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Forest Carbon Accounting Considerations in US Bioenergy Policy

Reid A. Miner, Robert C. Abt, Jim L. Bowyer, Marilyn A. Buford, Robert W. Malmsheimer, Jay O'Laughlin, Elaine E. Oneil, Roger A. Sedjo, and Kenneth E. Skog

Four research-based insights are essential to understanding forest bioenergy and "carbon debts." (1) As long as wood-producing land remains in forest, long-lived wood products and forest bioenergy reduce fossil fuel use and long-term carbon emission impacts. (2) Increased demand for wood can trigger investments that increase forest area and forest productivity and reduce carbon impacts associated with increased harvesting. (3) The carbon debt concept emphasizes short-term concerns about biogenic CO_2 emissions, although it is long-term cumulative CO_2 emissions that are correlated with projected peak global temperature, and these cumulative emissions are reduced by substituting forest bioenergy for fossil fuels. (4) Considering forest growth, investment responses, and the radiative forcing of biogenic CO_2 over a 100-year time horizon (as used for other greenhouse gases), the increased use of forest-derived materials most likely to be used for bioenergy in the United States results in low net greenhouse gas emissions, especially compared with those for fossil fuels.

Keywords: biogenic emissions, biomass energy, carbon debt, carbon dioxide, forestry investment, forest landowner, greenhouse gas, wood markets, wood products, wood fuel

A large and rapidly growing body of research focuses on the greenhouse gas (GHG) impacts of using forest bioenergy to substitute for fossil fuel and wood building products to substitute for concrete and steel, materials that require greater amounts of fossil fuel to produce than wood products. Forest bioenergy research on GHG impacts, especially from carbon dioxide (CO_2), sometimes produces widely varying and occasionally contradictory results. Differences can usually be explained by understanding the data used, the scenarios examined, the analytical framework employed, and the assumptions used in the analyses (e.g., see Cherubini et al. 2009, Lamars and Junginger 2013). In this review, we examine research on the GHG impacts of energy derived from forest biomass, which, for the purposes of this review, includes all parts of the tree, living and dead. The objective is to reveal insights that allow improved interpretation of research in this area. Our review is focused on the accounting for biogenic carbon and biogenic CO_2 and the potential impacts of CO_2 on global temperatures. Other concerns related to elevated atmospheric CO_2 (e.g., ocean acidity) are not addressed. GHGs other than CO_2 are discussed where relevant. This review does not address other aspects of using forest biomass for energy, such as the ecological implications of more intensive management for production of forest biomass. A number of potential issues have been identified regarding the sustainability of forest biomass removal including ecosystem structure, nutrient and carbon balances, biodiversity, and aquatic system impacts (e.g., see Berger et al. 2013). Biomass harvesting guidelines that attempt to address such issues are being developed (Evans et al. 2013a).

A Brief Review of the Research and Debate about GHG Benefits of Forest-Derived Energy

A review of research on GHG impacts of forest bioenergy reveals a 25-year transition from work that created a basic understanding of the life cycle benefits of displacing fossil fuels with forest biomass, to research focused on the timing of these benefits, and finally to research demonstrating the importance, in many settings, of markets and investment responses to the GHG mit-

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Affiliations: *Reid A. Miner (rminer@ncasi.org), National Council for Air & Stream Improvement, Research Triangle Park, NC. Robert C. Abt (bob_abt@ncsu.edu), North Carolina State University. Jim L. Bowyer (jimbowyer@comcast.net), Dovetail Partners. Marilyn A. Buford (mbuford@fs.fed.us), USDA Forest Service Research and Development. Robert W. Malmsheimer (rwmalmsh@esf.edu), SUNY College of Environmental Sciences and Forestry. Jay O'Laughlin (jayo@uidaho.edu), University of Idaho. Elaine E. Oneil (eoneil@u.washington.edu), Consortium for Research on Renewable Industrial Materials, University of Washington. Roger A. Sedjo (sedjo@rff.org), Resources for the Future. Kenneth E. Skog (kskog@fs.fed.us), USDA Forest Service Forest Products Laboratory.*

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igation benefits of forest bioenergy and the timing of those benefits.

Research on GHG mitigation benefits of forest bioenergy and the timing of those benefits extends back to at least the late 1980s (e.g., see Marland 1989). By 1990, in its First Assessment Report, the Intergovernmental Panel on Climate Change (IPCC)[1] had identified a number of "short-term" forest-based options for limiting GHG emissions, including,

> Partially replac[ing] fossil energy sources [with] sustainably managed sources of biomass which would reduce net emissions of additional CO_2 (IPCC 1990, p. 80)

In the 1990s, Marland and Schlamadinger (Schlamadinger and Marland 1996, Marland and Schlamadinger 1997) researched the temporal dynamics of forest systems to store carbon and to produce wood to displace fossil fuels either directly, via use as fuels, or indirectly, via displacement of more fossil fuel-intensive products. A key finding from their work was that

> The relative effectiveness of alternative forest and bioenergy strategies and their impact on net C [carbon] emissions strongly depend, for example, on the productivity of the site, its current usage, and the efficiency with which the harvest is used (Schlamadinger and Marland 1996, p. 275)

They also highlighted

> the importance of C sequestration as a function of time. In some…scenarios, biofuels strategies result in long-term benefits in net C fluxes despite small or negative benefits in the early stages (Schlamadinger and Marland 1996, p. 294)

Thus, by the mid-1990s, although the term "carbon debt" had not yet been coined, the concept was clearly understood, i.e., as a situation in which net emissions increase in the short term before long-term benefits are realized.

Through the 1990s and into the early 2000s, research into the mitigation potential of forests and forest-derived products expanded enormously. In the Fourth Assessment Report issued in 2007, IPCC synthesized research on the use of forests and forest products for GHG mitigation and concluded that

> In the long-term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fiber or energy from the forest, will generate the largest sustained mitigation benefit. Most mitigation activities require up-front investment with benefits and co-benefits typically accruing for many years to decades (Nabuurs et al. 2007, p. 543)

The timing of impacts and benefits was characterized in a general way with a graphic depicting the benefits from using forest-derived materials as being sustained, repeatable, and ongoing (Nabuurs et al. 2007, p. 550). Other than a brief mention of the use of purpose-grown plantation wood for energy, the IPCC 2007 report did not specifically address the use of industrial roundwood for energy. Instead, it highlighted forest-related fossil fuel displacement opportunities from using forest residues for fuel and using wood to displace more fossil fuel-intensive building materials (Nabuurs et al. 2007).

In 2012, IPCC released a report on renewable energy (IPCC 2012). In the chapter dealing with biomass energy, IPCC reiterated the findings of earlier assessments while elaborating on the conditions needed to ensure benefits from biomass energy; namely, sustainable production, efficient utilization, and retaining or increasing land-based carbon stocks.

In the early 2000s, a number of developments led to growing anxiety about the extent to which whole trees might be used for energy and the effects this would have on forests and wood markets. In the United States, there was a flurry of activity on Renewable Portfolio Standards (RPS), with 18 states launching or modifying programs between 2004 and 2006 (Wiser et al. 2007). In the European Union (EU), subsidies were already in place to encourage the use of renewable energy, including biomass (International Energy Association 2014), and in 2009 an EU-wide program was established under the Renewable Energy Directive (RED), requiring 20% of the energy consumed in the EU to be renewable by 2020 (EU 2009). As the RED was being crafted, the European paper industry conducted a study

> demonstrat[ing] that achieving the target of generating 20% of Europe's energy from renewable sources by 2020…would create a shortfall in the supply of wood from EU forests (Confederation of European Paper Industries 2007, p. 2)

As the EU demand for wood fuel increased, some of the supply came from outside of the EU. Wood pellets were often favored because they ship well and can be burned without major modifications to boilers. Much of the import demand for pellets has been satisfied by North America. Pellet exports to Europe more than doubled from 2011 to 2013, with the US South accounting for 63% of the North American export volume (Wood Resources International 2014). Feedstocks for pellets include not only residual materials but also small roundwood (Spelter and Toth 2009).

Increased interest in liquid biofuels in the United States under the updated Renewable Fuel Standard (40 CFR 80 subpart M) and in the EU under new sustainability criteria (EU 2009) focused attention on the GHG attributes of liquid biofuels. Several researchers pointed out the potential for a considerable carbon debt if the feedstocks for liquid biofuels were produced under conditions that caused large losses in ecosys-

Management and Policy Implications

Wood products and energy resources derived from forests have the potential to play an important and ongoing role in mitigating greenhouse gas (GHG) emissions. The methods used to characterize the mitigation benefits of using biomass for energy, however, are being debated without considering some key insights from published research studies. Many recent proposals for biogenic carbon accounting are based on a narrow analysis of the short-term and direct GHG emissions impacts of using forest biomass. This review suggests that a broader view of forest-based activities is needed. Of particular importance to understanding the emissions impacts of increased use of biomass from forests are fossil fuel substitution effects, markets for wood, causes for ongoing gains and losses in forest area and forest carbon, landowners' motivations, benefits and timing of investments in forestry, and the warming impact of near-term and long-term increases in CO_2 emissions. Studies that consider forest growth dynamics, landowner investment responses, and the warming impacts of biogenic CO_2 over a time horizon consistent with that used for other GHGs reveal low warming impacts from biogenic CO_2 associated with increased use of the types of forest biomass most likely to be used for energy in the United States. Such studies also show, for roundwood in particular, the importance of investment responses in contributing to low net emissions.

tem carbon stocks, e.g., via deforestation (Fargione et al. 2008, Gibbs et al. 2008).

Concerns about deforestation and the prospect of roundwood used as fuel caused a reassessment of the methods used to calculate the carbon benefits of biomass fuels. In 2009, a Policy Forum contribution in the journal *Science* attracted headlines when it suggested that a lack of attention to the timing of benefits from biomass energy, land-use change, and international transfers of carbon had created a "critical climate accounting error" (Searchinger et al. 2009).

In the United States, the search for options to meet RPS led to additional scrutiny of the GHG benefits of forest bioenergy. In 2010, the Manomet Report was released. It was later summarized in the peer-reviewed literature (Walker et al. 2013). Manomet examined a range of scenarios for producing forest bioenergy in Massachusetts. The forest bioenergy systems showed net benefits with lag times ranging from a few years to many decades, depending on the type of biomass used (residuals or roundwood) and the fossil-fuel energy system being displaced. Based on the modeling framework, scenarios, and assumptions used in the analysis, the times required to show net benefits from using whole trees to displace fossil fuel were calculated to be 45–75 years, prompting an Associated Press headline "Wood Worse Polluter Than Coal" (LeBlanc 2010).

The studies that have captured headlines, however, represent only a small fraction of the work that has been, and is still being, performed to understand the GHG impacts of using biomass for energy. Many of the studies now being done represent elaborations on the initial findings from the work in the 1990s in that they focus on direct carbon impacts and show net GHG benefits from using forest-derived biomass for energy, with the time required to see benefits ranging from zero years to many decades and sometimes longer (e.g., see McKechnie et al. 2011, Lippke et al. 2012, Mitchell et al. 2012, Zanchi et al. 2012, Agostini et al. 2013, Lamars and Junginger 2013).

Most recently, IPCC reviewed the current state of knowledge on biomass energy in the IPCC Fifth Assessment Report (Smith et al. 2014). This material expands on the findings of the Fourth Assessment Report by examining the many factors that affect the net GHG benefits of biomass energy. It states that

> Bioenergy could play a critical role for climate change mitigation, if conversion of high carbon density ecosystems (forests, grasslands and peatlands) is avoided and best-practice land management is implemented (Smith et al. 2014, p. 6)

Well-established findings regarding the timing of benefits from forest-based energy are reiterated in the report in a statement indicating that

> …in the specific case of existing forests that may continue to grow if not used for bioenergy, some studies employing counterfactual baselines show that forest bioenergy systems can temporarily have higher cumulative CO_2 emissions than a fossil reference system… (Smith et al. 2014, p. 89)

The report highlights the low life cycle GHG emissions associated with biomass energy produced from residues and fast-growing tree species. Also noted are (1) the importance of combined biomass energy and carbon storage and sequestration systems for meeting long-term emissions targets, (2) the life cycle GHG benefits of sustainably produced long-lived wood products, (3) the variable timing in forest-based mitigation, and (4) the need for active forest management, especially in developed countries where the forest carbon sink is declining as forests mature. In discussing the role of market forces, IPCC notes that

> growing markets for tree products can provide incentives for maintaining or increasing forest stocks and land covers, and improving forest health through management (Smith et al. 2014, p. 93)

At the same time, the report and annex note the need for attention to a range of issues to ensure the overall sustainability of forest-based mitigation activities. Among these issues are competition for land, water availability, biodiversity, and land tenure and land-use rights (Smith et al. 2014).

Whereas much of the research since the 1990s extends the life cycle and carbon debt analyses to new materials and scenarios, some of the more recent research uses a broader, integrated assessment approach that includes market dynamics. This allows insights into the factors contributing to the size and timing of benefits from forest bioenergy. Relying on both old and new research, we have identified four important insights related to the GHG impacts of forest-based energy systems. These insights are not intended to replace the many studies that have been performed on this topic but rather to assist in their interpretation. These four insights speak to (1) the consensus on long-term GHG benefits associated with using sustainably produced wood (meaning, in this context, wood produced under conditions ensuring sustained yields while land remains in forest), (2) the importance of wood markets in achieving near-term and long-term GHG benefits from forests, (3) the need to evaluate carbon debt in a framework that also considers drivers of projected peak global temperatures, and (4) the need to evaluate biogenic CO_2 emissions using metrics that are consistent with those used for other GHGs and that reflect forest and market dynamics. Several of these insights involve timing of benefits and emissions. Regarding the discussion of timing in the following material, "near term" means a few decades or less and includes immediate impacts, whereas "long-term" generally means longer than a few decades. Not all of these insights are recent, but all are important to understanding the factors affecting GHG emissions impacts associated with increased use of forest biomass for energy and interpreting studies that attempt to characterize these impacts.

Research Insight 1

As long as land remains in forest, long-term carbon mitigation benefits are derived from sustainably managed working forests that provide an ongoing output of wood and other biomass to produce long-lived products and bioenergy, displacing GHG-intensive alternatives.

Although the *timing* of benefits from substituting sustainably produced forest-based fuels and products for more GHG-intensive alternatives is sometimes debated (Biomass Energy Resource Center [BERC] 2012, Mitchell et al. 2012, Walker et al. 2013), the fact that these ultimate benefits exist is not. Agreement on this issue is based on an extensive body of research, dating at least to the mid-1990s (Schlamadinger and Marland 1996, Marland and Schlamadinger 1997), and reinforced by many more recent studies and reviews. Two types of substitution effects have been the most researched: forest-based energy as a substitute for fossil fuels (Nabuurs et al. 2007, Kirkinen et al. 2008, Nicholls et al. 2008, Bauen et al. 2009, Cherubini et al. 2009, Jones et al. 2010, Malmsheimer et al. 2011, McKechnie et al. 2011, Daigneault et al. 2012, Gaudreault et al. 2012, IPCC 2012, Lippke et al. 2012, Mitchell et al. 2012, Nepal et al. 2012, 2014, Repo et al. 2012, Zanchi et al. 2012, Agostini et al. 2013, Gaudreault and Miner 2013, Lamars and Junginger 2013, White et al. 2013, Dwivedi et al. 2014, Smith et al. 2014) and the use of wood-based building materials in place of alternatives (Perez-Garcia et al. 2005, Nabuurs et al. 2007, Upton et al. 2008, Oneil and Lippke 2010, Sathre and O'Connor 2010, Malmsheimer et al. 2011, Smith et al. 2014).

In considering the timing of fossil fuel substitution benefits, the International Energy Agency observed that

> Land suitable for producing biomass for energy can also be used for the creation of biospheric carbon sinks. Several factors determine the relative attractiveness of these two options [i.e., creating sinks or producing biomass energy], in particular land productivity, including co-products, and fossil fuel replacement efficiency…. A further influencing factor is the time scale that is used for the evaluation of the carbon reduction potential: a short time scale tends to favor the sink option, while a longer time scale offers larger savings as biomass production is not limited by saturation but can repeatedly (from harvest to harvest) deliver GHG emission reductions by substituting for fossil fuels (Bauen et al. 2009, p. 14)

Scientific consensus on the long-term benefits of managed forests and of sustainably produced forest products and fuels led the IPCC to highlight, as noted above, the sustained GHG mitigation benefits from using sustainably produced forest biomass (Nabuurs et al. 2007, p. 543). More recently, this conclusion has been echoed in the findings of Ryan et al. (2010). Based on an extensive literature review, they concluded that

> strategies that combine increased use of forest products to offset fossil fuel use (such as use of forest biomass energy and substitution [of wood products for steel and concrete building products]), in conjunction with increasing carbon storage on forested landscapes, are likely to produce the most sustainable forest carbon benefits (Ryan et al. 2010, p. 14)

Research Insight 2

The demand for wood keeps land in forest, provides incentives for expanding forests and improving forest productivity, and supports investments in sustainable forest management that can help offset the forest carbon impacts of increased demand.

Historical Context. The impacts of forest harvesting on carbon emissions are important yet counterintuitive. The history of US forests shows that increased demand can be met without reducing forested area or forest carbon stocks. The amount of forested area in the United States has been essentially constant since 1900 (Alvarez 2007, Smith et al. 2009) (Figure 1). Over this period, land use has not been static, but losses of forest in one place have been offset by increases elsewhere. Where forest has been lost, it has been due primarily to suburban and urban development (Smith et al. 2009).

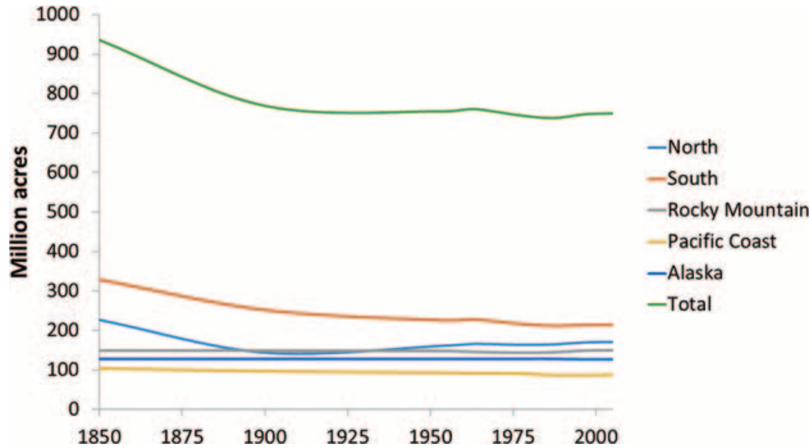

Figure 1. Forest area in the United States has been stable since about 1900. (Data from Alvarez 2007.)

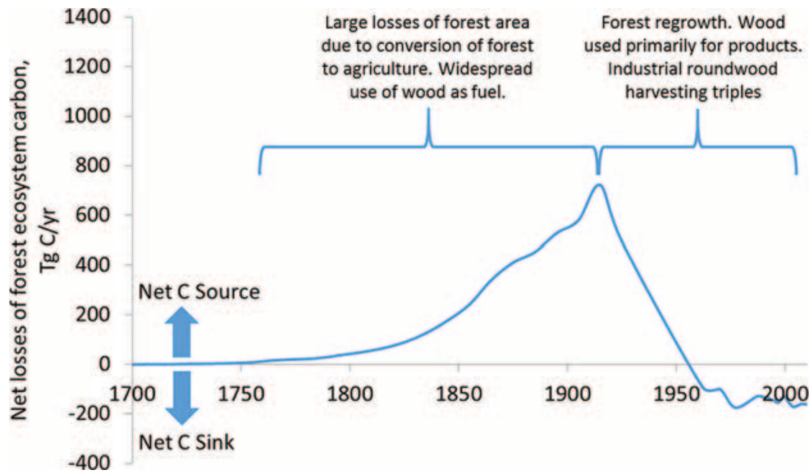

Figure 2. US forest ecosystems are now a net sink for carbon, having rebounded from over a century of deforestation.[2]

Forests in the United States currently function as a carbon sink (Figure 2). This is attributable to a stable forest area for more than 100 years and a long history of net forest growth in excess of mortality and removals at least since 1952 (Birdsey et al. 2006, Smith et al. 2009). Strong markets for forest products, evidenced by a near tripling of industrial roundwood harvests between 1900 and 1998, are also a factor as are improvements in forest management practices (Ince 2000, Birdsey and Lewis 2003, Birdsey et al. 2006).

Currently the net carbon sink in the United States is large enough to offset 14–15% of total annual CO_2 emissions in the United States (US Environmental Protection Agency [EPA] 2013). Approximately 91% of this sink is attributable to increases in carbon stored in forest ecosystems, and the remaining 9% is due to increased quantities of carbon stored in products in use (e.g., wood used in construction) and in landfills (US EPA 2013). Land clearing before the early 1900s and subsequent forest regrowth coupled with improved forest management and increased output of harvested wood products (resulting in increases in carbon stocks in products-in-use and in landfills) created the US forest carbon sink (Figure 2). Although forest carbon stocks have increased considerably since 1950s, it is unlikely that forest area or forest carbon stocks will return to precolonial levels due to the need for productive agricultural land and urban expansion.

Threats to Forest Area and Carbon Stocks. Looking ahead, the strength of the US forest carbon sink is expected to decline

Table 1. Land-use choice elasticities for the United States: percent change in probability of land changing (a) from forest to other land use categories and (b) from other land use categories to forest given a 1% change in net returns in the final use.

(a) Land starting in forest but ending in another category				(b) Land starting in a different category but ending in forest			
Final land use	1982–1987	1987–1992	1992–1997	Starting land use	1982–1987	1987–1992	1992–1997
Crops	0.21* (0.07)	0.28* (0.05)	0.29* (0.06)	Crops	0.88* (0.05)	0.75* (0.07)	0.31* (0.04)
Pasture	0.10 (0.06)	0.04 (0.09)	−0.01 (0.06)	Pasture	0.22* (0.03)	0.08 (0.05)	0.00 (0.03)
Range	0.28 (0.23)	−0.56 (3.46)	0.23 (0.33)	Range	0.08 (0.36)	0.03 (0.42)	0.13 (0.91)
Urban	0.23* (0.02)	0.30* (0.07)	0.80* (0.06)	Urban	Essentially no land converted to forest		

Adapted from Table 3 in Lubowski et al. (2008). Standard errors are in parentheses.
* Statistical significance at the 1% level. Effects were also tested at the 5% level, but this did not reveal any additional statistically significant effects.

as forests growing on previously cleared land mature and forested land is lost to development and other uses (Birdsey et al. 2006, USDA 2012). In Europe, there are already signs that the forest sink is becoming weaker (Nabuurs et al. 2013). Because of urbanization trends in the US South through 2060,

> total forest losses are forecasted to range from 11 million to 23 million acres, depending on the rate of population growth and the future of timber markets—low population growth with strong timber markets would yield the smallest losses. At 7–13 percent of current forest area, these losses would still equal nearly all the forests in Kentucky or South Carolina at the low end of the range, and nearly all the forests in Georgia or Alabama at the high end (Wear and Greis 2012, p. 23)

Climate change may increase damage caused to forests (Joyce et al. 2014) by wildfire (US Department of Agriculture/US Department of the Interior/National Association of State Foresters 2009, Anderegg et al. 2013), insect outbreaks (Bentz et al. 2010), and hurricanes (McNulty 2002, Negrón-Juárez et al. 2010). The net sink could also be affected by changes in wood production and landowner responses to changes in demand for wood (Hardie et al. 2000, Lubowski et al. 2008, Abt et al. 2010, 2012, Daigneault et al. 2012, Nepal et al. 2012, 2014, USDA Forest Service 2012, Wear and Greis 2012, Joyce et al. 2014) as well as public policies (Perez-Garcia et al. 2001, Beach et al. 2005, Langpap 2006, Zobrist and Lippke 2007, Zhang and Flick 2010, Van Deusen et al. 2012, Joyce et al. 2014). Continued increases in atmospheric CO_2 and other plant nutrients will probably increase forest growth and associated CO_2 removals from the atmosphere, although there is considerable uncertainty regarding the factors influencing this growth response (e.g., as regards nitrogen availability and genetic limitations) (Luo et al. 2004, Reich et al. 2006, Denman et al. 2007, Norby et al. 2010).

Table 2. Despite the large increases in harvesting, carbon stocks on private softwood timberland in the US South have remained essentially constant, due to expanded acreage of planted pine and improved management practices.

	Harvesting, carbon stocks, and planted acreage on privately owned softwood timberland in the South		
Year	Harvest (10^6 ft^3)*	Carbon (10^6 metric tons)†	Planted pine (acres)‡
1952	2,840	923	1.8
1999	5,522	883	32
Change, 1952–1999 (%)	+94	−4	+1,678

* Data are from Adams et al. (2006).
† 1999 values were assumed to be equal to the 1997 values shown in Mickler et al. (2004) for privately owned longleaf-slash, loblolly-shortleaf, and miscellaneous conifer forest. Values for 1952 were back-calculated from data in Mickler et al. (2004) showing the mean annual metric ton/year changes for carbon on each of these areas from 1953 to 1997.
‡ Data are from Fox et al. (2004).

The Role of Markets in Protecting Forest Area and Forest Carbon. Against this backdrop of concerns regarding forest area and loss of forest carbon stocks, research shows that demand for wood in the United States results in investments in forestry that help to prevent loss of forest and incentivize afforestation (Hardie et al. 2000, Lubowski et al. 2008). Forest area loss in the United States is caused primarily by urbanization and development, not demand for wood (Lubowski et al. 2008, USDA Forest Service 2012, Wear and Greis 2012). In the face of pressures to convert land to other uses, increased wood demand in the United States can slow the loss of forested area (USDA Forest Service 2012, Nepal et al. 2014). Empirical studies examining the economic factors affecting land-use change (e.g., Table 1) indicate that changes in economic returns to cropland and forestry affect the movement of land between these two uses (Lubowski et al. 2008). Table 1 illustrates, using land-use elasticities, how changes in net economic returns to different land-use categories result in more or less land in forest. Land constantly moves into and out of forest cover. Approximately 4.6% of the forest on nonfederal land in the lower 48 states in 1982 was converted to other uses by 1997. Over the same period, conversion of nonforest land to forest offset these losses by an amount equivalent to about 5.7% of the 1982 forest area (not considering federal land), resulting in a small net gain in forest area (based on data in Lubowski et al. 2008).

The benefits of investments in working forests due to markets for wood vary regionally and are especially clear in the southern United States (Nepal et al. 2014), where the amount of softwood harvested from private pine forests almost doubled between the early 1950s and the late 1990s (Adams et al. 2006) while carbon stocks in these forests remained essentially constant (Mickler et al. 2004) (Table 2). This was made possible, in part, by investments in working forests that greatly increased productivity and expanded the planted pine area from 1.8 million acres in the early 1950s to almost 32 million acres in the late 1990s. Some of this occurred on land not previously forested, but trends in natural pine acreage suggest that much of the planting of pine took place on land converted from natural pine (Fox et al. 2004, Wear and Greis 2012). Whereas the carbon implications of this conversion are relatively well understood, the ecological attributes of planted forests are a matter of ongoing study and debate (e.g., see Loehle et al. 2009, Evans et al. 2013b).

Market-related impacts have been the focus of a number of recent studies, some looking at a range of factors affecting land use and some looking specifically at the projected effects of an increase in demand for forest biomass for energy. A study of factors affecting land use in the US South found that "forestland share responds positively to increases in pine stumpage prices and negatively to increases in timber production costs" (Hardie et al. 2000, p. 670). Another study of land-use change in the United States found that "…the rise in timber net returns [was] the most important factor driving the increase in forest areas between 1982 and 1997" (Lubowski et al. 2008, p. 545). A study of the impacts of increased wood demand for bioenergy in Alabama, Florida and Georgia found that

> …in scenarios in which prices increase, there is more timberland area than occurs under the baseline scenario without bioenergy demand. This, in turn, leads to a higher level of carbon sequestration in the standing forest than occurs with the baseline scenario (Abt et al. 2012, p. 536)

Another study of land-use impacts attributable to increased use of wood for energy found that "[h]igher timber prices lead to new investments in forestry, and in particular in new forestlands in the unconstrained land scenario" (Daigneault et al. 2012, p. 5669). This same study projected that, as a result of increased wood demand for energy, US forest area could expand by 4 to 8.6 million acres by 2015 and 11.9 to 26.9 million acres by 2030 (Daigneault et al. 2012).

A recent assessment by the USDA Forest Service predicted losses in US forest area through 2060 due primarily to urbanization, noting that

> [f]orest area losses are most pronounced under…scenarios, in which population- and income-driven urbanization consumes rural lands at the highest rate. The acceleration of forest loss for [an historical bioenergy demand scenario] relative to [a comparable high bioenergy demand scenario] reflects the role that strong wood products markets can have on retaining or even expanding forestland in parts of the United States (USDA Forest Service 2012, p. 59)

The connection between markets for wood and keeping land in forest is also evident in an assessment of the global patterns of deforestation:

> in general, the data show that the global regions with the highest levels of industrial timber harvest and forest product output are also regions with the lowest rates of deforestation (Ince 2010, p. 32)

Such information led IPCC to observe that

> the anticipation of future bioenergy markets may promote optimized forest management practices or afforestation of marginal land areas to establish managed plantations, so contributing to increased forest carbon stocks. Rather than leading to wide-scale loss of forestlands, growing markets for tree products can provide incentives for maintaining or increasing forest stocks and land cover, and improving forest health through management (Smith et al. 2014)

Although these and other studies provide strong evidence that increased demand for wood leads to increased forested area, the impacts on forest carbon can vary, especially over periods of decades. For instance, the USDA Forest Service (2012) assessment discussed above found that at the end of the study period (2060) projected carbon stocks were likely to be lower in the high biomass demand scenario even while the forest area was likely to be greater. This was due to increased harvesting and more intensive forest management.

Other studies, however, have found that where the investment response to increased demand is strong, it can increase both forest area and forest carbon stocks, especially where investments are made in anticipation of increased demand (Daigneault et al. 2012, Sedjo and Tian 2012, Nepal et al. 2014). Market response varies by region and is especially strong in the US South, where one study found that the investment response would be adequate to offset essentially all of the projected drawdown in forest carbon stocks associated with more than a 4-fold increase in the amounts of forest-derived energy (Nepal et al. 2014). The investment response and the associated carbon impacts can also be affected by policies that constrain market participation (Latta et al. 2013). These varying findings highlight the need for those studying forest bioenergy to consider the many complex ways in which market forces impact forest-related activities and investments.

Landowner responses to increased demand and the carbon impacts of those responses are affected by many things besides markets. This is especially true in the case of small private nonindustrial forest owners, now more commonly referred to as family forest owners (Butler 2008). These landowners supply more than one-half of the wood harvested in the United States (Haynes et al. 2007). A robust body of research shows that these landowners respond to price signals, public policies, and various incentives and disincentives to producing wood. For instance, a meta-analysis of forest management practices of family forest owners found that among the drivers typically studied, policy variables are most likely to be identified as drivers of landowner behavior, followed by plot/resource conditions, owners' characteristics, and market drivers, although the differences in the importance of the various drivers are small (Beach et al. 2005). This and many other studies make it clear that in addition to markets, public policies and other nonmarket factors are important drivers of landowner behavior (Perez-Garcia et al. 2001, Langpap 2006, Zobrist and Lippke 2007, Zhang and Flick 2010, Joshi and Mehmood 2011, Miller et al. 2012, Van Deusen et al. 2012, Aguilar et al. 2013, Becker et al. 2013, Latta et al. 2013, Rozance and Rabotyagov 2014). Research on factors that influence landowner behavior suggests substantial potential for affecting landowner behavior in ways that could have important carbon implications. Policies that provide incentives for landowners to expand forest area, make forests more productive, and store more carbon could have important carbon benefits. On the other hand, policies that increase transaction costs to landowners or devalue forest biomass could have negative carbon consequences, by reducing incentives for investments in working forests, reducing biomass supplies, limiting afforestation activities, and leading to increased conversion of forests to other land uses. The potential for landowner response to disincentives could be especially important in the case of materials collected for energy due to their low value, as revealed in a survey of loggers and landowners in North Carolina (Fielding et al. 2012).

Markets and landowner behavior are often ignored in studies of the carbon impacts of increased use of forest-derived bioenergy, even though many studies have shown that landowners are responsive to forest policies, incentives, and disincentives. Although there are uncertainties associated with studies that provide an integrated analysis of the dynamic interactions between forests, other land uses, and markets for traditional forest products as well as biomass energy, these integrated assessments provide important insights into the carbon implications of different policies. There are many examples of such integrated assessments (see Abt et al. 2010, 2012, Ince et al. 2011, Daigneault et al. 2012, Nepal et al. 2012, 2014, Sedjo and Tian 2012, USDA 2012, Abt and Abt 2013, Latta et al. 2013, White et al. 2013, Nepal et al. 2014).

Research Insight 3

Although forest bioenergy systems sometimes produce near-term increases in CO_2, they typically result in lower cumulative CO_2 emissions over time, and cumulative CO_2 emissions, according to the IPCC, are the best predictor of future peak global temperatures.

The time required for biomass energy systems to show net GHG benefits relative to that for alternative systems has received considerable attention (Vaughan et al. 2009, Cherubini et al. 2011, McKechnie et al. 2011, BERC 2012, Mitchell et al. 2012, US EPA 2012, Agostini et al. 2013, Helin et al. 2013, Lamars and Junginger 2013, Walker et al. 2013). Some of the focus on timing stems from concern about "tipping points" and "abrupt changes."

IPCC defines a tipping point as "a hypothesized critical threshold when global or regional climate changes from one stable state to another stable state" (IPCC 2013, p. 1463). In its most recent analysis, IPCC reports that

> A small number of studies using simplified models find evidence for global-scale 'tipping points'; however, there is no evidence for global-scale tipping points in any of the most comprehensive models evaluated to date in studies of climate evolution in the 21st century. There is evidence for threshold behavior in certain aspects of the climate system, such as ocean circulation and ice sheets, on multi-centennial-to-millennial timescales. There are also arguments for the existence of regional tipping points, most notably in the Arctic although aspects of this are contested (IPCC 2013, p. 129)

IPCC defines an "abrupt change" as

> a large-scale change in the climate system that takes place over a few decades or less, persists (or is anticipated to persist) for at least a few decades, and causes substantial disruptions in human and natural systems (IPCC 2013, p. 1448)

In summarizing the research on abrupt changes IPCC concludes that

> Several components or phenomena in the climate system could potentially exhibit abrupt or nonlinear changes, and some are known to have done so in the past.… For some events, there is information on potential consequences, but in general there is low confidence and little consensus on the likelihood of such events over the 21st century (IPCC 2013, p. 1033)

Whereas the science on tipping points and abrupt changes continues to advance, a consensus has developed on the importance of *cumulative* CO_2 emissions as a predictor of peak global temperatures (Allen et al. 2009, Meinshausen et al. 2009, IPCC 2013). For instance, IPCC (2013) indicates that although there is evidence that short-lived, non-CO_2 GHGs such as aerosols and methane may have near-term temperature impacts within several decades, CO_2 appears to be different:

> taking into account the available information from multiple lines of evidence (observations, models and process understanding), the near linear relationship between cumulative CO_2 emissions and peak global mean temperature is well established in the literature and robust for cumulative total CO_2 emissions up to about 2000 PgC [petagrams of carbon]. It is consistent with the relationship inferred from past cumulative CO_2 emissions and observed warming, is supported by process understanding of the carbon cycle and global energy balance, and emerges as a robust result from the entire hierarchy of models (IPCC 2013, p. 102)

IPCC suggests that for warming to be limited to 2° C at the time net CO_2 emissions are reduced to zero, total cumulative emissions from all anthropogenic sources over the entire industrial era would need to be limited to about 1 trillion tonnes of carbon (IPCC 2013, p. 927). As of 2011, cumulative anthropogenic emissions of CO_2 during the industrial era were estimated to be 555 billion tonnes of carbon (IPCC 2013, p. 467).

IPCC specifically addresses the issue of emissions timing in the context of cumulative emissions, noting that

> The concept of cumulative carbon also implies that higher initial emissions can be compensated by a faster decline in emissions later or by negative emissions. However, in the real world short-term and long-term goals are not independent and mitigation rates are limited by economic constraints and existing infrastructure (IPCC 2013, p. 1113)

Thus, IPCC's concern with higher CO_2 emissions in the near-term reflects concern about what these near-term increases mean about the likelihood of meeting cumulative CO_2 emissions targets in the longer term, especially given economic and infrastructure constraints to making large reductions in the future if emissions are allowed to increase unabated in the near term. In the words of one of the studies cited by IPCC in its most recent assessment

> … the relationship between cumulative emissions and peak warming is remarkably insensitive to the emission pathway (timing of emissions or peak emission rate). Hence policy targets based on limiting cumulative emissions of carbon dioxide are likely to be more robust to scientific uncertainty than emission-rate or concentration targets (Allen et al. 2009, p. 1163)

Elsewhere, IPCC's Fifth Assessment Report addresses the "relationship between short-term action and long-term targets" (Bruckner et al. 2014, p. 67). The IPCC report states that

> Unlike short-lived species (e.g., CH_4 [methane], CO [carbon monoxide], NOx [nitrogen oxides], and SO_2 [sulfur dioxide]) for which stable concentrations are associated with stable emissions, stable concentrations of CO_2 ultimately in the long-term require net emissions to decline to zero. Two important implications follow from this observation. First, it is cumulative emissions over the entire century that to a first approximation determines the CO_2 concentration at the end of the century, and therefore no individual year's emissions are critical…. Second, minimization of global social cost implies an immediate initiation of global emissions mitigation, relative to a reference, no-climate-policy scenario…. (Bruckner et al. 2014, p. 67–68)

The IPCC text reveals that the primary concern about not meeting near-term limits is related to "lock-in" of high GHG technology and infrastructure that fail to accomplish any significant change in terms of the low-carbon energy share and therefore fail to reduce long-term cumulative emissions (Bruckner et al. 2014, p. 68). IPCC's concern about lock-in is focused on conventional fossil fuel energy systems, not energy systems that yield significant long-term reductions in cumulative emissions, such as energy systems supplied with forest biomass from land that remains in forest. Indeed, bioenergy is among the "low-carbon primary energy" sources included in IPCC's energy sector transformation pathways (Clark et al. 2014, p. 40).

These various findings are directly relevant to the discussions about the timing of emissions and benefits associated with the use of forest-derived fuels. When systems using forest biomass are compared with those using fossil fuels for energy, forest-based energy systems can sometimes show higher near-term emissions of CO_2, incurring a carbon debt (Fargione et al. 2008, Gibbs et al. 2008, Jones et al. 2010, McKechnie et al. 2011, Mitchell et al. 2012, US EPA 2012, Zanchi et al. 2012, Agostini et al. 2013, Lamars and Junginger 2013). As discussed above, however, in the case of forest-derived fuels, as long as wood-producing land remains in forest, this short-term increase in emissions is reversed in essentially all cases, and in the long-term, the use of forest-based systems provides lower cumulative CO_2 emissions than fossil fuel-based alternatives (see Research Insight 1). Research Insight 4 (below) elaborates on the times required to achieve lower cumulative emissions from forest biomass fuels. Given the consensus on the importance of reducing long-term cumulative emissions, near-term increases in CO_2 emissions should not be judged in isolation. Forest bioenergy systems may sometimes produce higher near-term CO_2 emissions, but they typically produce lower cumulative CO_2 emissions, making their climate impact fundamentally different from that of fossil fuel systems that increase near-term CO_2 emissions without reducing cumulative emissions.

Research Insight 4

The net warming impacts of biogenic CO_2 should be assessed using a framework that is consistent with that used for other GHGs and, especially in the case of biogenic CO_2 from roundwood, reflects the effects of market-induced investments and forest growth dynamics. When assessed in such a framework, the types of forest-derived biomass likely to be used for energy in the US typically have low (sometimes less than zero) warming impacts.

The GHG impacts of forest-derived energy should be calculated with a metric for biogenic CO_2 that is consistent with the approach used for other GHGs. The impacts of other GHGs are commonly examined on the basis of global warming potentials (GWPs) derived from the 100-year time-integrated global mean radiative forcing of a pulse emission (Forster et al. 2007). Radiative forcing is

> the change in the…radiative flux (expressed in W m^{-2}) at the tropopause or top of atmosphere due to a change in an external driver of climate change, such as…a change in the concentration of carbon dioxide… (IPCC 2013, p. 1460)

Cumulative radiative forcing from a pulse emission of a GHG is determined by calculating the radiative forcing each year, attributable to the amount of GHG remaining in the atmosphere, and summing the annual values over a period, often 100 years. A 100-year GWP for a GHG is equal to the cumulative radiative forcing of a pulse emission of the GHG over 100 years divided by the cumulative radiative forcing of a pulse emission of CO_2 over 100 years. GHGs with higher cumulative radiative forcing than CO_2 have GWPs greater than 1, whereas those with cumulative radiative forcing less than CO_2 will have GPWs less than 1. The impacts of biogenic CO_2 can likewise be characterized by taking net CO_2 fluxes, including removal of CO_2 from the atmosphere by growing trees, and cumulative radiative forcing into account. Several approaches for accomplishing this have been suggested.

Table 3. Development of 100-year global warming potentials for biogenic CO_2 using methods comparable to those used for other GHGs.

	Cherubini et al. (2011) approach*	Helin et al. (2013) approach*
20-yr GWP for biomass energy CO_2 emission	0.54	0.85
100-yr GWP for biomass energy CO_2 emission	0.12	0.26

Two approaches for calculating global warming potentials for biogenic CO_2 (not considering investment-related responses to increased demand) are shown. The example is based on increased production of bioenergy from planted loblolly pine on a 20-year rotation (based on growth curves developed from planted pine data in Smith et al. 2006).
* Calculations are contained in Supplemental Spreadsheet S1.

Cherubini et al. (2011), for instance, suggest using GWP_{bio} which is a global warming potential for biogenic CO_2. It is comparable to GWPs for other gases in that it reflects the net radiative forcing associated with the release of a pulse emission of a GHG, in this case, biogenic CO_2, relative to a pulse emission of fossil fuel CO_2. It differs from other GWPs only in that it also considers the removal of a pulse emission of biogenic CO_2 from the atmosphere by regrowth of the harvested vegetation. This metric characterizes the emissions from the system in isolation, not considering foregone carbon storage and, in the case of trees, foregone sequestration attributable to tree growth. The Cherubini et al. (2011) GWP for biogenic CO_2 will be equal to 1.0 if there is no regrowth to offset the emissions, between 0.0 and 1.0 if a portion of the biogenic CO_2 is removed from the atmosphere by regrowth, and 0.0 if all of the CO_2 is removed via regrowth instantaneously after harvest.

Helin et al. (2013) suggested an approach that differs from that of Cherubini et al. (2011) in that it also considers foregone storage and sequestration, i.e., the removals of CO_2 from the atmosphere and incremental carbon storage that would have occurred had the biomass not been used. As a result, GWPs for biogenic CO_2 developed using the Helin et al. (2013) approach are larger than those developed using the Cherubini et al. (2011) approach. At a given point in time, the Helin et al. (2013) GWP for biogenic CO_2 will be equal to 1.0 if the amount of CO_2 removed by regrowth after harvest is exactly the same as the foregone CO_2 removals attributable to additional growth that would have occurred had the wood not been harvested. It will be greater than 1.0 if the CO_2 removed by regrowth over time is less than the foregone CO_2 removals attributable to the additional growth lost by harvesting and less than 1.0 if the CO_2 removed by regrowth over time is greater than the foregone CO_2 removals.

The Cherubini et al. (2011) approach is appropriate if the objective is to characterize the biogenic CO_2 emissions from the biomass energy system in isolation, without considering foregone carbon storage and sequestration. The approach of Helin et al. (2013) is appropriate where the objective is to assess the net change in biogenic CO_2 emissions associated with an increase in the use of biomass for energy relative to a scenario where this energy demand is met without increasing the use of biomass.[S] Table 3 compares results from the two approaches when applied to planted loblolly pine on a 20-year rotation. The results from both methods and the differences between them vary from one situation to another, depending primarily on growth rates.

In either case, if these metrics are developed for 100-year periods, they can be used in studies in which other gases are characterized using 100-year GWPs, recognizing that

[S] Supplementary data are available with this article at http://dx.doi.org/10.5849/jof.14-009.

the investment response is not accounted for. Increasingly, however, the impacts of biogenic CO_2 are being judged over periods of less than 100 years in analyses in which other GHGs are examined using 100-year GWPs. Using a time horizon of less than 100 years to judge the significance of emissions of biogenic CO_2 implies that wood energy emissions are more damaging than fossil fuel-derived CO_2 and other GHGs, which has no scientific basis. If the time horizon for judging impacts from biogenic CO_2 is less than 100 years, then logically it should be less than 100 years for all GHGs. In other words, consistent time horizons should be used for judging the radiative forcing of all GHGs. IPCC has developed 20-year GWPs (Forster et al. 2007, IPCC 2013), which can be used to judge the cumulative radiative forcing of a pulse emission of many GHGs over a 20-year period. Because these values are static, however, their use can be problematic in systems involving more than pulse emissions of CO_2 at time zero (e.g., see Levasseur et al. 2010, Gaudreault and Miner 2013). For example, because methane has a relatively short lifetime in the atmosphere, methane leaks from natural gas systems exert most of their warming impacts in the years immediately after their release (Alvarez et al. 2012). This dynamic process is important to understanding the timing of impacts involving methane, but it is ignored when using GWPs because GWPs represent cumulative warming impacts over a specific period, usually 100 years. In such cases, dynamic modeling of radiative forcing may be necessary, as suggested by Levasseur et al. (2010), Cherubini et al. (2011), and Helin et al. (2013).

Insights into the GHG attributes of biomass energy can be gained by using the metrics described above. Although these metrics address the direct impacts of using biomass, they miss potentially important indirect impacts, such as market-related impacts on forestry, including investment responses that expand forest area or increase forest productivity, as described in Research Insight 2 above, and the impacts of natural disturbances that might be affected by a decision to increase forest output. The inclusion of impacts related to market-induced investments that expand forest area and improve forest management provides a more robust assessment of the effects of using forest biomass (Daigneault et al. 2012, Galik and Abt 2012). If indirect market-related effects, such as landowner responses to increased demand, are excluded, one can overlook carbon impacts associated with private landowner responses to increased demand that in most cases tend to increase supplies of forest biomass, improve forest management, and offset carbon losses associated with increased harvesting (Daigneault et al. 2012, Nepal et al. 2014).

With an understanding of the importance of using consistent metrics and consideration of the effects of the investment response, it is possible to examine the GHG impacts associated with increased use of specific types of forest-derived biomass. This is done in the following sections addressing logging residue, roundwood from private land, wood from federal land in the West, mill residues, and large trees.

Logging Residue from Private and Public Land. To sustain ecological functions and site productivity, it may be appropriate to retain a portion of logging residues on site, as recognized by various biomass harvesting guidelines (Evans et al. 2013a). Within these constraints, using some of these residues to displace fossil fuel reduces atmospheric GHGs (Lippke et al. 2012, Zanchi et al. 2012, Smith et al. 2014). To estimate the CO_2 reductions associated with increased use of logging residues and their timing, one compares a system in which increased amounts of logging residues are used for energy and an alternative system in which they are not. In cases in which the alternative to using these residues for energy is burning them on site, as is the practice in many US locations, the carbon returns to the atmosphere at the same time for the two systems, so the net impact on biogenic carbon emissions is zero or less than zero if methane and nitrous oxide emissions from pile burning are higher than those from boilers (Jones et al. 2010). In these cases, the benefits of using residues for energy are observed immediately. Where the alternative to using logging residues for energy is leaving them on site to decay, net emissions of biogenic CO_2 may be temporarily higher for the case in which residues are used for energy and substitution benefits may be delayed for a period that depends primarily on the expected decay rate of forest residues (Lamars and Junginger 2013). Although decay half-lives of 100 years or longer have been reported, the majority of United States logging residue half-lives are less than 50 years and generally less than 20 years under warm conditions, with hardwoods generally decomposing more rapidly than softwoods (Zimmerman 2004, Radtke et al. 2009, Russell et al. 2014).

With use of the biogenic CO_2 impact metric developed by Helin et al. (2013) and two assumptions (that slash would have been left on site to decay, with an avoided decay half-life of 20 or 50 years, had it not been burned for energy, and that carbon released by decomposition enters the atmosphere as CO_2), the estimated 100-year GWPs for biogenic CO_2 from increased use of logging slash for energy are 0.30 and 0.56, respectively, not considering avoided fossil fuel emissions (calculations are shown in Supplemental Spreadsheet S2), meaning that the biogenic CO_2 from these sources exerts 30 and 56% of the net radiative forcing of fossil fuel CO_2 over 100 years. In circumstances representative of the eastern United States, where most logging residues are produced and residue decay half-lives are generally 20 years or less (Zimmerman 2004, Russell et al. 2014), the use of forest residues that otherwise would have been left to decay typically accomplishes net GHG benefits within a decade when displacing coal-based electricity and within two decades when displacing natural gas-based electricity, even though wood energy systems usually have somewhat lower efficiencies than fossil fuel systems (Zanchi et al. 2012, Lamars and Junginger 2013, Walker et al. 2013). Longer times to see net benefits can occur in situations in which residues are normally left to decay, and the decay rates are lower (McKechnie et al. 2011, Repo et al. 2012, Lamars and Junginger 2013). Residue decay rates appear to be lower in parts of the Northwest than in the eastern United States, and burning of logging residues on-site is most common in the Northwest, followed by the North, the Southwest, and the South (Cleaves et al. 1999, Zimmerman 2004).

Roundwood from Private Land. The biogenic CO_2 metric developed by Helin et al. (2013), described above, yields an estimated 100-year GWP for biogenic CO_2 from roundwood produced from forests evaluated by the Manomet Center (Walker et al. 2013) and used in accordance with Massachusetts biomass regulations,[3] of 0.68 (calculations are contained in Supplemental Spreadsheet S2), meaning that this biogenic CO_2 has 68% of the cumulative 100-year radiative forcing of fossil fuel CO_2. Lower GWPs are likely where the forest is faster growing (e.g., see Table 3 for a case involving 20-year rotation loblolly pine). Although the GWPs for biogenic CO_2 from old or slow-growing forest are likely to be higher (Marland and Schlamadinger 1997), it is important to consider the extent to which such trees are likely to be used for energy, a topic discussed in more detail below.

In many cases, to correctly characterize net emissions associated with using roundwood, indirect impacts related to market-induced investments must also be analyzed (e.g., see Daigneault et al. 2012, Galik and Abt 2012, Nepal et al. 2014). This can be a complex undertaking, however, and most studies ignore these indirect impacts and instead assume that forest management within a supply area will not change in response to increased demand for forest biomass; i.e., management practices, the amount of forested area, and the impacts of natural disturbances do not change (e.g., see McKechnie et al. 2011, BERC 2012, Mitchell et al. 2012, Zanchi et al. 2012, Walker et al. 2013). Historical evidence indicates, however, that increased demand for roundwood for products and energy from sustainably managed private forestland will increase revenue to landowners and often affect carbon stocks as landowners respond by retaining or expanding forest and intensifying forest management (Fox et al. 2004, Lubowski et al. 2008, Daigneault et al. 2012, Nepal et al. 2014). Because of the time needed to produce wood, studies examining landowner responses to increased demand for wood often look at the role of anticipatory investments made in expectation of future demand (Sedjo and Tian 2012). These anticipatory investments help prevent forest carbon stocks from declining, especially on private land, as the historical record in the southern United States demonstrates (Table 2) and recent research reinforces (Nepal et al. 2014). Empirical studies show that landowners have converted agricultural land to forest and refrained from converting forestland to other uses in response to higher revenues from forestland (Hardie et al. 2000, Lubowski et al. 2008).

The largest demand for wood biomass for energy from private lands is likely to occur where the investment response to increased demand for wood has been the strongest—the US South (Nepal et al. 2014). Research suggests that the impact of the investment response on forest carbon stocks is less significant in other regions of the country (Nepal et al. 2014). In the Pacific Northwest, another major wood-producing region, landownership patterns and economic forces, among other factors, have limited the extent to which investments in forestry result in increased forest area. In this region, essentially all land-use change has involved converting undeveloped agricultural and forestland into developed uses (Lettman et al. 2002, Bradley et al. 2007). Family forest owners in areas with strong development pressures in the Northwest and elsewhere are relatively unlikely to keep land in forest simply for the income from wood production (e.g., see Table 1 and Rozance and Rabotyagov 2014). In places with less development pressure such as eastern Washington state, however, higher forest values can lower the likelihood of development (Rozance and Rabotyagov 2014).

The impacts of roundwood harvesting on soil carbon are complex. Soil carbon can be affected by management treatments that impact site productivity or cause soil disturbance (Johnson et al. 2002, Jandl et al. 2007, Nabuurs et al. 2007). Conversion of nonforested land to forest generally increases soil carbon stocks, although decreases have been observed in some grassland ecosystems with very high initial soil carbon and in situations where peatland has been drained to allow plantations to be established (Nabuurs et al. 2007). However, in temperate forests, as long as land remains in forest, mineral soil carbon levels generally remain stable even though forest floor carbon levels may be affected by harvesting with the effects being related to changes in species composition and soil taxonomic order (Nave et al. 2010).

A general increase in the demand for biomass for energy could cause a reduction in biomass carbon stocks on the land if the new demand induces forest landowners to convert lands to produce annual or short-rotation biomass energy crops (White et al. 2013). Healthy markets for higher value-added forest products (e.g., lumber, panels, and paper), produced in parallel with wood biomass coproducts (e.g., thinnings and logging slash) that are sources of energy, can help minimize such losses. Munsell and Fox (2010), for instance, found that

> providing biomass for energy from pine plantations on cutover sites is most profitable when intensive management is used to produce a mixture of traditional forest products and biomass for energy

The economic benefit of integrated production of traditional products that help make removal of small diameter trees and other low-value biomass feasible for bioenergy or other purposes is also highlighted in works by Barbour et al. (2008), Evans (2008), Lowell et al. (2008), Nicholls et al. (2008), Abbas et al. (2013), and Aguilar et al. (2013). Although not examined in this review, the effects of changes in forest management on impacts of natural disturbances can also affect the net benefits of biomass energy, although important information gaps in some situations limit the extent to which this can be dealt with quantitatively (Gunn et al. 2014).

Obtaining GHG benefits from use of trees for energy requires retaining wood-producing land in forest. The timing of net CO_2 emissions from forest bioenergy systems and the time required to see net GHG benefits from increased use of trees for energy will depend on many factors including tree size and growth rate, displaced fossil fuel, life cycle GHG emissions from transport, processing and other activities, and the rate at which demand increases and the investment response to that demand. Where large trees are involved, the investment response is weak or not considered and the calculations include foregone sequestration, the net radiative forcing associated with biogenic CO_2 attributable to increased use of trees for energy may be close to that of fossil fuel CO_2 (e.g., estimated 100-year GWP of $\geq$0.9 for slow-growing trees), and in these cases many decades may be required to see net GHG benefits (McKechnie et al. 2011, Mitchell et al. 2012, Lamars and Junginger 2013, Walker et al. 2013). Where tree growth rates are relatively fast and there is a history of landowners responding to increased demand by expanding forest area and improving forest productivity, the net radiative forcing of biogenic CO_2 is far less than that of fossil fuel CO_2. For instance, the estimated 100-year GWP of biogenic CO_2 resulting from increased use of 20-year rotation pine is 0.26 if foregone sequestration is included in the calculation and 0.12 if it is not (Table 3). This does not consider the investment response to increased demand that would be expected to reduce this further. Where the investment response is strong and fast-growing trees are involved, the increased production of energy from a mix of forest biomass that includes trees can produce net GHG benefits within a decade or two, with shorter times being associated with studies that assume investments occur in anticipation of increased demand (Abt et al. 2010, 2012, Daigneault et al. 2012, Nepal et al. 2014).

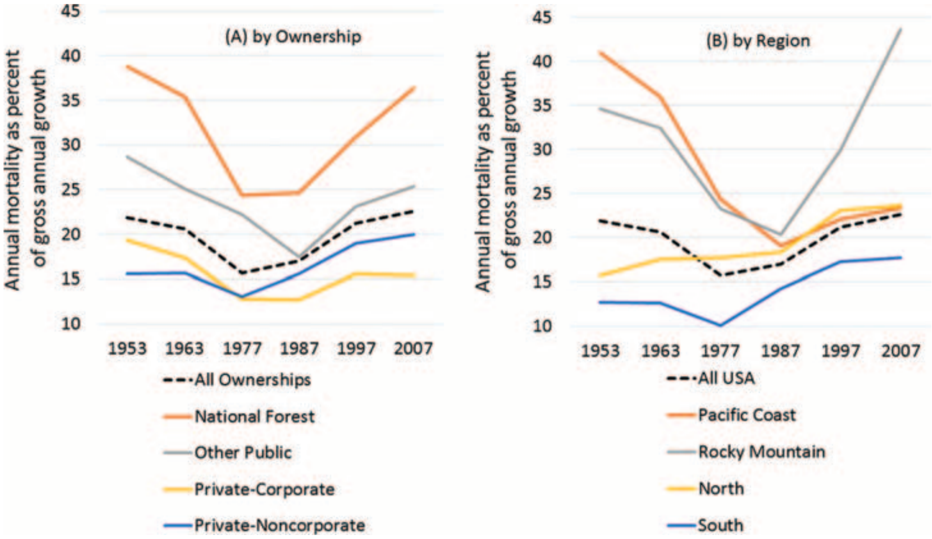

Figure 3. Mortality trends in US forests highlight the need for forest restoration on (A) national forest and other public ownerships, especially (B) in the Rocky Mountain region. The differences in mortality between public and private forests, as well as the regional differences, are probably related, in part, to different levels of investment in working forests which are known to be greater on private land and in the South (data from Smith et al. 2009).[4]

Wood from Federal Land in the West

Tree mortality rates in public forests are higher than the nationwide average (Figure 3A), especially in the Rocky Mountain region (Figure 3B). In the West, the wildfire threat involves more land than is threatened by insects, disease, and development (Kline et al. 2013). As a result of the increasing mortality trend since 1987 and the corresponding potential for large-scale intensive wildfires, there is considerable interest in improving forest conditions on public lands in the West by fuel reduction treatments, particularly thinning (Finkral and Evans 2008, Ager et al. 2010, Coleman et al. 2010, Jones et al. 2010, Oneil and Lippke 2010, Reinhardt and Holsinger 2010).

Because economic returns are a lower priority for public land, revenues from periodic harvesting in public forests are unlikely to influence either the scheduling of treatments to enhance growth or the conversion of forest to other land-use categories. However, markets for timber and biomass may facilitate or accelerate silvicultural treatments to produce conditions that reduce the risks of catastrophic fire, disease, or insect outbreaks. Although thinning treatments for fuel reduction reduce carbon on the landscape, they also reduce the threats of carbon loss from fire, especially in fire-prone areas, as well as from insects and disease (Hurteau et al. 2008, Hurteau and North 2009). The net carbon benefits of fuel reduction treatments are highly site-specific with benefits unlikely in forests that are not fire prone (Mitchell et al. 2009, Ager et al. 2010, Coleman et al. 2010, Jones et al. 2010, Oneil and Lippke 2010). The net carbon benefits of fuel reduction treatments can be enhanced by using the wood to make products or fuels that reduce fossil fuel-related carbon emissions via substitution effects (see references in Research Insight 1). For purposes of carbon accounting, however, if the goal of treatments is viewed as forest restoration then any wood that is not used for timber products can be viewed as logging residue and would have net GHG emissions similar to those of other logging residues, as discussed above.

Mill Residues. The use of biomass manufacturing residues for energy results in very low net impacts from biogenic GHGs. Almost all of these materials are now used for energy. This was not always the case, however. At one time, for instance, teepee or beehive burners were used to dispose of much of the residues produced by saw mills, but by 1992, or earlier, these became "essentially obsolete" (US EPA 1992). History suggests, therefore, that if mill residues were not used for energy, most of these materials would be wastes that would be either incinerated, in which case the atmosphere would see the same biogenic CO_2 emissions as if the material had been burned for energy, or disposed in landfills.

If the alternative to using mill residues for energy is disposing of these materials in landfills, then the net impact of burning for energy on biogenic emissions, in terms of warming (i.e., CO_2 equivalents), can actually be less than zero because of the warming potency of the methane generated in landfills (Gaudreault and Miner 2013). Methane is a far more potent GHG than CO_2, exerting 28 times the radiative forcing of CO_2 over 100 years (IPCC 2013). In some cases, the methane emissions from piles used to store residuals at bioenergy facilities may also need to be considered, and these may reduce the net benefits of using these materials for energy. These emissions appear to be small, however, compared with those from landfills (Pier and Kelly 1997). When assessment is extended to consider all GHGs and fossil fuel substitution, the overall benefits of using manufacturing residuals for energy are large and become evident in short periods (Gaudreault and Miner 2013, Lamars and Junginger 2013).

Large Trees and Bioenergy. Mature forests are important reservoirs of stored carbon (Mackey et al. 2013), and carbon debts associated with using large trees as sources of energy would be expected to be large and require considerable time to overcome (Mitchell et al. 2012). The use of large trees (i.e., those of a size suitable for use as sawtimber), however, is unlikely because of market and economic considerations. Studies have shown that even with substantial demand for forest biomass and conservative supply assumptions, larger and higher valued materials, such as sawtimber, are not likely to be used for energy even if smaller diameter trees, such as those used for pulpwood, may be diverted to the biomass energy market (Abt et al. 2010, Ince et al. 2011).

For instance, a recent forward-looking assessment examining a range of wood energy scenarios included one wherein the use of wood for energy increased by a factor of 3.7 by 2060 to a level at which wood fuel feedstock consumption was slightly greater than all other commercial uses. Even under these conditions, projected sawtimber prices remained above those of nonsawtimber. Only when the use of wood for energy far exceeded all other uses did projected sawtimber and nonsawtimber prices converge (USDA Forest Service 2012, p. 26 and Figure 83). Looking specifically at the southern United States, studies have found that increased demand for biomass energy is unlikely to increase the price of small diameter roundwood to even half that of typical prices for sawtimber (Abt et al. 2012, Abt and Abt 2013). Many factors combine to keep prices for biomass used for energy well below prices that would lead to harvest of sawtimber-size material for energy, including the small roundwood supply, the potential for increased utilization of residues, the complementary effect of forest biomass harvest on other products, and the probable supply and management response to higher prices. These factors make it very unlikely that wood suitable as sawtimber, in the southern United States or elsewhere, would be harvested for energy as long as a market for sawtimber is available.

Summary and Conclusions

It has been known for at least 25 years that there are substantial long-term CO_2 reduction benefits from forest bioenergy as long as wood-producing land remains in forest. It has also been known that, in some cases, increased use of forest bioenergy can result in higher near-term CO_2 emissions, causing delays in seeing net benefits. The length of this delay varies considerably, based on physical and biophysical factors, differences in biomass characteristics, and reference conditions.

Recently, researchers have examined forest bioenergy in a broader, integrated framework that also addresses market impacts. Based both on empirical data and modeling, these studies have determined that increased demand often leads to investments in forestry that increase forest area and incentivize improvements in forest management that, depending on circumstances, can also increase forest carbon stocks. This investment response has been found to be especially important in places such as the US South, where economic returns to land have been shown to directly affect gains and losses in forest area.

Recently researchers have also examined metrics for characterizing the impacts from biogenic CO_2 in ways consistent with those used for other GHGs. The research has produced several metrics for calculating 100-year GWPs for biogenic CO_2, reflecting net cumulative radiative forcing. The use of these metrics reveals that as long as land remains in forest, emissions of biogenic CO_2 associated with increased production of forest bioenergy have smaller, and often far smaller, net warming impacts than comparable emissions of CO_2 from fossil fuel.

The logic behind the selection of time horizons for judging the impacts of biogenic CO_2 has been clarified by IPCC, which has found that in the case of CO_2, there is a near linear relationship between cumulative emissions and projected peak global temperatures. The emphasis on near-term reductions in CO_2 emissions primarily reflects concern about the "lock-in" of technologies that do not help reduce cumulative emissions of CO_2 in the longer term. The near-term increases in CO_2 emissions sometimes caused by forest bioenergy are associated with energy systems that *reduce* cumulative emissions in the longer term, something that clearly differentiates them from fossil fuel systems that increase near-term emissions without reducing cumulative emissions.

The research described above provides the basis for a number of generalizations about the GHG impacts to be expected from increased use of different types of forest biomass for energy:

- Under almost all circumstances, the use of biomass residuals from manufacturing has been found to yield low to negative emissions of biogenic GHGs in relatively short times and essentially instantaneous benefits when displacing fossil fuels.
- Net biogenic CO_2 emissions from increased use of forest harvest residues for energy are highly dependent on whether harvest residues would have been burned in the forest or left to decay. When the alternative to using residuals for energy is burning them in the forest, the net emissions of biogenic CO_2 associated with using forest residues for energy are essentially zero, and benefits from displacing fossil fuel occur instantaneously. If the alternative is leaving the material to decay under conditions representative of the eastern United States, the net warming caused by biogenic CO_2 emissions is likely to be in the range of one-third to one-half of that associated with a pulse emission of an equal amount of fossil fuel CO_2. Under these conditions, GHG benefits from using forest residues to produce electricity are generally observed in less than a decade or two. Where decay rates are slower, as appears to be the case in the Pacific Northwest, longer times may be required to see net benefits.
- Considering the economic disincentives to using large trees for energy and the regions expected to provide wood for energy, the types of roundwood anticipated to be used to meet increased demand for bioenergy in the United States are likely to be associated with relatively low net biogenic CO_2 emissions impacts. Where growth rates are relatively high and the investment response is strong, net GHG benefits from increased use of trees for energy can be realized within a decade or two, depending on the fossil fuel being displaced and the timing of the investment response. Where tree growth is slow and the investment response is lacking, many decades may be required to see the net benefits from using roundwood for energy.

Legal, policy, and practical considerations, beyond the scope of this science- and economics-focused review, will affect how biogenic CO_2 is characterized under different laws, regulations, and forest management guidelines. Those working to understand the GHG impacts of increased reliance on forest bioenergy, however, need to consider the following: the relatively low net cumulative emissions impacts associated with increased use of the types of forest biomass most likely to be used for energy in the United States; the importance of the investment response in contributing to these low net emissions; and the possible effect on these investments of policies that attach an emissions liability to biogenic CO_2.

Endnotes

1. IPCC was established by the United Nations Environment Programme and the World Meteorological Organization in 1988 to provide scientific analysis of climate change, including its potential environmental and socioeconomic impacts. IPCC relies on a network of government-nominated scientists from around the world, operates under the auspices of the United Nations, and has a Secretariat located in Geneva, Switzerland.
2. The following sources of data were used to construct Figure 2. The history of changes in US forest carbon stocks is modeled after a similar figure in Birdsey et al. (2006): plotted data do not include below ground carbon; 1630–1930 data and conversion factors are from Birdsey et al. (2006), plotted data are 10-year averages; 1931–1961 data are interpolated; 1962–1987 data are from Birdsey and Heath (1995); and 1990–2010 data are from Table A240, Annex Section 3.13 in US EPA (2013). The statement on industrial roundwood harvesting is based on Ince (2000). Estimates developed before the beginning of the U.S. forest inventory program in the 1930s are likely to be less reliable than those developed since then. Data are available on request.
3. Information about the Massachusetts renewable energy portfolio standards is available online at www.mass.gov/eea/energy-utilities-clean-tech/renewable-energy/biomass/renewable-portfolio-standard-biomass-policy.html; last accessed May 24, 2014.

4. The following are notes on the data in Figure 3. Although annual mortality is often expressed as a percentage of growing stock volume, mortality as a percentage of gross growth is a more useful health indicator (O'Laughlin 1996) because stands can be considered healthy when growth and mortality are appropriately balanced (Norris et al. 1993). Missing from the literature are judgments on appropriate balance, which would vary by stand type and age. However, the mortality/growth rate depicted in Figure 3 is correlated to stand density (significant at the 0.01 level); thus, within any ownership class or region, higher mortality/growth rates can be attributed to denser stands (O'Laughlin and Cook 2003).

Literature Cited

Abbas, D., R. Handler, D. Dykstra, B. Hartsough, and P. Lautala. 2013. Cost analysis of forest biomass supply chain logistics. *J. For.* 111:271–281.

Abt, K.L., R.C. Abt, and C. Galik. 2012. Effect of bioenergy demands and supply response on markets, carbon, and land use. *For. Sci.* 58: 523–539.

Abt, R.C., and K.L. Abt. 2013. Potential impact of bioenergy demand on the sustainability of the southern forest resource. *J. Sustain. For.* 32:175–194.

Abt, R.C., C.S. Galik, and J.D. Henderson. 2010. *The near-term market and greenhouse gas implications of forest biomass utilization in the southeastern United States.* Nicholas School of the Environment, Working Pap. CCPP 10-01, Duke University, Durham, NC. 34 p.

Adams, D.M., R.W. Haynes, and A.J. Daigneault. 2006. *Estimated timber harvest by US region and ownership, 1950–2000.* USDA For. Serv., Gen. Tech. Rep. PNW-GTR-659, Pacific Northwest Research Station, Portland, OR. 64 p.

Ager, A.A., M.A. Finney, A. McMahan, and J. Cathcart. 2010. Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. *Nat. Hazards Earth Syst. Sci.* 10: 2515–2526.

Agostini, A., J. Giuntoli, and A. Boulamanti. 2013. *Carbon accounting for forest bioenergy: Conclusions and recommendations from a critical literature review.* Joint Research Centre Institute for Energy and Transport, Scientific and Policy Rep. EUR 25354, European Commission, Luxembourg, Belgium. 85 p.

Aguilar, F.X., M.J. Daniel, and L.L. Narine. 2013. Opportunities and challenges to the supply of woody biomass for energy from Missouri nonindustrial privately owned forestlands. *J. For.* 111:249–260.

Allen, M.R., D.J. Frame, C. Huntingford, C.D. Jones, J.A. Lowe, M. Meinshausen, and N. Meinshausen. 2009. Letter: Warming caused by cumulative carbon emissions towards the trillionth tonne. *Nature* 458:1163–1166.

Alvarez, M. 2007. *The state of America's forests.* Society of American Foresters, Bethesda, MD. 68 p.

Alvarez, R.A., S.W. Pacala, J.J. Winebrake, W.L. Chameides, and S.P. Hamburg. 2012. Greater focus needed on methane leakage from natural gas infrastructure. *Proc. Natl. Acad. Sci. USA* 109:6435–6440.

Anderegg, W.R.L., J.M. Kane, and L.D.L. Anderegg. 2013. Consequences of widespread tree mortality triggered by drought and temperature stress. *Nat. Clim. Change* 3:30–36.

Barbour, R.J., X. Zhou, and J.P. Prestemon. 2008. Timber product output implications of a program of mechanical fuel treatments applied on public timberland in the western United States. *For. Policy Econ.* 10:373–385.

Bauen, A., G. Berndes, M. Junginger, M. Londo, and F. Vuille. 2009. *Bioenergy—A sustainable and reliable energy source: A review of status and prospects.* International Energy Agency IEA Bioenergy, Rotorua, New Zealand. 108 p.

Beach, R., S. Pattanayak, J. Yang, B. Murray, and R. Abt. 2005. Economic studies of nonindustrial private forest management: A review and synthesis. *For. Policy Econ.* 7:261–281.

Becker, D., D. Erylimaz, J. Klapperich, and M. Kilgore. 2013. Social availability of residual woody biomass from nonindustrial private woodland owners in Minnesota and Wisconsin. *Biomass Bioenergy* 56:82–91.

Bentz, B.J., J. Régniere, C.J. Fettig, E.M. Hansen, J.L. Hayes, J.A. Hicke, R.G. Kelsey, et al. 2010. Climate change and bark beetles of the western United States and Canada: Direct and indirect effects. *Bioscience* 60:602–613.

Berger, A., B. Palik, A. D'Amato, S. Fraver, J. Bradford, K. Nislow, D. King, and R. Brooks. 2013. Ecological impacts of energy-wood harvests: Lessons from whole-tree harvesting and natural disturbance. *J. For.* 111: 139–153.

Biomass Energy Resource Center. 2012. *Biomass supply and carbon accounting for southeastern forests.* Biomass Energy Resource Center, Montpelier, VT. 123 p. + appendices.

Birdsey, R.A., and L.S. Heath. 1995. Carbon changes in US forests. P. 56–70 in *Productivity of America's forests and climate change*, Joyce, L.A. (ed.). USDA For. Serv., Gen. Tech. Rep. RM-GTR-271, Rocky Mountain Research Station, Fort Collins, CO. 70 p.

Birdsey, R.A., and G.M. Lewis. 2003. Current and historical trends in use, management, and disturbance of US forestlands. P. 15–33 in *The potential of US forest soils to sequester carbon and mitigate the greenhouse effect*, Kimble, J.M., L. Heath, R.A. Birdsey, and R. Lal (eds.). CRC Press, New York.

Birdsey, R., K. Pregitzer, and A. Lucier. 2006. Forest carbon management in the United States: 1600–2100. *J. Environ. Qual.* 35: 1461–1469.

Bradley, G., A. Erickson, A. Robbins, G. Smith, L. Malone, L. Rogers, M. Connor. 2007. Forest land conversion in Washington State. P. 238–302 in *Future of Washington's forest and forest industries study*, Edmonds, R., B. Boyle, B. Lippke, J. Perez-Garcia, I. Eastin, and G. Bradley (principal investigators). College of Forest Resources, University of Washington, Seattle, WA. 406 p.

Bruckner, T., I.A. Bashmakov, Y. Mulugetta, H. Chum, A.D. Navarro, J. Edmonds, A. Faaij, et al. 2014. Energy systems. Chapter 7 in *Climate change 2014: Mitigation of climate change.* Working Group III contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Edenhofer, O., et al. (eds.). Available online at report.mitigation2014.org/drafts/final-draft-postplenary/ipcc_wg3_ar5_final-draft_postplenary_chapter7.pdf; last accessed August 22, 2014.

Butler, B.J. 2008. *Family forest owners of the United States, 2006.* USDA For. Serv., Gen. Tech. Rep. NRS-27, Northern Research Station, Newtown Square, PA. 72 p.

Clark, L., K. Jiang, K. Akimoto, M. Babiker, G. Blanford, K. Fisher, J. Vanden, J.C. Hourcade, V. Krey, et al. 2014. Assessing transformation pathways. Chapter 6 in *Climate change 2014: Mitigation of climate change.* Working Group III contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Edenhofer, O., et al. (eds.). Available online at report.mitigation2014.org/drafts/final-draft-postplenary/ipcc_wg3_ar5_final-draft_postplenary_chapter6.pdf; last accessed August 20, 2014.

Confederation of European Paper Industries. 2007. *Bio-energy and the European pulp and paper industry—An impact assessment: A summary of the conclusions of the report.* Confederation of European Paper Industries, Brussels, Belgium. 4 p.

Cherubini, F., N. Bird, A. Cowie, G. Jungmeier, B. Schlamadinger, and S. Woess-Gallasch. 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. *Res. Conserv. Recycling* 53:434–447.

Cherubini, F., G. Peters, T. Berntsen, A. Strømman, and E. Hertwich. 2011. CO_2 emissions from biomass combustion for bioenergy: Atmospheric decay and contribution to global warming. *Glob. Change Biol. Bioenergy* 3:413–426.

Cleaves, D., T. Haines, and J. Martinez. 1999. Prescribed burning costs: Trends and influences in the National Forest System. P. 277–287 in *Proc. of the Symposium on fire economics, planning, and policy: Bottom lines*, González-Cabán, A., and P. Omi (eds.). USDA For. Serv., Gen. Tech. Rep. PSW-GTR-173, Pacific Southwest Research Station, Albany, CA. 332 p.

Coleman, M., D. Page-Dumroese, J. Archuleta, P. Badger, W. Chung, T. Venn, D. Loeffler, et al. 2010. Can portable pyrolysis units make biomass utilization affordable while using bio-char to enhance soil productivity and sequester carbon? P. 159–168 in *Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proc. of the 2009 National Silviculture Workshop,* Jain, T.B., R. Graham, and J. Sandquist (eds.). USDA For. Serv.,

Proc. RMRS-P-61, Rocky Mountain Research Station, Ogden, UT. 351 p.
DAIGNEAULT, A., B. SOHNGEN, AND R. SEDJO. 2012. Economic approach to assess the forest carbon implications of biomass energy. *Environ. Sci. Technol.* 46:5664–5671.
DENMAN, K.L., G. BRASSEUR, A. CHIDTHAISONG, P. CIAIS, P.M. COX, R.E. DICKINSON, D. HAUGLUSTAINE, ET AL. 2007. Couplings between changes in the climate system and biogeochemistry. Chapter 7 in *Climate change 2007: The physical science basis.* Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller (eds.). Cambridge University Press, Cambridge, UK. 587 p.
DWIVEDI, P., M. KHANNA, R. BAILLIS, AND A. GHILARDI. 2014. Potential greenhouse gas benefits of transatlantic wood pellet trade. *Environ. Res. Lett.* 9(2014):024007. 11 p. Available online at iopscience.iop.org/1748-9326/9/2/024007/pdf/1748-9326_9_2_024007.pdf; last accessed May 13, 2014.
EVANS, A.M. 2008. *Synthesis of knowledge from woody biomass removal case studies: A summary of knowledge from the Joint Fire Science Program.* USDA For. Serv. and Forest Guild, Santa Fe, NM. 39 p.
EVANS, A.M., R.T. PERSCHEL, AND B.A. KITTLER. 2013a. Overview of forest biomass harvesting guidelines. *J. Sustain. For.* 32:89–107.
EVANS, J.M., R.J. FLETCHER JR., J.R.R. ALAVALAPATI, A.L. SMITH, D. GELLER, P. LAL, D. VASUDEV, M. ACEVEDO, J. CALABRIA, AND T. UPADHYAY. 2013b. *Forestry bioenergy in the Southeast United States: Implications for wildlife habitat and biodiversity.* National Wildlife Federation, Merrifield, VA. 275 p.
EUROPEAN UNION. 2009. Directive 2009/28/EC of the European Parliament and of the Council [of the European Union] on the promotion of the use of energy from renewable sources. *Official J. Eur. Union* 5.6.2009. 47 p.
FARGIONE, J., J. HILL, D. TILMAN, S. POLASKY, AND P. HAWTHORNE. 2008. Land clearing and the biofuel carbon debt. *Science* 319:1235–1238.
FIELDING, D., F. CUBBAGE, M.N. PETERSON, D. HAZEL, B. GUGELMANN, AND C. MOORMAN. 2012. Opinions of forest managers, loggers and forest landowners in North Carolina regarding biomass harvesting guidelines. *Int. J. For. Res.* 2012:256141. 15 p. Available online at dx.doi.org/10.1155/2012/256141; last accessed May 13, 2014.
FINKRAL, A.J., AND A.M. EVANS. 2008. Effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. *For. Ecol. Manage.* 255:2743–2750.
FORSTER, P., V. RAMASWAMY, P. ARTAXO, T. BERNTSEN, R. BETTS, D.W. FAHEY, J. HAYWOOD, ET AL. 2007. Changes in atmospheric constituents and in radiative forcing. Chapter 2 in *Climate change 2007: The physical science basis.* Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller (eds.). Cambridge University Press, Cambridge, UK. 234 p.
FOX, T.R., E.J. JOKELA, AND H.L. ALLEN. 2004. The evolution of pine plantation silviculture in the southern United States. P. 63–82 in *Southern forest science: Past, present, and future*, Rauscher, H.M., and K. Johnsen (eds.). USDA For. Serv., Gen. Tech. Rep. SRS-75, Southern Research Station, Asheville, NC. 394 p.
GALIK, C.S., AND R.C. ABT. 2012. The effect of assessment scale and metric selection on the greenhouse gas benefits of woody biomass. *Biomass Bioenergy* 44:1–7.
GAUDREAULT, C., B. MALMBERG, B. UPTON, AND R. MINER. 2012. Life cycle greenhouse gases and non-renewable energy benefits of kraft black liquor recovery. *Biomass Bioenergy* 46: 683–692.
GAUDREAULT, C., AND R. MINER. 2013. *Greenhouse gas and fossil fuel reduction benefits of using biomass manufacturing residues for energy production in forest products manufacturing facilities.* National Council for Air and Stream Improvement, Inc., Tech. Bull. 1016, Research Triangle Park, NC. 74 p.
GIBBS, H., M. JOHNSTON, J. FOLEY, T. HOLLOWAY, C. MONFREDA, N. RAMANKUTTY, AND D. ZAKS. 2008. Carbon payback times for crop-based biofuel expansion in the tropics: The effects of changing yield and technology. *Environ. Res. Lett.* 3(2008):034001. 10 p. Available online at iopscience.iop.org/1748-9326/3/3/034001/pdf/1748-9326_3_3_034001.pdf; last accessed May 13, 2014.
GUNN, J.S., M.J. DUCEY, AND A.A. WHITMAN. 2014. Late-successional and old-growth forest carbon temporal dynamics in the Northern Forest (Northeastern USA). *For. Ecol. Manage.* 312:40–46.
HARDIE, I., P. PARKS, P. GOTTLEIB, AND D. WEAR. 2000. Responsiveness of rural and urban land uses to land rent determinants in the US South. *Land Econ.* 76:659–673.
HAYNES, R.W., D.M. ADAMS, R.J. ALIG, P.J. INCE, J.R. MILLS, AND X. ZHOU. 2007. *The 2005 RPA timber assessment update.* USDA For. Serv., Gen. Tech. Rep. PNW-GTR-699, Pacific Northwest Research Station, Portland, OR. 212 p.
HELIN, T., L. SOKKA, S. SOIMAKALLIO, K. PINGOUD, AND T. PAJULA. 2013. Approaches for inclusion of forest carbon cycle in life cycle assessment—A review. *Glob. Change Biol. Bioenergy* 5:475–486.
HURTEAU, M., AND M. NORTH. 2009. Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. *Front. Ecol. Environ.* 7:409–414.
HURTEAU, M.D., G.W KOCH, AND B.A. HUNGATE. 2008. Carbon protection and fire risk reduction: Toward a full accounting of forest carbon offsets. *Front. Ecol. Environ.* 6:493–498.
INCE, P. 2000. *Industrial wood productivity in the United States, 1900–1998.* USDA For. Serv., Res. Note FPL-RN-0272, Madison, WI. 14 p.
INCE, P.J. 2010. Global sustainable timber supply and demand. P. 29–41 in *Sustainable development in the forest products industry*, Rowell, R.M., F. Caldeira, and J.K. Rowell (eds.). Universidade Fernando Pessoa, Portal, Portugal. 281 p.
INCE, P.J., A.D. KRAMP, K.E. SKOG, D. YOO, AND V.A. SAMPLE. 2011. Modeling future US forest sector market and trade impacts of expansion in wood energy consumption. *J. For. Econ.* 17: 142–156.
INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE (IPCC). 1990. *Climate change: The IPCC response strategies.* Contribution of Working Group III of the Intergovernmental Panel on Climate Change, Bernthal, F.M. (chmn.). Intergovernmental Panel on Climate Change, Geneva, Switzerland. 269 p.
INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE (IPCC). 2012. *Renewable energy sources and climate change mitigation.* Special report prepared by Working Group III of the Intergovernmental Panel on Climate Change, Edenhofer, O., R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, et al. (eds.). Cambridge University Press, Cambridge, UK. 1075 p.
INTERGOVERNMENTAL PANEL ON CLIMATE CHANGE (IPCC). 2013. *Climate change 2013: The physical science basis.* Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, et al. (eds.). Cambridge University Press, Cambridge, UK and New York. 1535 p.
INTERNATIONAL ENERGY ASSOCIATION. 2014. *United Kingdom renewables obligation 2002, last amended 2009.* IEA/IRENA Joint Policies and Measures database. Available online at www.iea.org/policiesandmeasures/renewableenergy/?country=United Kingdom; last accessed May 15, 2014.
JANDL, R., M. LINDNER, L. VESTERDAL, B. BAUWENS, R. BARITZ, F. HAGEDORN, D. JOHNSON, ET AL. 2007. How strongly can forest management influence soil carbon sequestration? *Geoderma* 137:253–268.
JOHNSON, D., J. KNOEPP, W. SWANK, J. SHAN, L. MORRIS, D. VANLEAR, AND P. KAPELUCK. 2002. Effects of forest management on soil carbon: Results of some long-term sampling studies. *Environ. Pollut.* 116(Suppl. 1):S201–S208.
JONES, G., D. LOEFFLER, D. CALKIN, AND W. CHUNG. 2010. Forest treatment residues for thermal energy compared with disposal by onsite burning: Emissions and energy return. *Biomass Bioenergy* 34:737–746.
JOSHI, O., AND S. MEHMOOD. 2011. Factors affecting nonindustrial private forest landowners' willingness to supply woody biomass for bioenergy. *Biomass Bioenergy* 35:186–192.
JOYCE, L.A., S.W. RUNNING, D.D. BRESHEARS, V.H. DALE, R.W. MALMSHEIMER, R.N. SAMPSON, B. SOHNGEN, AND C.W. WOODALL. 2014. Forests. P. 175–194 in *Climate change impacts in the United States: The third national climate assessment*, Melillo, J.M., T.C. Richmond, and G.W. Yohe (eds.). US Global

Change Research Program, Washington, DC. 829 p.

Kirkinen, J., T. Palosuo, K. Holmgren, and I. Savolainen. 2008. Greenhouse impact due to the use of combustible fuels: Life cycle viewpoint and relative radiative forcing commitment. *Environ. Manage.* 42:458–469.

Kline, J.D., B.K. Kerns, M.A. Day, and R.B. Hammer. 2013. Mapping multiple forest threats in the Northwestern United States. *J. For.* 111:206–213.

Lamars, P., and M. Junginger. 2013. The 'debt' is in the detail: A synthesis of recent temporal forest carbon analyses on woody biomass for energy. *Biofuels Bioproducts Biorefining* 7:373–385.

Langpap, C. 2006. Conservation of endangered species: Can incentives work for private landowners? *Ecol. Econ.* 57:558–572.

Latta, G., J. Baker, R. Beach, S. Rose, and B. McCarl. 2013. A multi-sector intertemporal optimization approach to assess the GHG implications of US forest and agricultural biomass electricity expansion. *J. For. Econ.* 19: 361–383.

LeBlanc, S. 2010. *Mass. study: Wood worse polluter than coal.* Associated Press, Boston, MA, June 11. Available online at www.boston.com/yourtown/cambridge/articles/2010/06/11/mass_study_wood_power_worse_polluter_than_coal/; last accessed May 13, 2014.

Lettman, G., D.L. Azuma, K.R. Birch, A.A. Herstrom, and J.D. Kline. 2002. *Land use change on non-federal land in western Oregon, 1973–2000.* Oregon Department of Forestry, Salem, OR. 48 p.

Levasseur, A., P. Lesage, M. Margni, L. Deschênes, and R. Samson. 2010. Considering time in LCA: Dynamic LCA and its application to global warming impact assessments. *Environ. Sci. Technol.* 44:3169–3174.

Lippke, B., R. Gustafson, R. Venditti, P. Steele, T.A. Volk, E. Oneil, L. Johnson, M.E. Puettmann, and K. Skog. 2012. Comparing life-cycle carbon and energy impacts for biofuel, wood product, and forest management alternatives. *For. Prod. J.* 62:247–257.

Loehle, C., T.B. Wigley, E. Schilling, V. Tatum, J. Beebe, E. Vance, P. Van Deusen, and P. Weatherford. 2009. Achieving conservation goals in managed forests of the Southeastern Coastal Plain. *Environ. Manage.* 44:1136–1148.

Lowell, E.C., D.R. Becker, D. Larson, R. Rummer, and L. Wadleigh. 2008. An integrated approach to evaluating the economic costs of wildfire hazard reduction through wood utilization opportunities in the southwestern United States. *For. Sci.* 54:273–283.

Lubowski, R.N., A.J. Plantinga, and R.N. Stavins. 2008. What drives land-use change in the United States? A national analysis of landowner decisions. *Land Econ.* 84:529–550.

Luo, Y., B. Su, W.S. Currie, J.S. Dukes, A. Finzi, U. Hartwig, B. Hungate, et al. 2004. Progressive nitrogen limitation of ecosystem responses to rising atmospheric carbon dioxide. *Bioscience* 54:731–739.

Mackey, B., I.C. Prentice, W. Steffen, J.I. House, D. Lindenmayer, H. Keith, and S. Berry. 2013. Untangling the confusion around land carbon science and climate change mitigation policy. *Nat. Clim. Change* 3:552–557.

Malmsheimer, R.W., J.L. Bowyer, J.S. Fried, E. Gee, R.L. Izlar, R.A. Miner, I.A. Munn, E. Oneil, and W.C. Stewart. 2011. Managing forests because carbon matters: Integrating energy, products, and land management policy. *J. For.* 109(7S):S7–S50.

Marland, G. 1989. The role of forests in addressing CO_2 greenhouse. P. 199–212 in *Global climate change linkages: Acid rain, air quality, and stratospheric ozone*, White, J.C., W.R. Wagner, and C.N. Beal (eds.). Elsevier, New York. 272 p.

Marland, G., and B. Schlamadinger. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. *Biomass Bioenergy* 13:389–397.

McKechnie, J., S. Colombo, J. Chen, W. Mabee, and H. Maclean. 2011. Forest bioenergy or forest carbon? Assessing the trade-offs in greenhouse gas mitigation with wood-based fuels. *Environ. Sci. Technol.* 2011:789–795.

McNulty, S.G. 2002. Hurricane impacts on US forest carbon sequestration. *Environ. Pollut.* 116:S17–S24.

Meinshausen, M., N. Meinshausen, W. Hare, S.C.B. Raper, K. Frieler, R. Knutti, D.J. Frame, and M.R. Allen. 2009. Letter: Greenhouse-gas emission targets for limiting global warming to 2° C. *Nature* 458:1158–1162.

Mickler, R.A., J.E. Smith, and L.S. Heath. 2004. Forest carbon trends in the southern United States. P. 383–394 in *Southern forest science: Past, present, and future*, Rauscher, H.M., and K. Johnsen (eds.). USDA For. Serv., Gen. Tech. Rep. SRS-75, Southern Research Station, Asheville, NC. 394 p.

Miller, K., S. Snyder, and M. Kilgore. 2012. An assessment of forest landowner interest in selling forest carbon credits in the Lake States, USA. *For. Policy Econ.* 25:113–122.

Mitchell, S.R., M.E. Harmon, and K.E.B. O'Connell. 2009. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. *Ecol. Appl.* 19:643–655.

Mitchell, S.R., M.E. Harmon, and K.E.B. O'Connell. 2012. Carbon debt and carbon sequestration parity in forest bioenergy production. *Glob. Change Biol. Bioenergy* 4:818–827.

Munsell, J., and T. Fox. 2010. An analysis of the feasibility for increasing woody biomass production from pine plantations in the southern United States. *Biomass Bioenergy* 34:1631–1642.

Nabuurs, G.J., O. Masera, K. Andrasko, P. Benitez-Ponce, R. Boer, M. Dutschke, E. Elsiddig, et al. 2007. Forestry. Chapter 9 in *Climate change 2007: Mitigation.* Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Metz, B., O.R. Davidson, P.R. Bosch, R. Dave, and L.A. Meyer (eds.). Cambridge University Press, Cambridge, UK. 584 p.

Nabuurs, G.J., M. Lindner, P.J. Verkerk, K. Gunia, P. Deda, R. Michalak, and G. Grassi. 2013. First signs of carbon sink saturation in European forest biomass. *Nat. Clim. Change* 3:792–796.

Nave, L., E. Vance, C. Swanston, and P. Curtis. 2010. Harvest impacts on soil carbon storage in temperate forests. *For. Ecol. Manage.* 259:857–866.

Negrón-Juárez, R., D.B. Baker, H.C. Zeng, T.K. Henkel, and J.Q. Chambers. 2010. Assessing hurricane-induced tree mortality in US Gulf Coast forest ecosystems. *J. Geophys. Res. Biogeosci.* 115:1–11.

Nepal, P., P.J. Ince, K.E. Skog, and S.J. Chang. 2012. Projection of US forest sector carbon sequestration under US and global timber market and wood energy consumption scenarios, 2010–2060. *Biomass Bioenergy* 45:251–264.

Nepal, P., D. Wear, and K. Skog. 2014. Net change in carbon emissions with increased wood energy use in the United States. *Glob. Change Biol. Bioenergy.* doi: 10.1111/gcbb.12193.

Nicholls, D.L., R.A. Monserud, and D.P. Dykstra. 2008. *A synthesis of biomass utilization for bioenergy production in the western states.* USDA For. Serv., Gen. Tech. Rep. PNW-GTR-753, Pacific Northwest Research Station, Portland, OR. 48 p.

Norby, R.J., J.M. Warren, C.M. Iversen, B.E. Medlyn, and R.E. McMurtrie. 2010. CO_2 enhancement of forest productivity constrained by limited nitrogen availability. *Proc. Natl. Acad. Sci. USA* 107:19368–19373.

Norris, L.A., H. Cortner, M.R. Cutler, S.G. Haines, J.E. Hubbard, M.A. Kerrick, W.B. Kessler, et al. 1993. *Sustaining long-term forest health and productivity.* Task Force Rep. Society of American Foresters, Bethesda, MD. 83 p.

O'Laughlin, J. 1996. Forest ecosystem health assessment: Definition, measurement and management implications. *Ecosyst. Health* 2(1):19–39.

O'Laughlin, J., and P.S. Cook. 2003. Inventory-based forest health indicators: Implications for national forest management. *J. For.* 101(2):11–17.

Oneil, E.E., and B.R. Lippke. 2010. Integrating products, emissions offsets, and wildfire into carbon assessments of inland northwest forests. *Wood Fiber Sci.* 42:144–164.

Perez-Garcia, J.M., J. Edelson, and K. Zobrist. 2001. *New proposed forest practices rules small business economic cost statement.* Washington State Department of Natural Resources, Olympia, WA. 37 p. + appendices.

Perez-Garcia, J., B. Lippke, J. Comnick, and C. Manriquez. 2005. An assessment of carbon pools, storage, and products market substitution using life-cycle results. *Wood Fiber Sci.* 37: 140–148.

Pier, P., and J. Kelly. 1997. Measured and estimated methane and carbon dioxide emissions from sawdust waste in the Tennessee Valley under alternative management strategies. *Bioresour. Technol.* 61:213–220.

RADTKE, P., R. AMATEIS, S. PRISLEY, C. COPENHEAVER, D. CHOJNACKY, J. PITTMAN, AND H. BURKHART. 2009. Modeling production and decay of coarse woody debris in loblolly pine plantations. *For. Ecol. Manage.* 257:790–799.

REICH, P.B., B.A. HUNGATE, AND Y. LUO. 2006. Carbon-nitrogen interactions in terrestrial ecosystems in response to rising atmospheric carbon dioxide. *Annu. Rev. Ecol. Evol. Syst.* 37: 611–636.

REINHARDT, E., AND L. HOLSINGER. 2010. Effects of fuel treatments on carbon-disturbance relationships in forests of the northern Rocky Mountains. *For. Ecol. Manage.* 259:1427–1435.

REPO, A., R. KANKANEN, J. TUOVINEN, R. ANTIKAINEN, M. TUOMI, P. VANHALA, AND J. LISKI. 2012. Forest bioenergy climate impact can be improved by allocating forest residue removal. *GCB Bioenergy* 4:202–212.

ROZANCE, M.A., AND S.S. RABOTYAGOV. 2014. Washington State small forest landowners: Who intends to develop their forestlands and when? *J. For.* doi: http://dx.doi.org/10.5849/jof.12-103.

RUSSELL, M., C. WOODALL, S. FRAVER, A. D'AMATO, G. DOMKE, AND K. SKOG. 2014. Residence times and decay rates of downed woody debris biomass/carbon in eastern US forests. *Ecosystems* 17:765–777.

RYAN, M.G., M.E. HARMON, R.A. BIRDSEY, C.P. GIARDINA, L.S. HEATH, R.A. HOUGHTON, R.B. JACKSON, ET AL. 2010. A synthesis of the science on forests and carbon for US forests. *Issues Ecol.* 13:1–16.

SATHRE, R., AND J. O'CONNOR. 2010. Meta-analysis of greenhouse gas displacement factors of wood product substitution. *Environ. Sci. Policy* 13:104–114.

SCHLAMADINGER, B., AND G. MARLAND. 1996. The role of forest and bioenergy strategies in the global carbon cycle. *Biomass Bioenergy* 10: 275–300.

SEARCHINGER, T., S.P. HAMBURG, J. MELILLO, W. CHAMEIDES, P. HAVLIK, D.M. KAMMEN, G.E. LIKENS, ET AL. 2009. Policy forum: Fixing a critical climate accounting error. *Science* 326: 527–528.

SEDJO, R., AND X. TIAN. 2012. Does wood bioenergy increase carbon stocks in forests? *J. For.* 110:304–311.

SMITH, J., L. HEATH, K. SKOG, AND R. BIRDSEY. 2006. *Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States.* USDA For. Serv., Gen. Tech. Rep. NE-343, Northeastern Research Station, Newtown Square, PA. 216 p.

SMITH, P., M. BUSTAMANTE, H. AHAMMAD, H. CLARK, H. DONG, E.A. ELSIDDIG, H. HABERL, R. HARPER, ET AL. 2014. Agriculture, forestry and other land use (AFOLU). Chapter 11 in *Climate change 2014: Mitigation of climate change.* Working Group III contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Edenhofer, O., et al. (eds.). Available online at report.mitigation2014.org/drafts/final-draft-postplenary/ipcc_wg3_ar5_final-draft_postplenary_chapter11.pdf; last accessed August 22, 2014.

SMITH, W.B., P.D. MILES, C.H. PERRY, AND S.A. PUGH. 2009. *Forest resources of the United States, 2007.* USDA For. Serv., Gen. Tech. Rep. WO-78, Washington, DC. 336 p.

SPELTER, H., AND D. TOTH. 2009. *North America's wood pellet sector*. USDA For. Serv., Res. Pap. FPL-RP-656, Forest Products Laboratory, Madison, WI. 21 p.

UPTON, B., R. MINER, M. SPINNEY, AND L. SMITH. 2008. The greenhouse gas and energy impacts of using wood instead of alternatives in residential construction in the United States. *Biomass Bioenergy* 32:1–10.

USDA FOREST SERVICE. 2012. *Future of America's forest and rangelands*: *Forest Service 2010 Resources Planning Act assessment.* USDA For. Serv., Gen. Tech. Rep. WO-87, Washington, DC. 198 p.

US ENVIRONMENTAL PROTECTION AGENCY. 1992. Emission factors for waste incineration in conical burners without controls. In *Compilation of air pollutant emission factors, AP-42.* 3 p. Available online at www.epa.gov/ttnchie1/ap42/ch02/final/c02s07.pdf; last accessed May 15, 2014.

US ENVIRONMENTAL PROTECTION AGENCY. 2012. *SAB review of EPA's accounting framework for biogenic CO_2 emissions from stationary sources: Report to the EPA administrator.* US Environmental Protection Agency Science Advisory Board, Washington, DC. 49 p. + appendices.

US ENVIRONMENTAL PROTECTION AGENCY. 2013. *Inventory of US greenhouse gas emissions and sinks: 1990–2011.* EPA 430-R-13-001, Washington, DC.

US DEPARTMENT OF AGRICULTURE/US DEPARTMENT OF THE INTERIOR/NATIONAL ASSOCIATION OF STATE FORESTERS. 2009. *Quadrennial fire review 2009.* Washington, DC. 44 p. + appendices.

VAN DEUSEN, P., T. WIGLEY, AND A. LUCIER. 2012. Cumulative effects of constraints on forest management. *J. For.* 110:123–128.

VAUGHAN, N.E., T.M. LENTON, AND J.G. SHEPHERD. 2009. Climate change mitigation: Trade-offs between delay and strength of action required. *Clim. Change* 96:29–43.

WALKER, T., P. CARDELLICHIO, J.S. GUNN, D.S. SAAHD, AND J.M. HAGAN. 2013. Carbon accounting for woody biomass from Massachusetts (USA) managed forests: A framework for determining the temporal impacts of wood biomass energy on atmospheric greenhouse gas levels. *J. Sustain. For.* 32:130–158.

WEAR, D.N., AND J.G. GREIS. 2012. *The Southern Forest Futures Project: Summary report.* USDA For. Serv., Gen. Tech. Rep. SRS-GTR-168, Southern Research Station, Asheville, NC. 54 p.

WHITE, E., G. LATTA, R. ALIG, K. SKOG, AND D. ADAMS. 2013. Biomass production from the US forest and agriculture sectors in support of a renewable electricity standard. *Energy Policy* 58:64–74.

WISER, R., C. NAMOVICZ, M. GIELECKI, AND R. SMITH. 2007. *Renewables portfolio standards: A factual introduction to experience from the United States.* US Department of Energy, Rep. LBNL-63569, Ernest Orlando Lawrence Berkeley National Laboratory, Berkeley, CA. 15 p.

WOOD RESOURCES INTERNATIONAL. 2014. North American wood pellet exports to Europe double in two years. *Biomass Mag.* May 6. Available online at biomassmagazine.com/articles/10311/north-american-wood-pellet-exports-to-europe-double-in-2-years; last accessed May 15, 2014.

ZANCHI, G., N. PENA, AND N. BIRD. 2012. Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel. *Glob. Change Biol. Bioenergy* 4:761–772.

ZHANG, D., AND W.A. FLICK. 2010. Sticks, carrots, and reforestation investment. *Land Econ.* 77:443–456.

ZIMMERMAN, G. 2004. *Dynamics of coarse woody debris in North American forests: A literature review.* National Council for Air and Stream Improvement, Inc., Tech. Bull. No. 877, Research Triangle Park, NC. 55 p.

ZOBRIST, K.W., AND B.R. LIPPKE. 2007. Economic costs of different riparian management regulations in the Pacific Northwest. *West. J. Appl. For.* 22:36–41.