissions	-2.9	-2.5	-3.3	-3.0	-3.4	-3.4
Net C Stock	137.5	140.1	165.2	163.5	180.9	176.1

*Note:* All units are average carbon stock ( $\text{Mg C ha}^{-1}$ ) for first five rotations. All scenarios included base site index = 22 m, rotation length = 25 years, bedding, weed control at planting and at age 1, and NP fertilization at age 5 and 10 years.

in *in situ* rather than *ex situ* C pools. Across species, living pine tree biomass and  $FF_D$  were largely increased from  $\sim 60$  and  $32 \text{ Mg C ha}^{-1}$  to  $\sim 90$  and  $46 \text{ Mg C ha}^{-1}$ , respectively, as planting density increased from 500 to 2500 trees  $\text{ha}^{-1}$  (Table 8.6). The C storage in woody products was reduced by 0.1 and  $6.2 \text{ Mg C ha}^{-1}$ , for loblolly pine and slash pine stands, respectively, as planting density increased from 500 to 2500 trees  $\text{ha}^{-1}$ . Even though C storage in woody products was affected little by planting density, the C stored in different woody products varied depending on planting density scenario. For instance, ST decreased from 32 and  $41 \text{ Mg C ha}^{-1}$  to 7 and  $1.3 \text{ Mg C ha}^{-1}$ , for loblolly pine and slash pine stands, respectively, when planting density was increased from 500 to 2500 trees  $\text{ha}^{-1}$ . In contrast, CNS followed an opposite trend, increasing from 14 and  $9 \text{ Mg C ha}^{-1}$  to 37 and  $39 \text{ Mg C ha}^{-1}$ , for loblolly pine and slash pine stands, respectively, for the same planting density increment (Table 8.6). From a total silviculture C cost perspective, fertilization accounted for  $0.57 \text{ Mg C ha}^{-1}$  (two fertilizations), representing between 18% and 27% of the total silvicultural C emissions. Harvest and transportation of woody products accounted for more than 50% of the total silvicultural C emissions.

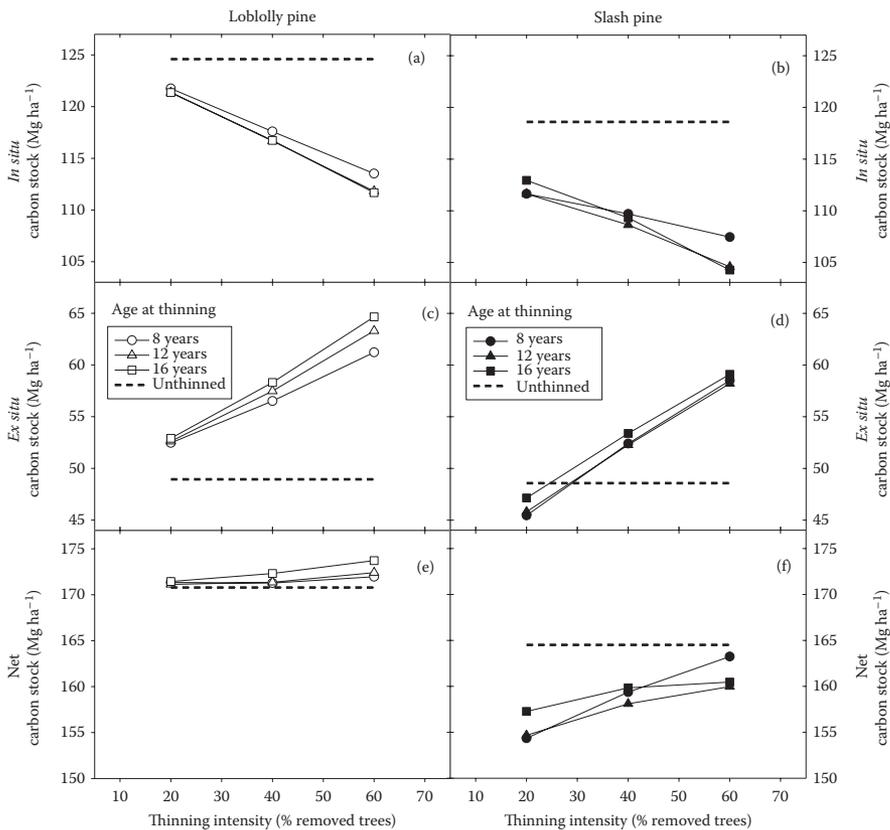
In general, after 200–250 years, C flux in the woody products converged to stable values, reaching quasi-equilibrium minimum and maximum values. At rotation ages of 15 and 45 years, *in situ* C stocks were 141 and  $178 \text{ Mg C ha}^{-1}$  for loblolly pine stands (Figure 8.13a and b), and 136 and  $193 \text{ Mg C ha}^{-1}$  for slash pine stands, respectively (Figure 8.13c and d). For loblolly pine stands, total woody products C stock increased each rotation from 57 and  $102 \text{ Mg C ha}^{-1}$  during the first rotation, up to 91 and  $153 \text{ Mg C ha}^{-1}$  at the end of the fifth rotation, for rotation ages of 15 and 45 years, respectively (Figure 8.13a and b). In the case of slash pine stands, *ex situ* C stock increased from 50 and  $105 \text{ Mg C ha}^{-1}$  during the first rotation, up to 77 and  $152 \text{ Mg C ha}^{-1}$  at the end of the fifth rotation, for rotation ages of 15 and 45 years, respectively (Figure 8.13c and d). Differences in tree size (diameter and height) and number of trees remaining due to different rotation age scenarios created different woody products pools that had different life spans. While for 15 years rotation length, PW represented 12% and 15% of *ex situ* C stock for loblolly pine and slash pine stands, respectively, ST accounted for 12% and 4% of *ex situ* C stock for loblolly pine and slash pine stands, respectively.



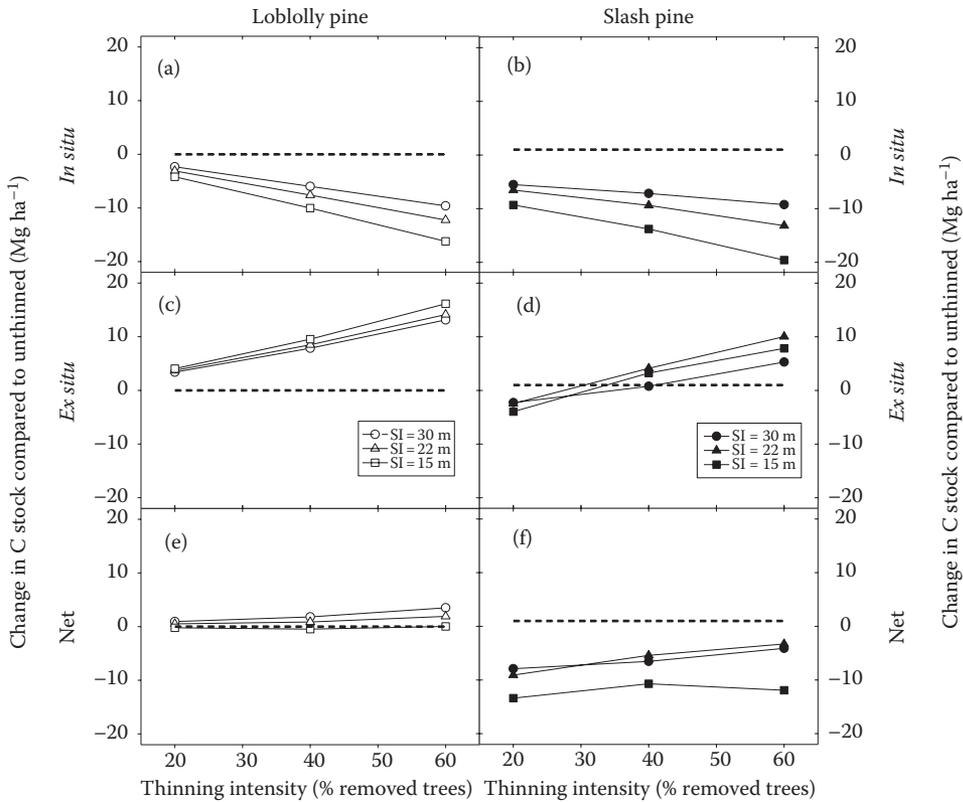
**FIGURE 8.13** *In situ* and *ex situ* C stock for unthinned loblolly pine (LOB) and slash pine (SLA) plantations under different rotation length for a 200-year simulation period.

On the other hand, for 45-years rotation length, PW accounted for 2% of total *ex situ* C stock for both species, and ST accounted for 69% and 56% of total *ex situ* C stock for loblolly pine and slash pine stands, respectively (Figure 8.13a–d). In general, C stored in products derived from PW (e.g., paper, packing material, office supplies, etc.) had a negligible effect on net C sequestration; between harvest events (thinning or clear cutting), the amount of C stored diminished toward zero, while C stored in CNS and ST increased between harvests (data not shown).

In general, for a stand planted on land with a SI = 22 m with 1500 trees ha<sup>-1</sup> and managed under a 25-year rotation, thinning had a small effect on net C stock. Even though the effect was minimal for loblolly pine (Figure 8.14e), there was a 4.9% overall reduction in net C stock in slash pine (Figure 8.14f). For both species, for any given thinning intensity, there was a small effect due to the age of thinning on *in situ*, *ex situ*, and net C stocks (Figure 8.14). Increments in thinning intensity produced a quasi-constant decline in *in situ* C stock, independent of thinning age. *Ex situ* C storage had an opposite response to thinning: the more intensive the thinning regime, the more gain in woody products C storage, and this effect was only slightly affected by thinning age (Figure 8.14c and d). The reduction of *in situ* C stock due to thinning was counteracted by increasing the *ex situ* C stock, producing a null effect on net C stock. However, for loblolly pine plantations with thinning intensity of 60%, there was a small (<2%) increment on net C stock. In the case of slash pine plantations, on the other hand, thinning reduced net C stock by 3.6%. Across thinning age and intensity, there was an average change in net C stock of 1.0 and -5.9 Mg C ha<sup>-1</sup>, for loblolly pine and slash pine stands, respectively (Figure 8.14e and f).



**FIGURE 8.14** Effects of stand age at thinning and thinning intensity on average *in situ*, *ex situ*, and net C stock plantations for first five rotations for loblolly pine (open symbols, a, c, e) and slash pine (black symbols, b, d, f) plantations.



**FIGURE 8.15** Effects of site index and thinning intensity on average *in situ*, *ex situ*, and net C stock for first five rotations for loblolly pine (open symbols, a, c, e) and slash pine (black symbols, b, d, f) plantations.

The combined effect of thinning intensity and SI was expressed as the change in C stock compared to unthinned stands. Overall, independent of stand SI, thinning has a small effect on average net C stock (Figure 8.15a–f). However, *in situ* C stock was more reduced in stand with base SI = 15 m (Figure 8.15a and b). The effect of low thinning intensity (i.e., 20%) on *in situ* C stock was similar across SI. For example, the reduction in *in situ* C stock was between 2.3 and 4.2 Mg C ha<sup>-1</sup>, for loblolly pine stands, and between 5.5 and 9.3 Mg C ha<sup>-1</sup>, for slash pine stands, both thinned with 20% removal intensity. When thinning intensity was increased to 60% removal of living trees, the reduction in *in situ* C stock was between 9.6 and 16.2 Mg C ha<sup>-1</sup>, and 9.3 and 19.6 Mg C ha<sup>-1</sup>, for loblolly pine and slash pine stands, respectively.

## Discussion

When stand C density was compared between loblolly and slash pine under similar levels of site quality (base SI = 22 m) and silvicultural inputs over a 25-year rotation length, living pine C stocks of loblolly and slash pine were generally similar. Nevertheless, there was a trend that under shorter rotation (i.e., 15 years), net C stock of loblolly pine was 18.3% larger than slash pine, mostly due to faster initial growing rates of loblolly pine trees. This general trend has been documented elsewhere. For example, Colbert et al. (1990) reported, for 4-year-old trees, that loblolly pine stands had 63% more needle biomass and 36% more aboveground biomass than slash pine stands growing under similar site and silvicultural characteristics. At age 6, at the same site, Dalla-Tea and Jokela (1991) pointed out that loblolly pine stands had 40% more LAI and 68% more aboveground biomass than slash pine stands. Burkes et al. (2003) also found for 4-year-old trees, that leaf and fine root biomass of slash pine stands were significantly smaller than in the loblolly pine stands. However, on the

same stands measured by Colbert et al. (1990) and Dalla-Tea and Jokela (1991), Vogel et al. (2011) reported that at age 26 years slash pine had 4.7% more *in situ* C stock than loblolly pine. The latter authors reported that under unfertilized conditions, even with sustained elimination of understory vegetation, living tree C stocks of slash pine were larger than loblolly pine. When nutritional limitations were eliminated through fertilizer additions, living tree stand biomass of loblolly pine was larger than of slash pine. As nutritional demands and the responses to fertilization for loblolly pine tend to be larger than slash pine (Jokela et al. 2010), differences in nutrient requirements and nutrient use efficiency between the two species should be taken in account when developing sustainable and ecological forestry regimes. In our analysis, the fertilization regime used included two applications, which may not be sufficient to support the demands of loblolly pine, especially under longer rotations scenarios.

Increasing initial planting density in the range tested in this study had a positive effect on net C storage, and the effects of planting density on C storage were most apparent in the *in situ* C pool, affecting both living tree biomass and  $FF_D$  biomass accumulation. Even though raising the planting density increased the proportion of fixed C used in stem production in loblolly pine (Burkes et al. 2003), this effect was not reflected in the *ex situ* C pool. As planting densities increased, there was a tendency to decrease sawtimber products yields, affecting the average *ex situ* C pools; however, the increase in forest floor, coarse woody debris, and total living tree C storage largely counteracted that negative effect.

Increasing the rotation length increased C stock in both species. Reports for other conifer species (Liski et al. 2001; Harmon and Marks 2002) have indicated similar effects of rotation length on C storage; that is, extended rotations increased C sequestration in conifer forest plantations. As Canadell and Raupach (2008) pointed out, longer harvesting cycles represent one of the major management strategies used to increase forest C density. Nevertheless, the inclusion of biomass harvest for fossil fuel offset might change our conclusions, especially when shorter rotations include provisions for improved technology at the end of each rotation. Further research is needed in this area, and this model is a tool to address these types of questions.

Due to compensating effects of *in situ* and *ex situ* C storage, net C stocks were only minimally affected by thinning. Most of the studies that have addressed the impacts of thinning on C budgets in pine ecosystems have only reported the responses of living pine biomass (Balboa-Murrias et al. 2006; Skovsgaard et al. 2006; Chiang et al. 2008; Finkral and Evans 2008), and only a few studies have reported the impacts of thinning on total *in situ* C (Boerner et al. 2008; Jiménez et al. 2011) or NEP fluxes (Campbell et al. 2009; Dore et al. 2010). All the previously cited studies concluded that there was a reduction in pine C sequestration after thinning. Garcia-Gonzalo (2007), in a similar analysis that included *ex situ* C pools for mixed coniferous stands in Finland, reported a net reduction between 25 and 33 Mg C ha<sup>-1</sup> in living trees biomass and a net increase between 30 and 45 Mg C ha<sup>-1</sup> in harvested timber. Even though the wood extracted in thinning was primarily pulpwood that had an impact on *ex situ* C sequestration, increased growth of residual trees due to thinning promoted the production of larger tree size classes at final harvest. These long-lived products increased the *ex situ* C pool, compensating for the reduction in *in situ* C associated with thinning. When *ex situ* pools were considered, the possible economic benefits of thinning were not in opposition to maintaining or increasing net C stock.

Reporting of C stock in harvested wood is not mandatory under the Intergovernmental Panel on Climate Change (IPCC 2006), but the enhancement of that C pool could provide important GHG emission offsets. The *ex situ* C pool could be influenced by both the final utilization of particular products, and also by substituting wood for more C-intensive materials. If waste wood and forest biomass residues were used as substitutes for fossil fuels (Galik 2009), or if long-lasting wood products take the place of more C-intensive materials like concrete or steel (Perez-Garcia 2005), then the mitigation impacts of *ex situ* C stocks could be even larger.

Using published models to evaluate the effects of planting density, thinning, and rotation length on C sequestration for loblolly pine and slash pine plantations in the southeastern United States, we

conclude that: (1) shorter rotations were not as effective for C sequestration as extended rotations that increased average net C stock; (2) increasing initial planting density had a positive effect on net C storage; (3) if woody products, which accounted for ~30% of the net C stock, were incorporated into the C inventory, thinning will have a tendency to be C neutral because of the counteracting effects of *in situ* and *ex situ* C stocks; and (4) emissions due to silvicultural and harvest activities were small compared to the magnitude of the total stand C stock.

## MANAGEMENT OPTIONS TO DEAL WITH CLIMATE CHANGE IMPACTS

### THE POTENTIAL FOR AND APPLICATION OF BIOENERGY IN SOUTHERN FORESTS

In 2009, total energy usage in the United States was estimated at approximately 94.5 quadrillion BTUs (Quads). Sources of energy in decreasing order of importance were petroleum (37%), natural gas (25%), coal (21%), nuclear (9%), biomass (4%), and other sources such as hydroelectric, wind, geothermal, and solar (4% in total) (US DOE 2011). This dependence upon many nonrenewable and high C emissions sources places the United States in a challenging economic, political, and ecological position. In recent years, increasing emphasis has been placed on developing increased bioenergy production capacity in the United States. Excellent reviews of the early technological developments and pathways forward have been provided by Tolbert and Wright (1998) and Tuskan (1998). Later summaries of further developments and outstanding challenges have been provided by Mayfield et al. (2007), Buford and Neary (2010), and Dale et al. (2011). There are several driving forces behind this push for bioenergy, including potential reductions in greenhouse gas emissions from use of nonrenewable C sources, desire to maintain or increase direct C sequestration, the desire to decrease United States' dependence upon foreign energy supplies, and the desire to increase economic diversity and potential economic developments of many depressed rural economies. These diverse goals can be addressed through the increased production and utilization of renewable bioenergy crops in the United States. Increased bioenergy production goals were formalized in the Energy Independence and Security Act of 2007 (EISA). The U.S. Renewable Fuels Standards (RFS2) is an integral part of this Act and became effective in 2010. As part of this act, the United States has committed to producing and using 136 billion liters of renewable transportation fuel per year by 2022. A significant component of this target is currently being met through corn-based ethanol. Other assessments have also addressed potential bioenergy production scenarios in the United States and while not yet formally incorporated into governmental laws, they view the bioenergy production potentials in a longer-term and more aggressive way. In 2005 the Department of Energy (DOE) and Oak Ridge National Laboratory developed a report titled "Biomass as a feedstock for a bioenergy and bioproducts industry: the technical feasibility of a billion-ton annual supply," which was updated in 2011 (US DOE 2011). Total available biomass at any given time period is a function of the rate of technological advancements made with potential bioenergy crops and the prices paid by biomass consumers (US DOE 2011). Total biomass available in dry Mg is projected to be approximately 430 million Mg in 2012 based on current growth projections. This increases to 1.0–1.5 billion Mg under low and high rates of technological advancements, with a base cost of \$54 per Mg of biomass delivered (US DOE 2011). Under these modeling approaches, two important changes occur that dramatically increase the supply of woody biomass. At higher prices, a portion of woody biomass in the southern states normally harvested for pulpwood production is redirected toward bioenergy production. In addition, dedicated energy crop production increases dramatically. By 2030, energy crop production has been projected to increase to approximately 635 million Mg under some production scenarios. This is approximately 45% of 1.5 billion Mg.

In order for these aggressive deployment projections to occur, continued technology developments are needed. For the southeastern United States, potential candidate lists of dedicated energy crops were first presented by Wright (1994). This list of candidate species has been updated and

for the Southeast now includes switchgrass (*Panicum virgatum* L.), *Populus* species or hybrids, various tropical grasses, loblolly (*Pinus taeda* L.) or slash (*Pinus elliottii* Engelm) pine, sweetgum (*Liquidambar styraciflua* L.) and sorghum [*Sorghum bicolor* (L.) Moench] (Dale et al. 2011). In the more southern sections of the southeastern zone, eucalyptus species and hybrids are being investigated as a potential woody feedstock. Kline and Coleman (2010) summarized the potential advantages and disadvantages of the woody crop species based on previous experiences growing these species across the South. For *Populus*, the primary challenges to deployment included narrow site adaptabilities, variable productivity rates, intensive management and input requirements, and a general lack of drought tolerance. Potential advantages include high growth capacity under ideal conditions, extensive ongoing genetic research including genomic mapping, the existence of commercial stands, and the potential for further improvements in the medium term (Kline and Coleman 2010). For sweetgum, its advantages include being the most adaptable hardwood species across the region, being native to the region, having existing baseline research to support further development, and having the potential for improvement in the medium term. Disadvantages for sweetgum include more moderate productivity levels and more limited commercial deployment (Kline and Coleman 2010). Eucalypts possess the primary advantage of having the greatest potential growth rates among the woody species being considered. They are also generally adaptable to marginal site conditions, and have the potential for rapid improvement. Their primary limitation is susceptibility to cold damage, limiting potential deployment range. There are also potential environmental concerns with the deployment of an exotic species. In addition, their water use patterns merit additional research (Kline and Coleman 2010).

Among these potential species, there are wide variations in the relative stage of development of both the genetic resources of each species and the development of associated cropping regimes needed to support high yields needed to make production of these crops viable. Loblolly pine and slash pine are sharp contrasts to the species previously described. Pine plantation production systems are well established and widely deployed across the South and southeast. Current pine plantation regeneration and management techniques reflect over 60 years of concentrated research to optimize genetic sources, plant production techniques, site preparation procedures, plantation establishment densities and configurations and optimized competition control and nutrient management techniques. These sustained efforts have led to dramatic yield improvements and widespread adoption of pine plantation management across the South (Fox et al. 2007). Pine plantation acreage dramatically increased from less than a million hectares in 1952 to more than 13 million hectares in 1999 (Weir and Greis 2002). This represents a large and potentially available resource pool that could serve a key role in bioenergy production in the South.

Ultimately, the bioenergy crops that are developed and deployed across the South may be dependent upon current and future developments of conversion technologies. If direct burning of woody biomass or production of biofuels through gasification become the primary production processes, then pine may become the primary feedstock for these processes (Hinchee et al. 2009). Wood pellet production for heating and fuel purposes represents one significant and rapidly expanding market for pine biomass in the Southern United States. Global production expanded from approximately 8 million metric tons in 2007 to over 13 million tons in 2009. North American production constituted over 50% of worldwide wood pellet production in 2009 (Pirraglia et al. 2010). Consumption rates in Europe alone are projected to exceed 50 million tons per year in 2020 (Pirraglia et al. 2010). These growth rates are driven primarily by European Union agreements that require at least 20% of energy consumption to be provided by renewable energy sources by 2020. Rapid expansion in pellet production facilities is occurring across the United States with greatest production capacity expansions occurring across the South. Many of these facilities utilize abundant pine wood resources to meet growing demands and it seems that production and utilization of this resource will become increasingly important in the future.

In contrast to ethanol produced from corn, there is no commercial-scale cellulosic ethanol production in the United States currently and significant obstacles to commercial development exist.

These limitations include: (1) lignocellulosic feedstock costs and availability, (2) high pretreatment costs associated with recalcitrant cellulosic feedstocks, (3) enzyme costs, (4) excessive capital investment requirements, and (5) low ethanol prices (Gonzalez et al. 2011). These inherent limitations are further compounded when suitability of pine wood characteristics are considered. Pine wood is generally much more difficult to process than many alternate hardwood fibers and is currently much more suitable for bioenergy processes that utilize gasification or direct-firing technologies (Hinchee et al., 2009). It is possible to alter wood chemical or physical characteristics through either traditional breeding or genetic engineering approaches and these approaches are currently being used to reduce or modify lignin types and increase cellulose concentrations in woody crops. If successful, these approaches could facilitate cellulosic ethanol production efficiencies. These modifications, however, can have unintended, undesirable secondary effects including reductions in photosynthetic rates and increased cavitation potential (Nehra et al. 2005; Hinchee et al. 2009). These approaches can be further complicated since feedstock optimization for utilization by one bioenergy process actually make that feedstock less suitable for utilization by other processes. As an example, the caloric value of lignin is approximately 40% greater than the caloric value of cellulose (Hinchee et al. 2009). Decreasing lignin concentrations could make a feedstock more suitable for cellulosic ethanol production, but could actually make this material less desirable for pyrolysis-based technologies. Advancements in this area are advancing rapidly and this suggests production and processing changes may interact to influence production technology.

At the current time, southern pine including loblolly and slash pine represent a large and potentially accessible resource for bioenergy production in the Southern United States. Southern pines have been widely studied and their primary productivity limitations are fairly well understood. Among the characteristics that have led the widespread deployment of pine plantations across the South are the relatively broad site adaptability of pine and its relatively low sensitivity to soil moisture stress. Changing these inherent soil characteristics can be difficult or expensive and other potential bioenergy species having similar adaptations to these factors as pine would be desirable. Irrigation for woody crop bioenergy production would be cost prohibitive for all but the most unique niche applications. Selection of crop species that exhibit high productivity under only optimal soil moisture conditions would limit widespread deployment. Loblolly pine has demonstrated site adaptability and water use characteristics that are desirable. Munsell and Fox (2010) estimated that dedicated biomass production management regimes utilizing loblolly pine could produce approximately greater than 5 dry Mg C ha<sup>-1</sup> year<sup>-1</sup> across wide areas of the South using currently available technologies, but that these regimes would be unlikely to maximize returns to landowners. Under current scenarios, integrated bioenergy and solid wood production systems are likely to maximize economic returns to landowners and it is assumed that the pine biomass produced in these integrated production systems could be utilized for either solid wood products or bioenergy wood depending upon market.

Jokela et al. (2004) summarized growth responses of loblolly pine in seven integrated silvicultural studies across the South, and these studies found growth responses ranging from 2- to 3.5-fold in the best treatments within each individual study. Stem biomass production in these studies ranged from approximately 32 to 83 Mg C ha<sup>-1</sup> year<sup>-1</sup> at age 15 (annual production of approximately 2–5.7 Mg C ha<sup>-1</sup> year<sup>-1</sup>). Farnum et al. (1983) estimated that if the exhibited trends continued, the biological growth potential of loblolly pine could approach approximately 7 Mg C ha year<sup>-1</sup> if other resource limitations were alleviated. Sampson and Allen (1999) calibrated the BIOMASS model and estimated that loblolly pine annual stem growth could approach 8 Mg C ha<sup>-1</sup> year<sup>-1</sup> with treatments that ameliorated resource limitations. Our most complete understanding of the role of water and nutrient limitations of stand productivity for southern pines is provided by studies that allow complete experimental control of water and nutrient availability. The Southeast Tree Research and Education Site (SETRES) study was established in the Sand Hills of North Carolina to understand these relationships. In this study, loblolly pine plantations are grown in a factorial experiment with two nutrient levels and two water levels (Albaugh et al. 1998, 2004) available. Results from this study supported the conclusion that nutrient limitations rather than water limitations were the

primary limit to productivity in young to mid-rotation loblolly pine stands in the southeast and dramatic growth increases are possible with optimized stand nutrition.

In contrast to pine, hardwood silvicultural and tree improvement efforts in the South date back to the 1960s, but until recently hardwood fiber demand and the prices paid for hardwood sources were insufficient to sustain continuous, aggressive research efforts. Interest in and support for hardwood research programs have varied widely over the last 50 years and sustained progress has not been made. To date, progress with any of these species has limited their operational deployment to a mere fraction of the acreage dedicated to pine plantations. Dvorak and Hodge (1998) estimated that there were approximately 80,000 hectares of hardwood plantations across the South. Technological or economic limitations currently restrict wider deployment. Of the potential woody perennial bioenergy species listed for possible deployment across the South, long-term integrated research is far behind the current state of knowledge for pine research. This creates technical hurdles that limit successful implementation of a large-scale hardwood bioenergy production program (Kline and Coleman 2010).

Development of future bioenergy feedstocks will be closely linked to the productivity levels that can be achieved and the resulting economic benefits the landowner can accrue (US DOE 2011). Deployment of rapidly growing eucalyptus plantations has, in select geographic regions, the potential to produce large quantities of biomass on short rotations. Eucalyptus has been a genera of interest since the 1960s because of the remarkable productivity that can be obtained under ideal circumstances, that is, Brazil, but deployment in the southern United States has been limited by winter cold damage. Estimates of eucalyptus productivity in Brazil can range from approximately 4 Mg C ha<sup>-1</sup> year<sup>-1</sup> for low intensity silviculture to yields exceeding 15 Mg C ha<sup>-1</sup> year<sup>-1</sup> using intensive management regimes and elite clones (Stape et al. 2010). In the United States, eucalyptus research has focused on developing plants with sufficient cold tolerance to survive and grow well in the Southern United States. In the 1970s through mid-1980s, hundreds of eucalyptus species and individual seed sources were tested across the South and initial results were promising (Hunt and Zobel 1978), but a series of cold winters in the late 1970s and early 1980s, which caused catastrophic damage to test plantings, tempered enthusiasm for further research efforts (Jahromi 1982). In recent years, new research has been initiated to identify additional cold-tolerant eucalyptus genotypes by both private companies and by the Forest Productivity Cooperative at North Carolina State University and Virginia Tech. Current research indicates significant progress toward identifying species with good growth and cold tolerance. Several promising cold-tolerant eucalyptus species have been identified including *Eucalyptus benthamii*, *Eucalyptus macarthurii*, and *Eucalyptus viminalis*. In preliminary testing, these plants have demonstrated cold tolerance through USDA hardiness zones 8A with low winter temperatures of -12.2°C to -9.5°C.

There are significant differences between the bioenergy production scenarios currently being examined for the Southern United States and traditional forestry practices developed and practiced in the region. Management of bioenergy plantations is likely to be substantially more intensive than traditional forestry practices. Integrated cropping regimes will be utilized that combine the most appropriate genetics sources with more intensive site preparation, competition control practices, and nutrient management regimes. During peak plantation production ages, bioenergy plantations will carry more leaf area than traditional plantations. This will have the potential to create greater nutrient demands and the potential for greater water use to sustain high productivity. These changes will place greater demands on site resources to sustain productivity. Understanding and managing these potential changes will be critical to effectively manage these production systems. Process models may be the best available tool to integrate responses from a wide diversity of existing studies (Johnsen et al. 2001b). Various models have been developed for loblolly pine and many of these models focus on the relationships between nutrient availability, nutrient uptake, leaf area production, and subsequent plantation growth responses (Landsberg et al. 2001; Ducey and Allen 2001; Comerford et al. 2006; Sampson et al. 2006). Use of these existing or development of new models directly tailored to bioenergy production scenarios may be the most efficient way to evaluate effects of bioenergy production in the Southern United States.

Previous studies have indicated that biomass removals associated with harvest do have the ability to reduce nutrient capital on sites, but it is more difficult to demonstrate that plantation productivity is consistently and negatively impacted by increasing biomass removals (Johnson and Todd 1998; Powers et al. 2005). Concerns for excessive nutrient removals may be more valid after several subsequent rotations and when hardwood species rather than pine are the crop species of interest. Cation removals in particular can be higher for hardwoods compared to pine and these removals increase with increasing harvest utilization, that is, whole-tree harvests versus stem-only harvests (Johnson et al. 1982). Whole tree versus stem only harvests had N, P, K, and Ca removals 2.9, 3.1, 3.3, and 2.6 times greater, respectively, than removals for bole only harvests in Tennessee. The total removal of nutrients during harvesting was closely related to total biomass removals in harvesting, which were 2.6 times greater in the whole-tree harvest. Fifteen years following harvest in the same study there were no differences in tree growth, soil C or N concentrations or soil bulk density. There were however differences in soil and plant K, Ca, and Mg between the treatments (Johnson and Todd, 1998). These results suggest that greater removal of biomass does have the potential to alter soil nutrient status for some nutrients but that these changes could be relatively small and difficult to detect especially over shorter sampling intervals.

Harvesting impacts including varying levels of biomass removals and soil compaction effects have been investigated in other studies. In the late 1980s and 1990s, the U.S. Forest Service installed a series of studies called the Long-Term Soil Productivity Studies (LTSP) across the United States. These studies include differing levels of organic matter removals and soil compaction as core treatments. In general, whole-tree harvest did not negatively impact plant growth relative to stem-only harvest treatments. Assessments after 10 seasons across a range of 26 sites indicated that plant productivity was unaffected by biomass removals during harvest. Soil compaction treatments again yielded variable responses, with plant production increasing on sandy soils in response to compaction, and decreasing on compacted clay soils (Powers et al. 2005). Harvesting impacts were also examined by Eisenbies et al. (2006, 2007). In this study, loblolly pine growth and stand level productivity were not adversely affected by organic matter removal or soil physical disturbance if these factors were ameliorated by bedding treatments. The complex and often subtle responses to organic matter removal or soil physical disturbance suggest a degree of resiliency in many systems to short-term changes. Over longer time periods the full effects are unknown. Given the projected yields of short rotation woody crop species and short harvest intervals, it is likely that potential nutrient removals will need to be balanced by subsequent nutrient amendments to increase or even sustain productivity (Heilman and Norby 1998). There could be negative long-term impacts, but the relative responsiveness in many systems to silvicultural treatments suggests that these potential limitations could be corrected.

The final factor influencing bioenergy crop production, which may in turn be influenced by bioenergy cropping regimes, is water availability. The inherent assumption in current bioenergy developmental plans is that bioenergy crops will be developed on marginal farmlands, pastures, or other underutilized lands. Conversion of crop lands to perennial woody crop systems has the potential to increase water use. This occurs through two factors: generally greater canopy interception and associated water losses and greater annual evapotranspiration. Annual transpiration will be greater for woody perennial crops due to longer active growth periods. Merely converting from idle cropland to any highly productive bioenergy crop would be expected to increase water use. The magnitude of these water use increases can be substantial. Amatya et al. (2002) calculated that loblolly pine stands in North Carolina could have annual evapotranspiration rates approximately 30% greater than adjacent croplands.

## **OTHER MANAGEMENT OPTIONS**

Pine plantation management provides a clear opportunity for managing forests for productivity and C sequestration in light of climate change (Groninger et al 1999; Johnsen et al. 2001a, 2004). Over the

past 50 years, the productivity of planted pine has tripled (Fox et al. 2007a). In the Southern United States, N availability typically limits pine productivity (Fox et al. 2007b). Over 400,000 ha year<sup>-1</sup> of pine plantations are now fertilized with N (Albaugh et al. 2007). Nitrogen fertilization increases productivity directly but it may also act synergistically to increase productivity by eliminating nutrition as a limiting factor (see above section on elevated CO<sub>2</sub>). Fertilization can also directly decrease C losses via soil respiration (Butnor et al. 2003) under high and low atmospheric CO<sub>2</sub> concentrations, likely by decreasing heterotrophic respiration (Gough and Seiler 2004). Intensive pine management often includes site preparation; on wetter sites this often includes bedding. In an extensive review of the literature, Jandl et al. (2007) indicate that soil disturbance is the clearest means of decreasing soil C and abruptly increasing soil respiration (Gough et al. 2005). Other management tools that directly impact C sequestration include species selection (see case study 1), modifying initial planting density and rotation length (see case study 3), and thinning (see case studies 2 and 3).

## GENETICS

Rapid warming and/or changes in precipitation will likely result in local genetic populations being less well adapted. In the Western United States many species have steep altitudinal and latitudinal clines, and such within-species maladaptation could be substantial (St. Clair and Howe 2007). In the east, clinal variation of important conifers with extensive ranges appears less distinct but is still important, and such information on geographic variation is used to deploy seed sources. Deployment guidelines (Schmidting 2001) were partly based on data from the Southwide Southern Pine Seed Source Study (SSPSSS). For example, the SSPSS for loblolly pine was conducted using 15 wild seed sources at many different locations across the species range. Data were collected between the 1950s and 1980s, and therefore under historical weather conditions. Schmidting (1994) used these data to create regression models to relate growth to temperature variables. He predicted that “local” loblolly pine sources will experience a 10% decrease in height if mean annual temperature increases by 4°C. Longleaf pine and shortleaf pine were estimated to decrease by 12% and 5%, respectively. Such predictions indicate that seed transfer or breeding zones will have to be modified in the future (Spittlehouse and Stewart 2004). Schmidting’s (1994) approach only predicted responses to temperature but there is also variation in disease, insect, and drought resistance (Wells and Wakeley 1966, van Buijtenen et al. 1976; Wells 1983; Gonzalez-Beneke and Martin 2010). Also, provenances of loblolly pine west of the Mississippi River appear more drought tolerant than Atlantic Coastal Plain sources. However, because they grow faster, Atlantic Coastal Plain sources are commonly planted in the Mid-South subregion (Lambeth et al. 2005). Such a deployment strategy may need to be reconsidered as the climate in the Mid-South becomes hotter and drier.

Nearly all planted loblolly pine are the result of at least one cycle of genetic improvement (McKeand et al. 2006). Family variation in drought tolerance has been documented in other species (Johnsen et al. 1999). Breeding for ideotypes (Martin et al. 2001; Nelson and Johnsen 2008), genotypes with a specific combination of desirable traits, has been proposed as a way to match genotype to site. Maier et al. (2012) identified two highly productive loblolly pine genotypes that carry disparate amounts of leaf area, suggesting they also differ in resource water and/or nutrient efficiency. Ideotype development and deployment, using relatively short rotation lengths, might provide a means to respond to rapidly changing environmental conditions. Additionally, more efficient nutrient and water use might result in such genotypes taking better advantage of increased atmospheric CO<sub>2</sub> concentrations.

In the Southern United States, such extensive provenance and progeny test data, as discussed above, are mostly available for the major southern pines. For less intensively managed species whose ranges are predicted to shift dramatically, contract, or disappear, diversity will need to be conserved via *in situ* and *ex situ* genetic conservation (Ledig 1986). For many species, decisions on genetic deployment or conservation in light of climate must be made based on evaluations of life history and ecological traits (Aitken et al. 2008). Such an approach is being taken by the

National Forest (Region 8) in the Southern United States. Potter and Crane (2010) have developed the “Forest Tree Genetic Risk Assessment System: A Tool for Conservation Decision-Making in Changing Times.” Although not solely addressing climate change impacts, it accounts for threats to species or species populations as they interact with climate change. The system deals with intrinsic characteristics (population structure, fecundity, and seed dispersal) and external threats (climate change, insects, and diseases). Twelve species and species groups have been identified for conservation (Crane 2011): Atlantic white cedar, red spruce, seven threatened or endangered oaks, several ashes, butternut, longleaf pine, pitch pine, Fraser fir, September elm, yellow buckeye, and Ozark chinquapin. One method proposed to engender *ex situ* and *in situ* conservation of such species is to establish restoration seed reserves (RSR) (Echt et al. 2011). RSRs will be made up of progeny from trees selected across the species range planted *ex situ* and culled to produce seed production areas high in genetic variation. Populations of seedlings produced from seed from RSRs will presumably genetically buffer forests from climate change effects and other biotic threats, creating more resilient forests.

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