Nicholas Institute for Environmental Policy Solutions
Working Paper
NI WP 13-06
September 2013

An Initial Assessment of the Economics, Carbon Scores, and Market Impacts of Selected Woody Feedstock Biomass Systems

Christopher Galik*
Robert C. Abt**

* Nicholas Institute for Environmental Policy Solutions, Duke University
** Department of Forestry and Environmental Resources, North Carolina State University

Acknowledgments
The authors thank the Electric Power Research Institute for its generous grant in support of this work and Ronald W. Gonzales for his research assistance and technical guidance.

How to cite this report
Christopher Galik and Robert C. Abt. 2013. *An Initial Assessment of the Economics, Carbon Scores, and Market Impacts of Selected Woody Feedstock Biomass Systems* NI WP 13-06. Durham, NC: Duke University.

EXECUTIVE SUMMARY

The combustion of biomass for energy production has received a great deal of attention in both the policy arena and the scientific literature. Nevertheless, the greenhouse gas (GHG) effects of biomass use remain uncertain, and biomass combustion itself remains a controversial undertaking. This analysis identifies trends emerging in the scientific literature on the GHG emission intensity of biomass combustion and considers them in the context of an applied case study: the hypothetical repowering of a coal plant in Albany, Georgia.

The scientific literature review reveals that GHG emissions are influenced by a host of scenario-specific factors, including feedstock type; inputs and activities associated with biomass growth, harvest, and transportation; biomass processing technique; manner of biomass disposal; conversion technology; and indirect effects (e.g., "leakage"). All else equal, accounting approach and GHG neutrality assumptions can each drive research findings. In the case of woody biomass, especially where active markets exist for other uses of forest biomass, a change in accounting approach can take a scenario from poorly performing (high GHG emissions) to marginally beneficial (slight GHG emissions reduction). Alternatively, removing an assumption of GHG neutrality can take a scenario from strongly performing to poorly performing. The relationship among other parameters, such as feedstock growth, processing, and transportation, is more complex, however.

These general findings are borne out in the case study, which indicates that incremental differences are generated by different feedstock, transportation, and disposal assumptions on a hypothetical plant repowering. The influence of these individual factors is greatly outweighed, however, by general accounting approaches, system boundaries, and GHG neutrality assumptions, changes in any of which can yield widely diverging results. The case study shows that removal of certain GHG flows (e.g., displaced coal emissions or changes in forest carbon storage) would vastly alter the GHG emissions associated with plant repowering. Clear, transparent, and standardized assessment processes are therefore critical if studies of different situations are to be comparable and uncertainty about the true GHG effects of biomass combustion is to be reduced.

REVIEW OF THE LIFE-CYCLE ASSESSMENT LITERATURE

The GHG emissions associated with industrial biomass combustion remain the subject of great debate and uncertainty. Some of this uncertainty can be attributed to the many unknowns involved in calculating the true GHG emissions associated with biomass use, such as baseline conditions over time, induced shifts in land use and management, and changing market conditions. Beyond data gaps and measurement error, uncertainty is introduced by the literature itself. Proposed accounting frameworks reflect different assumptions, perspectives, timescales, and system boundaries, thereby yielding divergent results for seemingly similar scenarios. These collective sources of uncertainty are further examined below.

Accounting Frameworks Yield Differences

One theme emerging from the literature is that the approaches for tallying the GHG effects of biomass combustion can drive findings. This is the conclusion of multiple papers appearing in the peer-reviewed literature over the last few years. Galik and Abt (2012b) specifically evaluated the effects of different accounting approaches and reporting perspectives on observed GHG emissions from woody biomass cofiring in the state of Virginia, finding that a single bioenergy scenario resulted in widely different estimates of GHG emissions, depending on the accounting approach (**Figure 1**). Bird, Pena, and Zanchi (2012) also evaluated multiple accounting approaches for bioenergy production and likewise find wide disparities in results despite similar underlying scenarios.

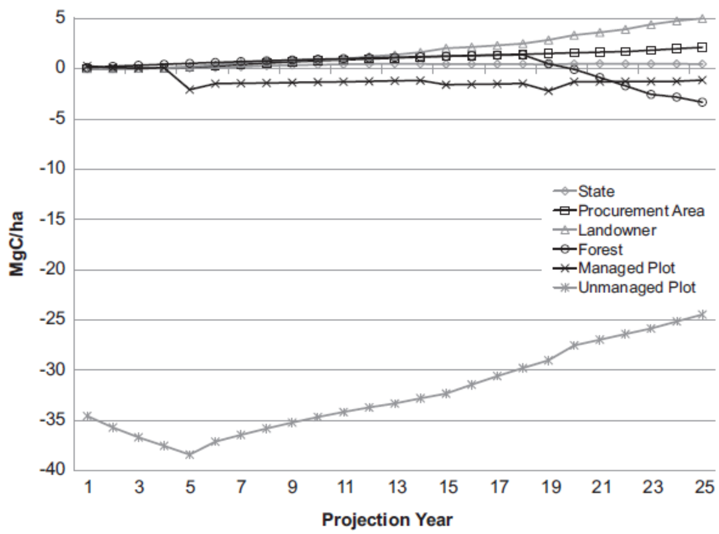

Figure 1. Cumulative greenhouse gas (GHG) balance under each assessment scale. Positive values represent improved GHG performance relative to a baseline, non-cofire scenario. *Source:* Galik and Abt (2012b).

The work by Galik and Abt (2012b) and Bird, Pena, and Zanchi (2012) focus on the role of accounting, generally. Other authors have reviewed the implications of different system boundaries, or what is and what is not included in the life-cycle assessment (LCA). The importance of carefully defining system boundaries dates back more than a decade (Schlamadinger et al. 1997) but is still being confronted. In their review of the LCA literature, Cherubini et al. (2009) discuss the sources of disparity among studies, finding that system boundaries and baseline scenario assumptions, together with other technical and input assumptions, influence results. Cherubini and Strømman (2011) reach a similar conclusion in their review and commentary, as do Djomo, Kasmioui, and Reinhart (2011) in their assessment of bioenergy production from short-rotation woody crops (SRWC).

Assumptions of GHG Neutrality Yield Differences

Like the accounting system used to conduct the LCA, assumptions of GHG neutrality factor heavily into the findings of an analysis. At issue is whether the carbon emitted during combustion is offset over the short run by carbon absorbed during growth. Analyses that assume GHG neutrality—that the difference between carbon absorbed and carbon emitted is minimal—generally exclude carbon emitted during combustion from the LCA. Analyses that do not assume GHG neutrality generally account for growth and combustion phases separately, in some cases even assessing indirect shifts in land use attributable to biomass use. These different perspectives translate into large differences in LCA findings, even for scenarios that are otherwise quite similar. Although examples of both assumptions can be found throughout the LCA literature, relatively recent analyses tend to acknowledge the inherent complexity of biomass carbon storage and emission more so than earlier ones. This trend is hardly universal, however. Some early LCAs devote significant energy to evaluating the GHG consequences of biomass growth and combustion (e.g., Schlamadinger et al. 1995), whereas some recent analyses begin with a blanket assumption of GHG neutrality (e.g., Zhang et al. 2010; Kabir and Kumar 2012).

GHG Neutrality Absolutely Assumed

Despite focusing on a variety of biomass feedstock production systems and combustion pathways, the following analyses all assume GHG neutrality—that is, that the carbon stored during the growth of the feedstock is roughly equal to the carbon emitted during its combustion, allowing biomass emissions to be excluded from study.[1] Depending on the feedstock, production system, and combustion pathway, this assumption can tilt the analysis in favor of biomass use or simplify the analysis while minimally affecting the final results.

Analyses in which GHG assumptions appear to drive results are common in the LCA literature. Gustavsson et al. (1995) consider a wide range of biomass feedstocks and fossil fuel alternatives, assuming GHG neutrality for multiple aspects of the production and conversion process. In doing so, they conclude that biomass can provide for significant CO_2 reductions. Liu et al. (2010) also assume GHG neutrality in their analysis of straw combustion. They cite diesel combustion and processing energy as the only sources of emissions and find a roughly 94% reduction in CO_2 emissions as compared with coal combustion. Relative to fossil fuel use, Zhang et al. (2010) find that use of woody biomass pellets reduces greenhouse gases by up to 91%, NOx by 47%, and SOx by 76%. Their analysis includes a sophisticated analysis of multiple harvest, production, and delivery pathways for both biomass and fossil fuel alternatives, but it is the assumption of biomass GHG neutrality that appears to drive the results.

In other analyses, the assumption of GHG neutrality is a relatively minor component in a more rigorous accounting structure. In several, alternative disposition decisions (i.e., how waste feedstocks might otherwise be disposed of) receive careful attention. In his review of cofiring experiences, Tillman (2000) describes how such decisions affect GHG emission results. Alternative disposition decisions also feature prominently in the calculations performed by Mann and Spath (2001), who present a robust and oft-cited life-cycle analysis of woody biomass cofiring with coal, finding net CO_2 reductions of 2-6% as compared with coal-only combustion. Similarly, Kabir and Kumar (2012), who evaluate multiple biomass feedstock production and consumption pathways, find that agricultural residues perform best from a CO_2 perspective, partly because of the feedstock's alternative disposition as waste.

GHG Neutrality Assumed with Caveats

Other analyses begin with an assumption of biomass GHG neutrality only if some other conditions continue to hold. Baxter (2005), for example, begins with an assumption of carbon neutrality "so long as the farming and forest products industries that produce these residues are conducted in a sustainable manner" (1295). Heller et al. (2004) argue that feedstock growth should be credited to the use through

[1] The studies reviewed here are but a fraction of the life-cycle analyses in the literature. For additional examples of analyses that assume GHG neutrality, see Gold and Tillman (1996), Heinz et al. (2001), Wihersaari (2005), and Jeswani, Gujba, and Azapagic (2011).

which the biomass was generated. Purpose-grown willow is therefore assumed to be GHG neutral, and credit for residues is given to the use for which the harvest occurred (though residues could be given credit for avoided disposal emissions in some situations). Rather than conditioning an assumption, Carpentieri, Corti, and Lombardi (2005) separately examine the influence of growth phase assumptions on the GHG balance of integrated biomass gasification combined-cycle applications, finding that accounting only for a system's process and combustion emissions results in a significantly different GHG outcome than that obtained by also accounting for feedstock growth. A similar parsing of growth and combustion phases is considered in Corti and Lombardi (2004).

No GHG Neutrality Assumed

Life-cycle analyses that do not assume GHG neutrality are generally characterized by separate accounting for biomass growth and combustion phases to estimate the net GHG consequences of biomass use. Qin et al. (2006) begin their assessment of switchgrass combustion by calculating carbon storage over the life of the feedstock, plus additional soil carbon storage, minus any carbon loss due to transportation. In all, they find that both dedicated combustion and cofiring can lead to GHG reductions, but that cofiring switchgrass yields greater GHG reductions per ton of feedstock than dedicated combustion due to the former's higher heat rate. In comparing the GHG effects of different biomass use scenarios over time, Schlamadinger et al. (1995) apply results from previous studies to estimate the temporal effects of forest residue use. They find the potential for short-term increases in relative emissions but also the potential for long-term reductions as displaced fossil emissions eventually outpace forest carbon loss. Keoleian and Volk (2007) find that significant reductions in greenhouse gases, SO_2, and NOx are possible through use of willow biomass but that the direction of change as compared with coal use depends on environmental attributes and feedstock source (e.g., cofiring both willow and willow combined with residues results in GHG reductions, but cofiring only willow achieves greater reductions).

Other Drivers of LCA GHG Conclusions

Accounting approaches, system boundaries, and GHG neutrality assumptions can in and of themselves drive conclusions about the GHG consequences of bioenergy production. As the literature clearly indicates, LCA results can be influenced by a host of other technical parameters and assumptions. Table 1 displays a variety of these drivers cited by a subset of the reviewed studies. In each case, the driver was the subject of a sensitivity analysis or was found to exert a particularly strong influence on GHG results.

Summary

Although Table 1 provides no indication of the magnitude of effects associated with each parameter, it shows the variety of input assumptions that can influence bioenergy GHG life-cycle analyses. Parameters, implicit GHG neutrality assumptions, and accounting approaches vary among the studies so that clear inferences about relative impacts are not possible. However, general trends are observable. All else equal, accounting approach and GHG neutrality assumptions can each drive GHG results. In the case of woody biomass combustion, especially where active markets exist for other uses of forest biomass, research shows that accounting approach can take a scenario from poorly performing (high GHG emissions) to marginally beneficial (slight GHG emissions reduction). Alternatively, removing an assumption of GHG neutrality can take a scenario from strongly performing to poorly performing. The relationship among other parameters is more complex, however. The production of pellets consumes more energy than simply bailing the feedstock but results in a product that is easier to transport. Research specifically focused on the effects of multiple parameters on the GHG score of different bioenergy systems also suggests that carbon price can fundamentally alter production relationships (Schmidt et al. 2010).

Table 1. Partial drivers of variation in bioenergy GHG determinations as reported by the subset of bioenergy LCA studies reviewed here.

Study	Feedstock type[a]	Biomass growth/ harvest[b]	Biomass processing[c]	Biomass disposal[d]	Indirect effects[e]	Transpor-tation[f]	Energy conversion (biomass)[g]	Energy conversion (fossil fuel)[h]
Carpentieri et al. (2005)		X				X	X	
Damen and Faaij (2006)		X	X		X	X		X
Gustavsson et al. (1995)								X
Gustavsson et al. (2011)			X			X		X
Heller et al. (2004)		X		X		X		
Heinz et al. (2001)			X					
Jeswani et al. (2011)	X							
Kabir and Kumar (2012)	X		X					
Keoleian and Volk (2007)	X							
Liu et al. (2010)				X				
Mann and Spath (2001)				X				
Petersen (2006)						X	X	X
Qin et al. (2006)			X					
Schlamadinger et al. (1995)		X						
Tillman (2000)				X				
Wihersaari (2005)			X					

[a] Refers to the type of biomass used; different types of biomass yield different levels of GHG emissions.
[b] Refers to the process of feedstock production and includes assumptions about starting conditions, growth, and inputs (e.g., fertilizer).
[c] Includes the emissions associated with preparation of the feedstock for combustion (e.g., bailing, chipping).
[d] Captures the alterative fates of biomass used for bioenergy.
[e] Changes induced by increased demand for bioenergy feedstock. Includes changes in land use and in land use management intensity.
[f] Includes emissions associated with moving biomass during its various stages of production.
[g] Captures the effects of different combustion pathways for the bioenergy system.
[h] Captures the effects of different combustion pathways for the fossil alternative.

This review suggests that the findings of bioenergy GHG life-cycle analyses must be carefully examined so as to fully appreciate their implications for policy and renewable energy production. Although bioenergy systems are complex, frameworks can help to standardize the analysis process. For example, guidance exists for the conduct of these analyses (e.g., ISO 14040 and 14044) and for comparisons of bioenergy systems to fossil alternatives (e.g., Schlamadinger et al. 1997). Assumptions must nonetheless be clearly specified and adequately justified, system boundaries must be appropriate to the situation at hand, and accounting approaches must always be relevant to the policy question being asked.

QUANTITATIVE ASSESSMENT OF A BIOMASS UTILIZATION SCENARIO

Previous work by the authors includes detailed quantitative assessments of biomass demand on forest composition and extent, the traditional forest products industry, and net GHG emissions (see, e.g., Galik, Abt, and Wu 2009; Abt, Galik, and Henderson 2010; Galik and Abt 2011; Abt, Abt, and Galik 2012; Galik and Abt 2012a-b). Others have developed advanced tools for scoring the economic and GHG consequences of particular biomass feedstock supply chain configurations (e.g., Daystar et al. 2012a). The following analysis combines aspects of this work to assess the multiple implications of converting a coal-fired power plant to woody biomass combustion.

The biomass demand scenario is modeled on the hypothetical conversion of Georgia Power's Plant Mitchell Unit 3 to 100% biomass power. Located in Albany, Georgia, a converted Plant Mitchell would be rated at 96 MW, requiring approximately 1.1 M green tons of woody biomass supply annually. Woody biomass to meet this demand would likely be drawn from surrounding counties (Figure 2). The imposition of this new demand on top of existing demand from pulp, sawtimber, and growing pellet markets is expected to have complex and far-reaching effects on timber product prices and the extent and composition of the region's forests. In turn, changes in the forested landscape will have implications for the net GHG emissions associated with conversion of Plant Mitchell to biomass combustion.

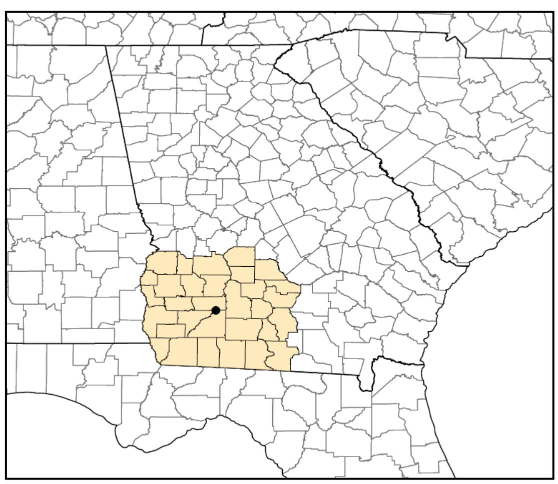

Figure 2. Approximate location of Georgia Power's Plant Mitchell (Albany, Georgia) and assumed woody biomass supply region.

Methods

The analysis assesses two biomass demand scenarios (Table 2). In the first scenario, the market determines the most efficient sources of biomass to use, with "efficient" being a function of both biomass cost and historical harvest and planting behavior. In this scenario, preference is given to utilization of the existing timber supply and logging residues. In the second scenario, a preference is given towards the use of urban wastes and residues, attempting to replicate a situation in which a policy or facility manager may express a preference for a particular feedstock type. We refer to these two scenarios as "market allocation" and "specified feedstock", respectively. Both employ the SubRegional Timber Supply Model (SRTS), a regional partial equilibrium harvest model used to study the impact of various supply and demand drivers for forest resources and related markets (see, e.g., Abt, Cubbage, and Abt 2009; Prestemon and Abt 2002). Both are likewise augmented with GHG emissions data derived from product supply chain life-cycle inventories (e.g., Daystar et al. 2012b).

Table 2. Overview of assumptions for two feedstock demand scenarios.

Parameter	**Conversion Factor or Equation**	**Scenario**	
		Market Allocated Feedstock	**Specified Feedstock Allocation**
Feedstock allocation	N/A	- Endogenous (SRTS-allocated)	- 35% forest residues - 15% whole tree - 50% waste
Plantation establishment and maintenance	kg CO_2e/gr tn [a] L: 19.6 M:14.6 H: 11.3	Emissions estimated for softwood plantations only	
Harvest	kg CO_2e/gr tn [a] L: 17.1 M:12.8 H: 10.1	Emissions estimated for all feedstock, regardless of species or component	
Transportation	kg CO_2e/gr tn [a] L: 4.8 M:4.0 H: 3.6	Assumes mean travel distance of 50km	
Residue decay	tC left @yr *t* [b] Softwoods: *exp(-t/17.88)* Hardwoods: *exp(-t/8.88)*	Emissions estimated for all residues, regardless of species	
Landfill emissions	tC left @yr *t* [b] Softwoods: *exp(-t/17.88)* Hardwoods: *exp(-t/8.88)*	N/A	Calculates alternate fate of waste feedstock stream in baseline scenario Assumes that 15% of decayed carbon is emitted (10% as CO_2; 5% as CH_4)

Note: "L", "M", and "H" refer to low, medium, and high productivity multipliers of 0.75, 1.00, and 1.25, respectively.
[a] Derived from LCI tool developed in Daystar et al. 2012b.
[b] Derived from Smith et al. (2006).

Emissions are estimated for plantation establishment, stand maintenance, harvest, and transportation on a delivered green ton basis, but they differ by feedstock component. Low, medium, and high values are derived from an Excel-based LCI tool (Daystar et al. 2012b) and the underlying references contained therein. Emissions for hardwood plantation establishment and maintenance are not estimated because it is assumed that hardwood stands regenerate naturally. Within softwoods, emissions associated with plantation establishment and maintenance are calculated only for the wholetree component of each biomass scenario, as residues are assumed to be a by-product of management for roundwood. Emissions associated with harvest and transportation are applied to both residue and roundwood components of both hardwood and softwood feedstocks, as fuel is expended for all regardless of whether they were purpose-grown or -established. Average transportation distance is estimated by first examining the distance from

Plant Mitchell to the outer border of the supply area identified in Figure 2. This distance varies, but at its greatest it is approximately 114 km. Therefore, the analysis assumes an average transportation distance of 50km by rounding down this maximum to 100km and dividing by two.

Purpose-harvested forest biomass emissions are captured in the difference between forest carbon in the base case and biomass demand scenarios. Waste combustion emissions are assumed to be 25% carbon (C) of input green ton volume (assuming 50% water weight and then 50% carbon of dry weight), converted to tCO_2e by multiplying by 3.667. On-site residue and landfill decay is estimated using first-order softwood- and hardwood-specific decay functions derived from Smith et al. (2006). Emissions from residues are estimated in two ways, one assuming instantaneous loss through slash pile burning and the other assuming gradual decay over time. In the case of residue decay, it is assumed that the difference in forest carbon storage is equal to the sum of the residues used for bioenergy in the biomass demand scenario in a given year plus the remaining portion of previous-year residues. In the case of landfill decay, it is assumed that only a fraction of total carbon is emitted and that the bulk of carbon is permanently sequestered. Of the 15% of carbon assumed to be emitted, 10% is assumed to be either directly emitted as carbon dioxide or emitted as methane but ultimately converted to carbon dioxide through flaring. The remaining 5% is emitted directly as methane and is converted to tCO_2e for the purposes of total emissions accounting by multiplying by 23.

The net GHG implications of each biomass demand scenario can be expressed as a function of these various parameters. Most simply, the equation is expressed as:

$$\textit{Net GHG Implications of Plant Mitchell Conversion} = (F_i - E_i) + P_i + W_i \quad (1)$$

where F_i is the change in forest carbon storage, E_i is the change in fossil power plant emissions, P_i is the change in process emissions, and W_i is the change in waste feedstock emissions, all for the *ith* biomass demand scenario. Two biomass demand scenarios are presented here, one in which SRTS allocates the feedstock endogenously and one in which specific types and amounts of feedstocks are specified.

Each component of this simple equation can be further broken down as follows:

$$F_i = FC_i - (FB + R_{ij}) \quad (2)$$

where FC_i represents forest carbon storage in the Plant Mitchell conversion for the *ith* biomass demand scenario, and *FB* represents the forest carbon storage in the base case. R_{ij} represents carbon storage attributed to residues used to supply Plant Mitchell for the *ith* biomass demand scenario and the *jth* residue disposition scenario. Two residue disposition scenarios are considered, one assuming instantaneous loss through slash pile burning (*R* equals 0) and one in which residues decay slowly over time.

Fossil emissions are fairly straightforward:

$$E_i = (EC_i - EB) \quad (3)$$

EC_i represents the fossil power plant emissions for the Plant Mitchell conversion for the *ith* biomass demand scenario, and *EB* represents the emissions in the base case.

Process emissions include those associated with plantation establishment, plantation maintenance, biomass harvest, and biomass transportation. They are calculated and applied as follows:

$$P_i = (GC_{ikl}+MC_{ikl}+HC_i+TC_i)-(GB_{kl}+MB_{kl}+HB+TB) \quad (4)$$

GC_{ikl} *and* MC_{ikl} represent the plantation establishment and stand maintenance process emissions for the Plant Mitchell conversion run for the *ith* biomass demand scenario, *kth* species (softwood only), and *lth* component (roundwood only). HC_i and TC_i represent the harvest and transportation emissions associated with the *ith* biomass demand scenario of the Plant Mitchell conversion run. GB_{kl} *and* MB_{kl} represent the plantation establishment and stand maintenance process emissions for the base case run for the *kth* species (softwood only) and *lth* component (roundwood only). HB_i and TB_i represent the harvest and transportation emissions in the base case run.

The final step is to calculate net waste feedstock emissions, estimated as:

$$W_i = L_i - D_i \quad (5)$$

where L_i represents emissions associated with biomass landfilled in the base case scenario and D_i represents the waste feedstock combusted in the biomass demand scenario, again for the *ith* biomass demand scenario. Waste feedstock combustion emissions are assumed to be zero in the base case scenario, and waste feedstock landfill emissions are assumed to be zero in the biomass demand scenario.

Results

Market Allocation of Feedstock

The net GHG consequences of biomass conversion can be expressed as an annual or cumulative figure. High variability in the year-to-year GHG benefit is attributed to Plant Mitchell conversion, though overall that conversion is beneficial from a GHG perspective (Figure 3). Of particular interest is the supply-side response, or the change in forest carbon storage over time. The Southeastern U.S. has historically experienced a great deal of movement in land between forest and agriculture, and any sudden increase or spike in demand is expected to generate a more pronounced land use response than it would in most regions. This is seen in the findings here. Early years show increases in forest carbon storage as new plantations are established and the loss of already-forested acres are slowed relative to the base case scenario, both in response to the new demand for forest biomass. In time, forest carbon declines as the plantations established at the beginning of the scenario mature to harvest age, generating more biomass and leading to a leveling off of prices. As planting slows and the availability of forest biomass declines, prices rise and the cycle repeats, the beginnings of which are seen in the later years of the figure.

The magnitude of the planting and land use change response is influenced by both present market conditions and modeling assumptions. Currently, pine sawtimber prices are low relative to pulpwood prices, meaning that any spikes in pulpwood price will have a more pronounced effect on timber rents, exacerbating the land use response. Agriculture rents are also held constant in this analysis, exacerbating price effect and land use change response further. All contribute to the "boom and bust" cycle seen below. Such cycles are typical in the southern forest inventory, but may be exacerbated in the present case by the above conditions and assumptions. Historically southwest Georgia has had active movement between marginal agriculture and forestry and this flexibility is reflected in the updated empirical model driving land use change (Hardie et al. 2000).

Also of note are the minimal influence of accounting for residues that would have remained on site in the base case as well as the increase in process emissions to harvest and transport biomass under the Plant Mitchell repowering scenario. For residues in particular, any differential in GHG benefit between the two residue scenarios decreases over time as the amount of new residue material added in any given year is partially offset by the degradation of previous-year residues.

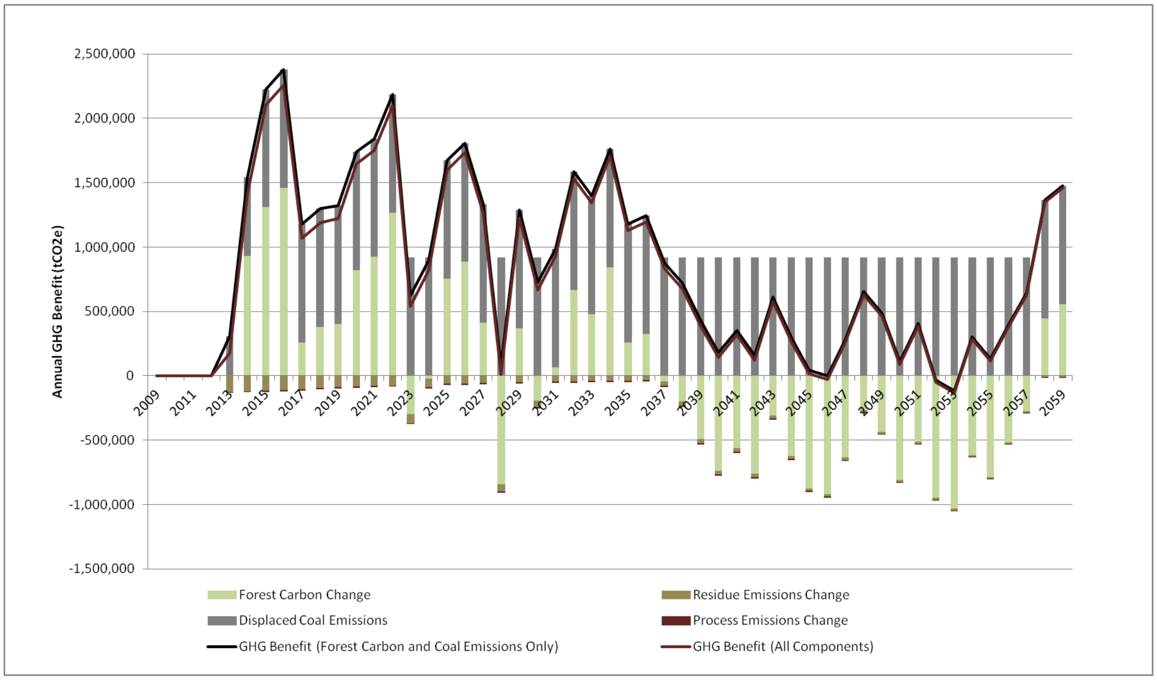

Figure 3. Annual GHG benefit of Plant Mitchell conversion to biomass, assuming market allocation of feedstock. Shown are the GHG benefits of repowering as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case ("Forest Carbon and Coal Emissions Only"). Also shown is the net GHG benefit of the scenario accounting for the increased process emissions associated with the repowering scenario and assuming that base case residues decay over time ("All Components"). Process emissions assume average productivity (i.e., a productivity multiplier of 1.00).

As would be expected, the cumulative GHG benefit associated with the Plant Mitchell conversion reflect annual trends (Figure 4). In particular, the cumulative GHG benefit generally increases over time. But unlike the annual GHG benefit, the cumulative GHG benefit reflects the increasing divergence of the two residue utilization scenarios as residues continually accumulate over time in the base case.

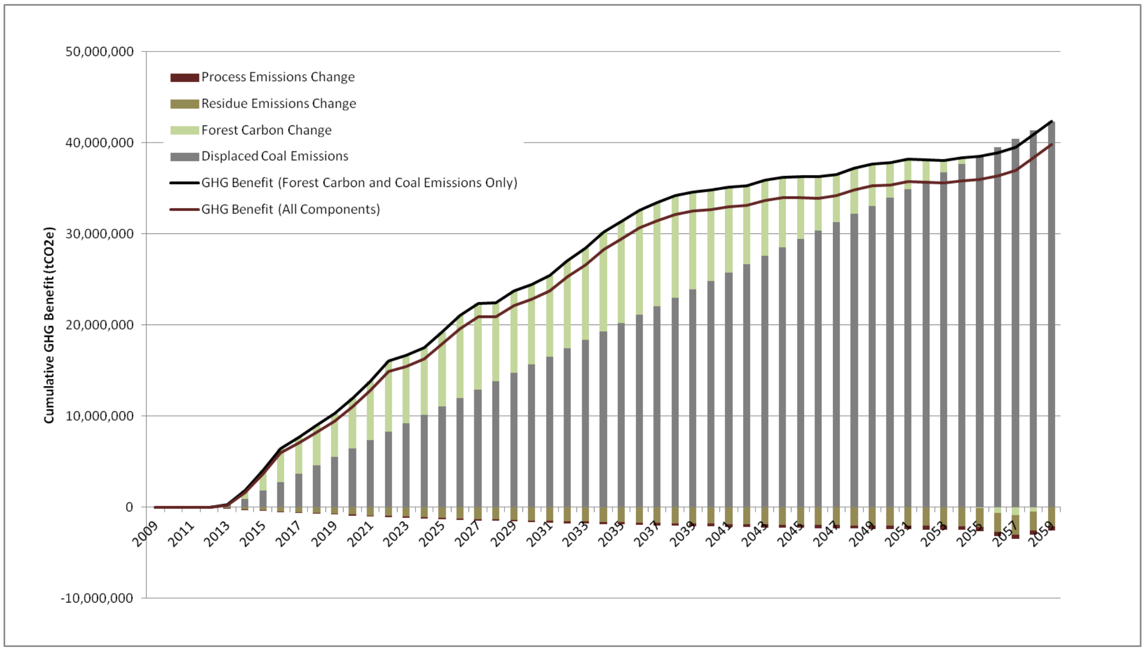

Figure 4. Cumulative GHG benefit of Plant Mitchell conversion to biomass assuming market allocation of feedstock. Shown are the GHG benefits of repowering as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case ("Forest Carbon and Coal Emissions Only"). Also shown is the net GHG benefit of the scenario accounting for the increased process emissions associated with the repowering scenario and assuming that base case residues decay over time ("All Components"). Process emissions assume average productivity (i.e., a productivity multiplier of 1.00).

Process emissions associated with plantation establishment, stand maintenance, harvest, and transportation vary over time and by assumed conversion factor (Figure 5). Relative to annual changes in both fossil emissions and forest carbon, process emissions generally play a very minor role. Aggregate effects are generally imperceptible at the supply-shed level when viewed at the same scale as cumulative GHG benefit figures. Adjustments for productivity levels do change absolute emissions levels, but the patterns are the same.

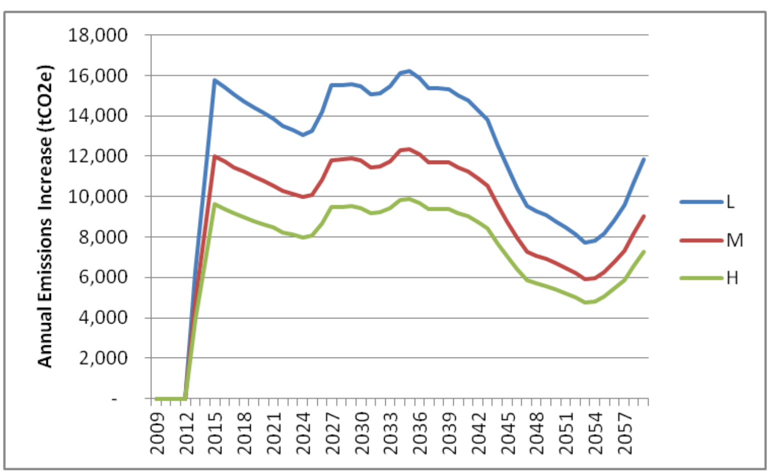

Figure 5. Annual increases in emissions (tCO2e) from plantation establishment, stand maintenance, biomass harvest, and transportation relative to the base case scenario, assuming market allocation of feedstock. Positive values reflect additional emissions attributable to biomass use. "L", "M", and "H" refer to low, medium, and high productivity multipliers of 0.75, 1.00, and 1.25, respectively.

In some instances, however, process emissions comprise a significant percentage of annual emissions change (Figure 6), particularly when reductions in fossil carbon are nearly matched by a corresponding decrease in forest carbon storage. In most years, process emissions are less than 5% of total annual change, but in others, they comprise a significant share of total emissions (e.g., years 2046 and 2052). The question of whether process emissions are relevant to assessments of GHG emissions associated with biomass use depends therefore on the timing of interest. The results here suggest that real-time or even annual reporting may highlight those instances in which process emissions comprise a significant share of total emissions in a given time period. Process emissions appear to matter little for long-lived projects, however, as they are outpaced by the magnitude of changes in cumulative GHG benefit (Figure 7).

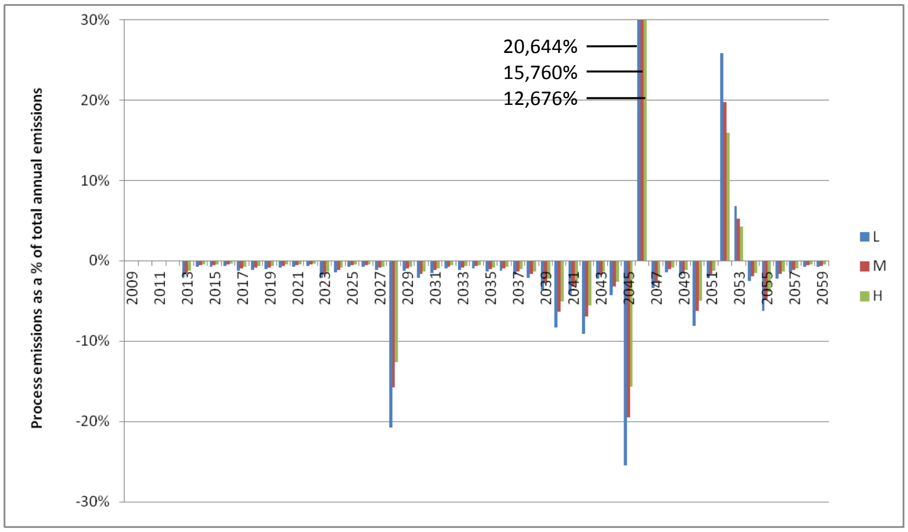

Figure 6. Emissions from plantation establishment, stand maintenance, biomass harvest, and transportation as a percentage of total annual GHG emissions as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case. Positive values imply that process emissions are increasing the magnitude of GHG emissions associated with biomass use; negative values imply that process emissions are decreasing the magnitude of GHG emissions reductions. "L", "M", and "H" refer to low, medium, and high productivity multipliers of 0.75, 1.00, and 1.25, respectively. Figure assumes market allocation of feedstock.

Defined Feedstock Allocation

An alternative approach to the market allocation framework employed by SRTS is the specification of discrete targets for individual categories of feedstock. Under such an approach, a certain pre-determined amount of biomass is sourced from harvest residues, a certain amount from urban wood waste, and a certain amount from purpose-grown roundwood. The GHG dynamics associated with this kind of scenario are likely different from those of a SRTS-defined market allocation scenario in which the biomass will likely come from different sources and in which the market effects that drive many of the important indirect effects (e.g., land use change, change in management intensity, industrial displacement) will also be different.

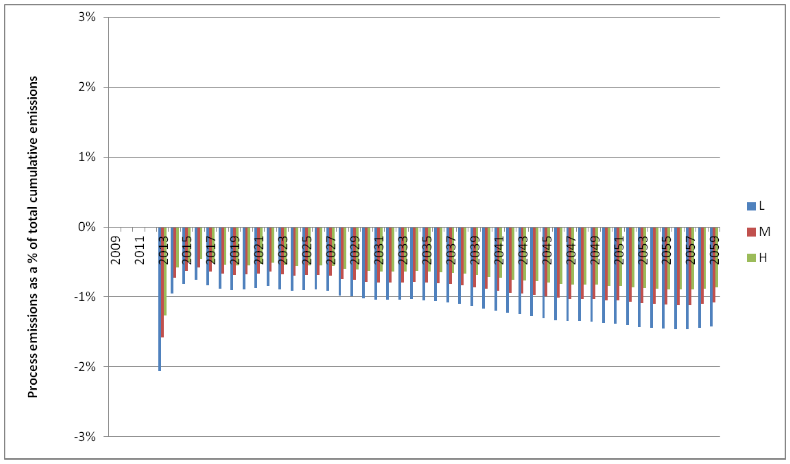

Figure 7. Emissions from plantation establishment, stand maintenance, biomass harvest, and transportation as a percentage of total cumulative GHG emissions as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case. Positive values imply that process emissions are increasing the magnitude of GHG emissions associated with biomass use; negative values imply that process emissions are decreasing the magnitude of GHG emissions reductions. "L", "M", and "H" refer to low, medium, and high productivity multipliers of 0.75, 1.00, and 1.25, respectively. Figure assumes market allocation of feedstock.

The specified feedstock scenario is intended to replicate sourcing assumptions used by others (Moore 2012) to assess the GHG implications of repowering Plant Mitchell. Like Moore (2012), this analysis assumes that 50% of the biomass feedstock comes from waste streams, 35% from various types of harvest residues, and 15% from various sources of roundwood. These categories and targets were input into SRTS and run using a series of modified assumptions. To account for the 50% of feedstock coming from waste, demand for harvested forest biomass was simply cut by half. Next, the logging residue utilization rate was set such that residues yield 35% of total facility demand. Finally, land use parameters were set such that enough new acres were planted to meet feedstock mix requirements. Although such modifications cannot capture fine distinctions within a feedstock category (e.g., residues from the harvest of purpose-grown biomass crops), they nonetheless capture the approximate GHG implications of using a predefined feedstock mix.

Because the analysis models the effects of using waste wood, it must also include an estimate of the emissions associated with their combustion and their alternative use. These emissions are estimated as described above. For the base case scenario, the analysis assumes that 80% of all carbon in combusted waste wood would have been permanently sequestered in a landfill. Of the 20% emitted over time, half is assumed to be emitted as carbon dioxide and half is emitted as methane, half of which is combusted for electricity. To estimate decay emissions, the analysis uses the first-order decay function described above; the methane portion of emissions are converted to units of tCO_2e by multiplying by 23, the difference in global warming potential between methane and carbon dioxide.

The overall GHG benefit associated with the specified feedstock allocation scenario, along with the contributions of individual emissions or sequestration components, can be seen in Figure 8. Of particular interest is the waste feedstock emissions change, measured as the difference between waste landfill emissions in the base case scenario and waste combustion emissions in the biomass demand scenario. In the early years, combustion of wastes generates more emissions than otherwise would have been generated. Over time, however, the increasing pool of landfilled wood in the base case scenario begins to generate levels of greenhouse gases sufficient to offset and even surpass annual combustion emissions. Turning to the cumulative GHG benefit, these latter-year gains are nonetheless insufficient to prevent waste feedstock emissions from yielding a net reduction in GHG benefit (Figure 9).

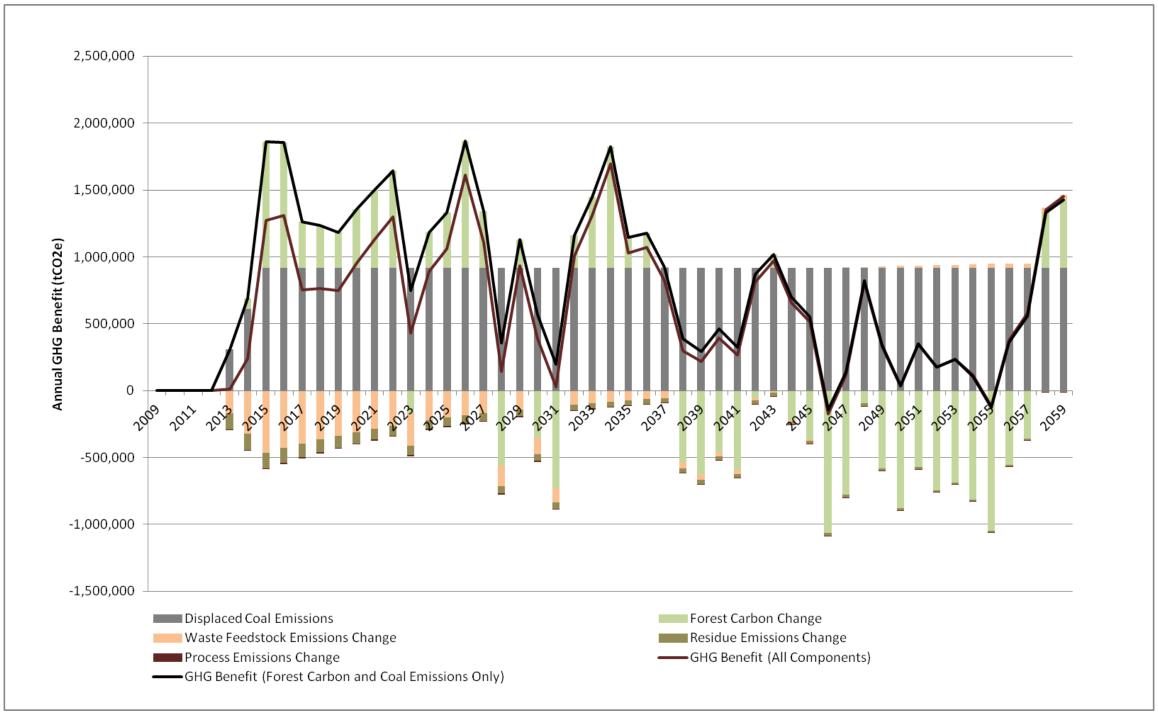

Figure 8. Annual GHG benefit of Plant Mitchell conversion to biomass assuming specified feedstock allocation. Shown are the GHG benefits of repowering as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case ("Forest Carbon and Coal Emissions Only"). Also shown is the net GHG benefit of the scenario accounting for the change in waste feedstock emissions and the increased process emissions associated with the repowering scenario assuming that base case residues decay over time ("All Components"). Process emissions assume average productivity (i.e., a productivity multiplier of 1.00).

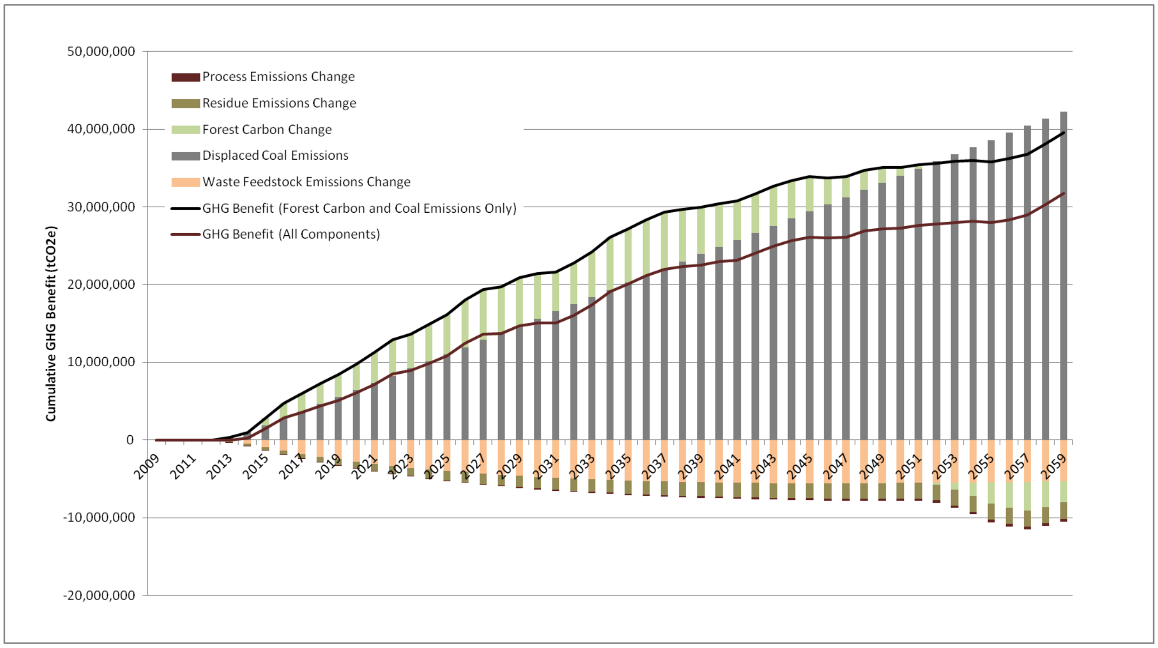

Figure 9. Cumulative GHG benefit of Plant Mitchell conversion to biomass assuming specified feedstock allocation. Shown are the GHG benefits of repowering as estimated by changes in plant fossil emissions and changes in forest carbon assuming that utilized residues are burned on site in the base case ("Forest Carbon and Coal Emissions Only"). Also shown is the net GHG benefit of the scenario accounting for the change in waste feedstock emissions and the increased process emissions associated with the repowering scenario assuming that base case residues decay over time ("All Components"). Process emissions assume average productivity (i.e., a productivity multiplier of 1.00).

As in the market-allocated feedstock example described above, process emissions are a small component of both annual and cumulative emissions. Total emissions trends are similar to those in the market-allocated feedstock example seen in Figure 5, while the percentage of annual and cumulative reductions comprised of process emissions is similar to that in the market-allocated examples seen in Figure 6 and Figure 7 except more muted. Annual shares rise above 10% in only one year and even then do not exceed 20%.

Comparison

Trends in annual GHG benefits are similar in the market-allocated feedstock example and specified feedstock example (Figure 10). In the early years, the market-allocated feedstock example shows higher GHG benefits stemming from a larger forest carbon response, because demand is not tempered by large allocations to waste and residue streams. The increase in emissions from waste combustion in the specified feedstock example further differentiates the two scenarios. The two scenarios increasingly align as waste emissions are tempered by offset landfill emissions and as the forest carbon response is tempered by greater timber supply and falling timber prices stemming from increased planting activity in the early years of the market-allocated feedstock example. The cumulative GHG benefit of the market-allocated feedstock example is larger than that of the specified feedstock example (Figure 11), mainly because of the former's increased supply-side response and absence of waste combustion emissions.

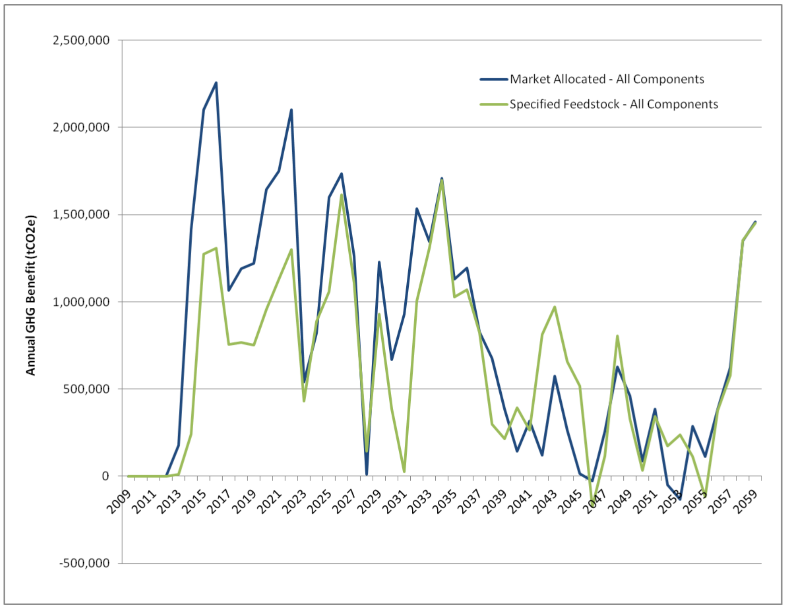

Figure 10. Comparison of the annual GHG benefit in the market-allocated feedstock and specified feedstock examples. In the market-allocated example, the figure includes the increased process emissions associated with the repowering scenario assuming that base case residues decay over time. In the specified feedstock example, the figure includes the change in waste feedstock emissions and the increased process emissions associated with the repowering scenario assuming that base case residues decay over time.

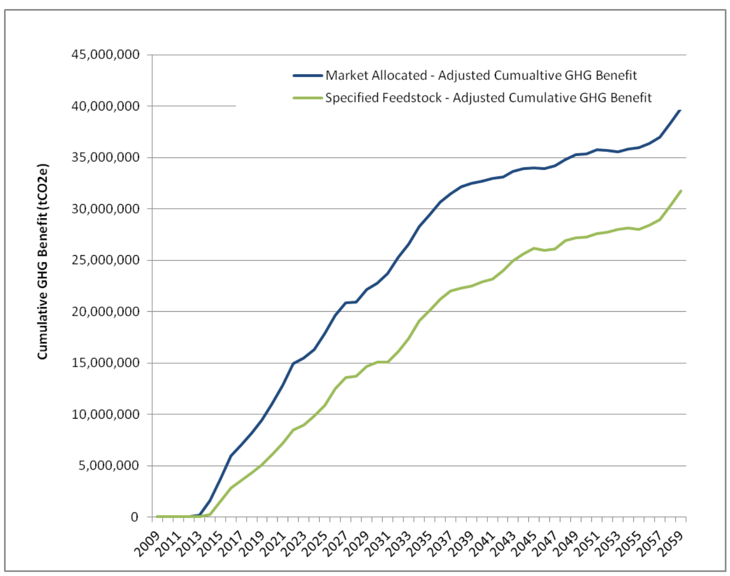

Figure 11. Comparison of the cumulative GHG benefit in the market-allocated feedstock and specified feedstock examples. In the market-allocated example, the figure includes the increased process emissions associated with the repowering scenario assuming that base case residues decay over time. In the specified feedstock example, the figure includes the change in waste feedstock emissions and the increased process emissions associated with the repowering scenario assuming that base case residues decay over time.

The difference in supply-side response is not unexpected, because the large waste and residue pools in the specified feedstock example limit the demand for purpose-harvested forest biomass. This reduced demand in turn mutes pressure on forest product prices, the ultimate driver of planting, harvest, and management behavior changes. This response is endogenous to SRTS as it is run normally, but steps were taken in the specified feedstock example to ensure that new plantings achieve target harvest allocations. This added, exogenous component is likely to further reduce forest supply response to biomass demand in the specified feedstock example as compared with the market-allocated example.

Trends in annual and cumulative change in forest acreage are similar to those in annual and cumulative GHG benefit, but key differences do emerge (Figure 12 and Figure 13). As expected, the increase in forested acres is larger in the early years of the market-allocated feedstock example than in the specified feedstock example; the decrease in forested acres is also larger in the later years of the former than in the latter. Apart from these differences, the two trends are roughly similar and suggest a net increase in forested acres in the face of increased biomass demand from the hypothetical conversion of Plant Mitchell.

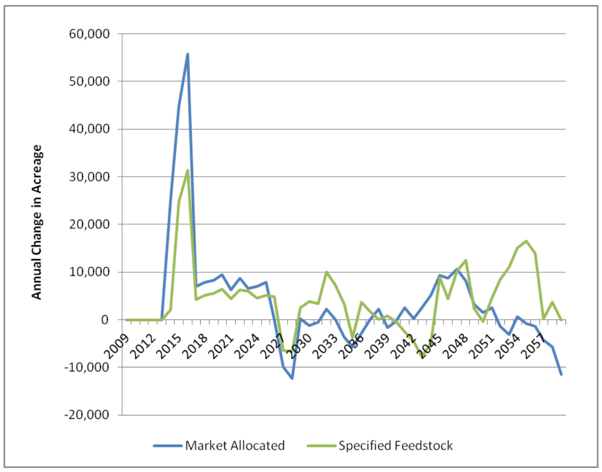

Figure 12. Comparison of annual change in forested acres in the market-allocated feedstock and specified feedstock examples.

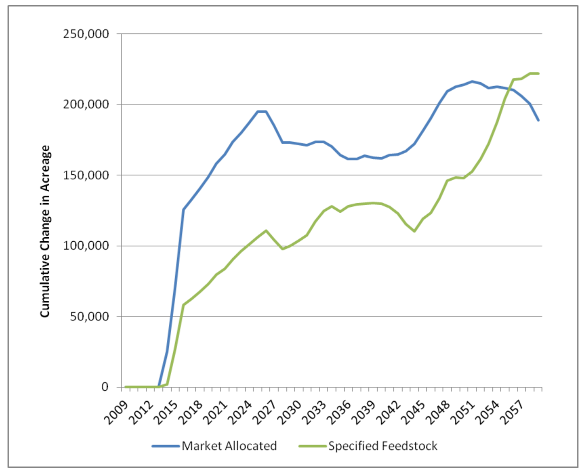

Figure 13. Comparison of cumulative change in forested acres in the market-allocated feedstock and specified feedstock examples.

CONCLUSIONS

The hypothetical plant repowering reviewed in this case study confirms the sizeable role played by accounting system, system boundary, and GHG neutrality assumptions. Taking a different approach in any of these three areas—whether by eliminating a focus on induced market effects, ignoring changes in forest carbon storage, or assuming that biomass combustion does not generate net GHG emissions—would yield vastly different results. The comparison of market- and specified feedstock allocation scenarios indicates the critical role of a change in accounting assumptions. The large influences of both forest carbon and landfill pools indicate the relevance of system boundary assumptions, and the size of the forest carbon pool relative to the displaced emissions pool reinforces the relevance of GHG neutrality assumptions. The analysis shows that biomass combustion in the hypothetical repowering case study reviewed here can be net GHG beneficial. An analysis based on a different set of assumptions is likely to generate a different set of findings based on the same underlying data, however (see, e.g., Galik and Abt 2012b). This realization reinforces the importance of standardizing GHG assessment techniques, ensuring that they are clear, transparent, and relevant to the questions at hand. This is true in the case of individual facility assessments, in policy determinations, and in the broader scientific literature that may ultimately inform them.

LITERATURE CITED

Abt, K.L., R.C. Abt, and C.S. Galik. 2012. Simulating supply response to bioenergy demands in the Southeastern U.S. *Forest Science* 58(5): 523–539.

Abt, R.C., F.W. Cubbage, and K.L. Abt. 2009. Projecting Southern timber supply for multiple products by subregion. *Forest Products Journal* 59: 7–16.

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. Durham, NC: Climate Change Policy Partnership, Duke University.

Baxter, L., 2005. Biomass-coal co-combustion: opportunity for affordable renewable energy. *Fuel* 84: 1295–1302.

Bird, D.N., N. Pena, and G. Zanchi. 2012. Zero, one, or in between: evaluation of alternative national and entity-level accounting for bioenergy. *GCB Bioenergy* 4: 576–587.

Carpentieri, M., A. Corti, and L. Lombardi. 2005. Life cycle assessment (LCA) of an integrated biomass gasification combined cycle (IBGCC) with CO_2 removal. *Energy Conversion and Management* 46: 1790–1808.

Cherubini, F., N.D. 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. *Resources Conservation and Recycling* 53: 434–447.

Cherubini, F., and A.H. Strømman. 2011. Life cycle assessment of bioenergy systems: State of the art and future challenges. *Bioresource Technology* 102: 437–451.

Corti, A., and L. Lombardi. 2004. Biomass integrated gasification combined cycle with reduced CO_2 emissions: performance analysis and life cycle assessment (LCA). *Energy* 29: 2109–2124.

Damen, K., and A. Faaij. 2006. A greenhouse gas balance of two existing international biomass import chains. *Mitigation and Adaptation Strategies for Global Change* 11: 1023-1050.

Daystar, J., C. Reeb, R. Venditti, R. Gonzalez, and M.E. Puettmann. 2012a. Life-cycle assessment of bioethanol from pine residues via indirect biomass gasification to mixed alcohols. *Forest Products Journal* 62: 314–325.

Daystar, J., C. Reeb, R. Gonzalez, T. Treasure, R. Venditti, R. Abt, and S. Kelley. 2012b. Integrated supply chain, delivered costs and life cycle assessment of several lignocellulosic supply systems for biofuels, bioenergy and bioproducts in the Southern U.S. Unpublished manuscript.

Djomo, S.N., O. El Kasmioui, and C. Reinhart. 2011. Energy and greenhouse gas balance of bioenergy production from poplar and willow: a review. *GCB Bioenergy* 3: 181–197.

Galik, C.S., and R.C. Abt. 2011. An Interactive Assessment of Biomass Demand and Availability in the Southeast United States. Durham, NC: Nicholas Institute for Environmental Policy Solutions, Duke University.

Galik, C.S., and R.C. Abt. 2012a. Forest biomass supply for bioenergy in the southeast: evaluating assessment scale. In *Monitoring Across Borders: Proceedings of the 2010 Joint Meeting of the Forest Inventory and Analysis (FIA) Symposium and the Southern Mensurationists,* edited by W. McWilliams and F.A. Roesch, 255–263. e-Gen. Tech. Rep. SRS-157. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station.

Galik, C.S., and R.C. Abt. 2012b. The effect of assessment scale and metric selection on the greenhouse gas benefits of biomass. *Biomass & Bioenergy* 44: 1–7.

Galik, C.S., R.C. Abt., and Y. Wu. 2009. Forest biomass supply in the Southeastern United States: implications for industrial roundwood and bioenergy production. *Journal of Forestry* 107: 69–77.

Gold, B.A., and D.A. Tillman. 1996. Wood cofiring evaluation at TVA power plants. *Biomass and Bioenergy* 10: 71–78.

Gustavsson, L., P. Börjesson, B. Johansson, and P. Svenningsson. 1995. Reducing CO_2 emissions by substituting biomass for fossil fuels. *Energy* 20: 1097–113.

Gustavsson, L., L. Eriksson, and R. Sathre. 2011. Costs and CO_2 benefits of recovering, refining and transporting logging residues for fossil fuel replacement. *Applied Energy* 88: 192–197.
Hardie, I.W., P.J. Parks, P. Gottlieb, and D.N. Wear. 2000. Responsiveness of rural and urban land uses to land rent determinants in the U.S. South. *Land Economics* 78: 659–673.
Heller, M.C., G.A. Keoleiana, M.K. Mann, and T.A. Volk. 2004. Life cycle energy and environmental benefits of generating electricity from willow biomass. *Renewable Energy* 29: 1023–1042.
Heinz, A., M. Kaltschmitt, R. Stülpnagel, and K. Scheffer. 2001. Comparison of moist vs. air-dry biomass provision chains for energy generation from annual crops. *Biomass and Bioenergy* 20: 197–215.
Jeswani, H.K., H. Gujba, and A. Azapagic. 2011. Assessing options for electricity generation from biomass on a life cycle basis: environmental and economic evaluation. *Waste Biomass Valor* 2: 33–42.
Kabir, M.K., and A. Kumar. 2012. Comparison of the energy and environmental performances of nine biomass/coal co-firing pathways. *Bioresource Technology* 124: 394–405.
Keoleian, G.A., and T.A. Volk. 2007. Renewable energy from willow biomass crops: life cycle energy, environmental and economic performance. *Critical Reviews in Plant Sciences* 24(5–6): 385–406.
Liu, H., K.R. Polenske, Y. Xi, and J. Guo. 2010. Comprehensive evaluation of effects of straw-based electricity generation: a Chinese case. *Energy Policy* 38: 6153–6160.
Mann, M., and P. Spath. 2001. A life cycle assessment of biomass cofiring in a coal-fired power plant. *Clean Products and Processes* 3: 81–91.
Moore, J. 2012. Carbon lifecycle budget study phase II. Moore Ventures. Unpublished report.
Petersen Raymer, A.K., 2006. A comparison of avoided greenhouse gas emissions when using different kinds of wood energy. *Biomass and Bioenergy* 30: 605–617.
Prestemon, J.P., and R.C. Abt. 2002. "Timber Products Supply and Demand." In *Southern Forest Resource Assessment,* edited by D.N. Wear and J.G. Greis, 299–325. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station.
Qin, X., T. Mohan, M. El-Halwagi, G. Cornforth, and B.A. McCarl. 2006. Switchgrass as an alternate feedstock for power generation: an integrated environmental, energy and economic life cycle assessment. *Clean Technology Environmental Policy* 8: 233–249.
Schlamadinger, B., M. Apps, F. Bohlin, L. Gustavsson, G. Jungmeier, G. Marland, K. Pingoud, and I. Savolainen. 1997. Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. *Biomass and Bioenergy* 13: 359–375.
Schlamadinger, B., J. Spitzer, G.H. Kohlmaier, and M. Lüdeke. 1995. Carbon balance of bioenergy from logging residues. *Biomass and Bioenergy* 8: 221–234.
Schmidt, J., S. Leduc, E. Dotzauer, G. Kindermann, and E. Schmid. 2010. Cost-effective CO_2 emission reduction through heat, power and biofuel production from woody biomass: A spatially explicit comparison of conversion technologies. *Applied* Energy 87: 2128–2141.
Smith, J.E., L.S. Heath, K.E. Skog, and R.A. Birdsey. 2006. Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States. GTR-NE-343. U.S. Durham, NH: Department of Agriculture, Forest Service Northeastern Research Station.
Tillman, D.A. 2000. Biomass cofiring: the technology, the experience, the combustion consequences. *Biomass and Bioenergy* 19: 365–384.
Wihersaari, M. 2005. Greenhouse gas emissions from final harvest fuel chip production in Finland. *Biomass and Bioenergy* 34: 82–90.
Zhang, Y., J. McKechnie, D. Cormier, R. Lyng, W. Mabee, A. Ogino, and H.L. MacLean. 2010. Life cycle emissions and cost of producing electricity from coal, natural gas and wood pellets in Ontario, Canada. *Environmental Science and Technology* 44: 538–544.