# Appendix A: IPCC Inventory Approach to Accounting for All Anthropogenic Greenhouse Gas Emissions

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## 1. Background on IPCC Guidelines

The Intergovernmental Panel on Climate Change (IPCC) developed a common system for countries to inventory all anthropogenic greenhouse gas (GHG) emissions, including fossil and biogenic carbon dioxide ($CO_2$) emissions, across all sectors in a way that reflects net physical additions of GHGs to the atmosphere in a year (IPCC, 2006). The IPCC system classifies source and sink categories into sectors, e.g., Energy, Industrial Processes, Agriculture, Land-Use Change and Forestry, and Waste.[1] In classifying specific source and sink categories, the IPCC needed to make decisions in cases where a category could reasonably be included in more than one sector. For example, the carbon dioxide emissions from the consumption of limestone during cement production are assigned to the Industrial Processes Sector, whereas carbon dioxide emissions from fossil fuel consumption used to provide useful heat for cement production are assigned to the Energy Sector. The IPCC system works because, as long as each country estimates all anthropogenic sources and sinks and classifies them in the same way, national greenhouse inventories are comparable and can facilitate international efforts and agreements to reduce emissions.

Recognizing that many anthropogenic factors influence emissions and sequestration in biological systems, the IPCC opted to reflect these factors comprehensively and holistically in an assessment of the entire Land-Use Change and Forestry (LUCF) sector as part of the Revised 1996 IPCC Guidelines, (Apps et al., 1997) and continued with this approach for the Agriculture, Forestry and Other Land Use (AFOLU) Sector in the updated 2006 IPCC Guidelines (IPCC, Vol. 4, 2006;).

As a result, net biogenic $CO_2$ emissions related to terrestrial carbon stocks, were "assigned" to the land sector (Land Use, Land Use Change and Forestry, or LULUCF), even if the emissions actually take place at facilities typically associated with a different IPCC sector.[2] Using this approach, countries have been able to communicate the contribution of their land areas to the global build-up of GHG concentrations through their Inventories in a consistent manner.

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[1] The 2006 IPCC Guidelines merged the Agriculture with Land-Use Change and Forestry into a single sector: Agriculture, Forestry and Other Land-Use (AFOLU).

[2] For example, anthropogenic activities that influence GHG storage and fluxes within biological systems, including terrestrial biomass that sequesters and stores carbon, are counted the land or LULUCF sector. Even if biomass is burned for energy, those biogenic $CO_2$ emissions are accounted for in the LULUCF sector where the carbon was stored, not the Energy sector.

The IPCC recommends that countries also calculate direct $CO_2$ emissions from bioenergy, but these estimates are not to be added to national total emissions. These $CO_2$ emissions should instead be itemized and presented separately to promote an overall understanding of a country's energy sector profile:

> *Biomass Fuels: Biomass fuels are included in the national energy and emissions accounts for completeness. These emissions should not be included in national $CO_2$ emissions from fuel combustion. If energy use, or any other factor, is causing a long-term decline in the total carbon embodied in standing biomass (e.g., forests), this net release of carbon should be evident in the calculation of $CO_2$ emissions described in the Land-Use Change and Forestry chapter. (IPCC, 1996)*[3]

The 2006 Guidelines state:

*Biomass is a special case:*

- *Emissions of $CO_2$ from biomass fuels are estimated and reported in the AFOLU sector as part of the AFOLU methodology. In the reporting tables, emissions from combustion of biofuels are reported as information items but not included in the sectoral or national totals to avoid double counting…*
- *For biomass, only that part of the biomass that is combusted for energy purposes should be estimated for inclusion as an information item in the Energy sector. (IPCC, 2006)*[4]

This system, in which $CO_2$ emissions from bioenergy are not directly added to national totals has, on occasion, been interpreted as an IPCC conclusion on the carbon neutrality of bioenergy. The IPCC Guidelines do not, however, provide any conclusions about the GHG mitigation benefits of bioenergy—they explain that biomass used for energy cannot not be automatically considered "carbon neutral" even if the biomass is thought to be produced sustainably (IPCC, 1996; 2006). The IPCC recognizes that biomass use for energy could have an impact on the net atmospheric contribution of emissions and that a comprehensive approach to account for all sources and sinks at the national level would be inclusive of that impact occurring within a country's borders.

## 2. Application of the IPCC Approach to Stationary Sources

Application of the IPCC classification system to $CO_2$ emissions from the consumption of biologically based feedstocks for an individual stationary source would lead to an outcome that excludes impacts on land-based emissions and sequestration. Stationary source emissions (fossil fuel emissions) are captured in one IPCC sector (Energy) and terrestrial fluxes (biomass fuels, such as fuelwood, and related emissions, along with other terrestrial biogenic carbon and carbon-based gases) in the Agriculture, Forestry and Other Land Use sector (AFOLU). In essence, if there is no

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[3] Page 1.10. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories: Reference Manual.
[4] These bullets are taken directly from the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 2: Energy, Chapter 2: Stationary Combustion, Section 2.3.3.4 Treatment of Biomass. See www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_2_Ch2_Stationary_Combustion.pdf.

corresponding accounting (i.e., of both the Energy and AFOLU sectors) or only incomplete accounting of land-based fluxes, then application of the IPCC national inventory approach to stationary source emissions estimation does not provide a complete picture of the true net atmospheric contribution associated with the biogenic $CO_2$ emissions from the stationary source (Pena et al., 2011).

The IPCC recognizes this limitation:

> *The IPCC methodologies are intended to estimate national, anthropogenic emissions and removals rather than life cycle emissions and removals. However the IPCC Guidelines can be used, with care for different purposes. For calculating emissions from substitutions, all the changes in emissions and removals must be accounted for. (IPPC, 1996; 2006)*

As noted above, the success of the IPCC approach relies on the completeness of the accounting for all emissions sources and sinks across all sectors. IPCC methods are built for national-level emissions inventory accounting for general full-sector coverage inventory use, whereas the framework presented in this report provides a more granular accounting method. For this reason, the IPCC classification approach is not designed to address the specific needs and questions that this framework addresses: how assess the net atmospheric contribution of biogenic $CO_2$ emissions associated with the production, processing and use of biogenic feedstocks at stationary sources, taking into account factors related to the biological carbon cycle.

# 3. References

Apps, M., T. Karajalainen, G. Marland, and B. Schlamadinger. 1997. *Accounting System Considerations: $CO_2$ Emissions from Forests, Forest Products, and Land-Use Change—A Statement from Edmonton*. Paris, France: International Energy Agency, Office of Energy Technology, Efficiency and R&D.

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# Appendix B: Temporal Scale

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# 1. Introduction

The purpose of this appendix is to discuss considerations related to the treatment of time in possible applications of the framework. It is important to consider possible treatments of time and the implications of these treatments in developing strategies for long-term and short-term emissions accounting, because the choice of treatment may have significant impacts on the outcome of an assessment framework application. While there is no single, scientifically correct method for the treatment of time when assessing biogenic emissions, there are a number of options for incorporating temporal dynamics into an assessment of biogenic carbon fluxes. The choice of temporal assessment method could ultimately depend on the context of a specific framework application.

This appendix discusses various aspects related to assessing time-dependent effects in the production, processing, and consumption of biogenic feedstocks. Considerations related to time can include a variety of issues such as:

- Emissions horizons and reporting periods (i.e., fluxes related to feedstock production may occur over many years, whereas reporting may be the current year);
- Interplay with spatial scale (i.e., implications of larger scales and shorter time frames versus smaller scales and longer time frames);
- Baseline perspective (i.e., is the analysis forward- or backward-looking, or both?); and
- Differences in temporal characteristics of different feedstocks (i.e., annual crops, short rotation energy crops, and longer rotation forestry systems).

In general, accounting for temporal effects will be most significant when considering future potential fluxes related to long rotation feedstocks (e.g., roundwood), activities that affect the equilibrium storage in soil carbon pools, decay rates, or in cases of significant land use change, where biogenic feedstock production has implications for long-term emissions changes in terrestrial carbon stocks.

Given that different temporal perspectives could be used by the framework, two different baseline approaches are evaluated in this framework report: retrospective reference point and future anticipated baseline. These baseline approaches use aspects of time. The retrospective reference point baseline does not take into account future potential biogenic emissions fluxes related to biogenic feedstock production, processing, and use. The future anticipated baseline, due to its prospective nature, can take into account such future potential fluxes. As such, most of the discussion in this appendix focuses on potential methods for considering time in terms of a prospective analysis.

This appendix provides various illustrative treatments of temporal dynamics when activities and related emissions fluxes do not fit neatly into single assessment time periods. As presented in Section 4, illustrative treatments for prospective applications of the framework in this appendix include a frontloading approach, a year-to-year carryover approach, and an annualized carryover approach. A discussion of discounting time is provided in Section 5.

# 2. Key Temporal Scale Considerations

The production, processing, and use of biogenic feedstocks for energy can, in some circumstances, have emission effects extending into the future and there are different methods to and perspectives about how to assess future emissions trajectories (Dornburg and Marland, 2008; Fargione, 2008; Kendall et al., 2009; Levasseur et al., 2010; Walker et al., 2010; Cherubini et al, 2011; Mitchell et al., 2012; Helin et al., 2013; Walker et al., 2013; Miner et al., 2014). Accounting for these emissions appropriately in different policy contexts may necessitate various decisions that reflect the goals and parameters of the policy. An application of the framework presented in this report that includes assessment over time may need to identify emissions and assessment horizons, reporting periods, the appropriate baseline method, the appropriate spatial scale, and the temporal characteristics. These considerations are discussed below in more detail.

## 2.1. Emissions Horizon, Assessment Horizon, and Time of Reporting

An application of the framework that includes assessment over time may need to articulate how biogenic $CO_2$ emissions fluxes over time from biogenic feedstock production, processing, and use relate to stationary source biogenic $CO_2$ emissions in a single period (e.g., time of biogenic feedstock use or reporting). It is not only a question of how far into the future must an analysis look, but also how these emissions are accounted for and valued over time, and when are they accounted for or reported. Thus, it may be necessary to distinguish between the "emissions horizon" and the "assessment horizon." The emissions horizon is the period of time during which the carbon fluxes resulting from actions taking place today actually occur, while the assessment horizon is a period of time selected for the analysis of the carbon fluxes. In effect, these time horizons can differ significantly.

For example, the emissions horizon reflects all future estimated net carbon fluxes associated with the production and harvest or removal of a feedstock today. Therefore, the emissions horizon may need to span a year to several decades, depending on the feedstock and production site conditions, to account for all these effects. The assessment horizon, however, may be a specified time frame over which estimated future effects may be taken into account. For example, a specific policy may allow the inclusion of future potential effects over 20 years, whereas the estimated emissions horizon is 50 years. The time of reporting may be a one-time event or an annual event at the time or in the year in which the harvest/removed feedstock is consumed at the stationary source. When making determinations about time frame per policy or program needs, one should consider how to address these different time horizons. Illustrative general methods for reconciling these different horizons are discussed in Section 4.

## 2.2. Temporal Differences between Feedstocks

Biogenic carbon fluxes related to biogenic feedstock growth, harvest, and/or collection, feedstock production site soil carbon levels, and land use and/or management change do in many cases occur over a period greater than one year. The consideration of multiyear time dynamics for biogenic feedstock growth is particularly relevant for long rotation feedstocks or feedstocks where carbon stored in biomass accumulates over time subject to biological growth functions and where feedstock production and/or collection affect landscape soil carbon dynamics or other land use changes. For long rotation feedstocks, the amount of biogenic $CO_2$ emissions from harvest and combustion may take years to be sequestered on the same site from which it was harvested. For logging residues, analysts may need to consider decay and associated landscape biogenic $CO_2$ emissions. For example, the collection and combustion of logging residues result in an immediate release of biogenic $CO_2$ emissions that otherwise might have instead occurred in the form of $CO_2$ and $CH_4$ over a series of years through natural decomposition on the forest floor. Concurrently, removal of the logging residues can cause increased emissions through loss of soil carbon over time, while also altering rates of forest growth and carbon sequestration. Changing management practices can also potentially affect mineral soil carbon pools (Buchholz et al., 2013).

Time dynamics may also be a relevant consideration for some agricultural feedstocks. For example, land use change such as the removal of forests for agricultural feedstock production could result in an initial release of carbon that is not fully recaptured in subsequent use of the land for agriculture. Furthermore, cultivation of perennial bioenergy feedstocks such as switchgrass can lead to long-term increases in soil organic carbon relative to annual crops due to extensive root systems (belowground biomass) and reduction of tillage disturbances. Also, changing management practices, such as removing agricultural residues like corn stover, may reduce decay-related emissions but also reduce soil carbon inputs and thus long-term soil organic carbon stocks.

## 2.3. Interactions between Spatial and Temporal Scales

Temporal aspects of biogenic carbon fluxes can also depend on the choice of spatial scale. In some circumstances, assessing biogenic carbon fluxes at a small spatial scale for a long period of time can result in similar outcomes to those from considering a large spatial scale over a short period of time. For example, the harvest of a long-rotation feedstock, such as roundwood, on a significantly small spatial scale (e.g., plot or stand) will initially result in biogenic carbon emissions, but over enough time, replanted trees (e.g., assuming similar species, conditions) will sequester approximately the same amount of carbon that was released by the previous harvest. However, if that same amount of harvest is considered over a larger spatial scale (e.g., a stand within a region), the biogenic carbon emitted from the harvested stand will be balanced out by sequestration in that region from the continued growth of unharvested roundwood and any reforestation activities in the region over a relatively short time frame (likely shorter than regrowth of the stand itself).

## 2.4. Temporal Differences between Baselines

The retrospective reference point baseline and future anticipated baseline approaches both include treatments of time. However, the way in which these two baseline approaches consider time is markedly different. The retrospective reference point baseline approach is inherently backward-looking (because it evaluates measured or modeled emissions fluxes over a specific time frame in the past), while the future anticipated baseline approach is inherently forward-looking (because it evaluates points in time along different future simulations).

When the reference point baseline approach is applied retrospectively, it takes into account net atmospheric biogenic $CO_2$ contributions associated with biogenic feedstock production on the landscape by assessing differences in biogenic stocks and flows between two points in time in the past. Under this baseline approach, one must decide which specific reference points in time to use, including the length of time between reference points (e.g., 5, 10, 15 years, or other?) and the location of the points in the chosen time horizon (e.g., at what point in time was data first collected, when were the most recent data produced?). Integration of future multiyear fluxes (e.g., from potential decay, soil carbon equilibrium changes) is not necessary when values for framework terms are derived through a backward-looking approach (i.e., the retrospective reference point baseline). Appendices H and I show illustrative equation term calculations and case study applications for forest- and agriculture-derived feedstocks using the retrospective reference point baseline approach.

The future anticipated baseline approach assesses the estimated net change in carbon stocks between two projected future scenarios at the same specified point in time, that is, between a business-as-usual (BAU) scenario and an alternative scenario with changes in estimated environmental, economic, and/or policy conditions (e.g., Searchinger et al., 2009). Because this baseline approach can be used to project future biogenic carbon-based fluxes associated with biogenic feedstock production, processing, and use, there are more considerations about how to represent and incorporate elements of time into such an analysis than in the retrospective reference point approach. Integration of future multiyear fluxes (e.g., from potential decay, soil carbon equilibrium changes, other land use and/or management change effects) may be necessary for framework terms representing biogenic landscape attribute values (*GROW, AVOIDEMIT, SITETNC*, and *LEAK*, if included) and possibly process attributes (depending on treatment of biogenic carbon losses through the supply chain, including storage losses or carbon stored in final products, as captured by the *L* and *P* terms). Appendices J, K, and L, respectively, discuss future anticipated baseline considerations, possible baseline construction methods, and illustrative forest- and agriculture-derived feedstock case study applications using this baseline. Waste-derived feedstocks, as discussed in detail in Appendix N, are assessed in this report by using potential alternative pathways and related GHG pathways for those materials, which in many cases include consideration of future potential methane emissions from decomposition if not used for energy.

# 3. Illustration of General Temporal Dynamics Using Decay Rates

The magnitude of an emissions pulse (meaning, in this context, the cumulative biogenic carbon-based emissions over a time period) may depend on how far into the future an analysis is extended. In theory, one could look as far into the future as required to physically account for a multiyear carbon flux (i.e., the entire emissions horizon over which the flux occurs). In practice, however, a shorter time frame may be warranted in specific accounting circumstances, especially if the fluxes toward the tail end of a multiyear flux pattern are very small or a specific program or policy application necessitates a specific, shorter time frame.

To simply explain the general dynamics of time, this appendix uses concepts called the "Fraction of Carbon Remaining" ($FCR_t$) and "Fraction of Carbon Emitted" ($FCE_t$) to illustrate the implications of different choices of time frame when assessing emissions flux dynamics over time ($t$). Using the specific context of natural decay from logging residue feedstock as an example, $FCR_t$ is the amount of carbon that remains (in terms of $mtCO_2e$) on the site ($CR_t$) after a particular time frame divided by the magnitude of the original carbon pool ($CR_0$), assuming a particular decay rate:

$$\boldsymbol{FCR_t = \frac{CR_t}{CR_0}} \qquad \textbf{(EQ. B.1)}$$

Where:

$$\boldsymbol{CR_t = CR_0 \times (1 - decay\ rate)^t} \qquad \textbf{(EQ. B.2)}$$

$FCE_t$ is calculated as:

$$\boldsymbol{FCE_t = 1 - FCR_t} \qquad \textbf{(EQ. B.3)}$$

$FCR_t$ and $FCE_t$ are unit-free (i.e., dimensionless) values by their definitions. Table B-1 provides examples of the impact of different accounting time frames on the emissions pulse accounting (i.e., $FCE_t$ over the defined time period) from the natural decay of 1 $mtCO_2e$ woody residue feedstock left onsite. Note that these are not emissions due to biogenic feedstock harvest or consumption, but emissions related to decay of the logging residue if left onsite. The representative values presented in Table B-1 and depicted in Figure B-1 illustrate the fraction of carbon emissions over three time frames: 20 years, 30 years, and 100 years.

Table B-1 shows that for a low decay rate of 5% loss per year, 64% of the biogenic $CO_2$ is emitted over 20 years, whereas 99% of the biogenic $CO_2$ is emitted over 100 years. However, for a high decay rate of 25% loss per year, nearly all biogenic $CO_2$ is emitted within the first 20 years.

**Table B-1. Theoretical Illustration of How the Impact of Time Depends on the Natural Decay Rate and Time Period**

| Loss/Year (decay rate) | Cumulative FCE | | |
|---|---|---|---|
| | Time Period (t) | | |
| | 20 years | 30 years | 100 years |
| **5%** | 0.64 | 0.79 | 0.99 |
| **10%** | 0.88 | 0.96 | 1.00 |
| **25%** | 1.00 | 1.00 | 1.00 |

Figure B-1 illustrates the annual and cumulative FCE, as well as the FCR, over a 100-year time frame using a 5% annual decay rate assumption.

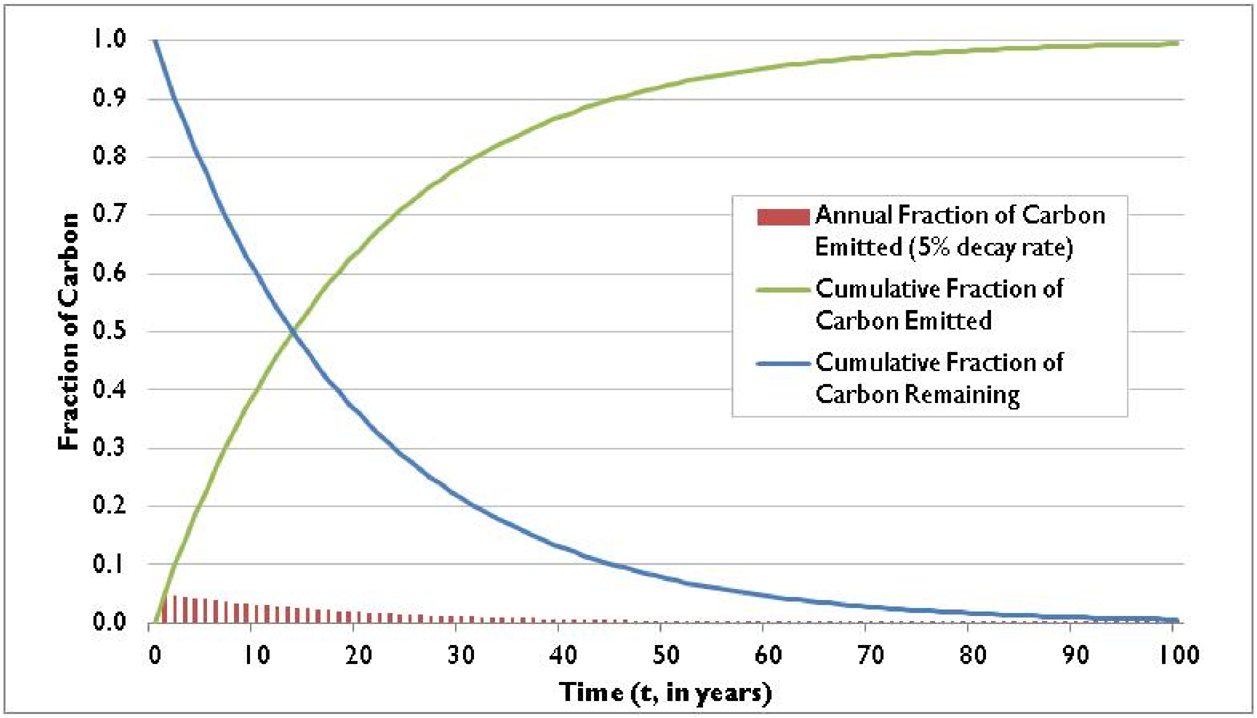


**Figure B-1. Annual Fraction of Carbon Emitted (FCE), Cumulative Fraction of Carbon Emitted (FCE), and Fraction of Carbon Remaining (FCR), Dependent on the Decay Rate and Time Period.**

## 4. Potential Methods for Assessing Multiyear Fluxes

In terms of the *BAF* equation, the assessment treatment of multiyear carbon fluxes within a prospective analysis allows for the estimation of biogenic $CO_2$ emissions associated with certain feedstocks (i.e., woody biomass) with slow rates of natural decay (in the case of residues or fallen trees) and/or long growth periods (referred to generally as "long-rotation" feedstocks). However, accurately capturing these multiyear landscape effects related to feedstock production, processing, and use can be challenging in the context of an assessment framework application that may need to estimate and report annual biogenic $CO_2$ emissions from a stationary source.

Various terms in the *BAF* equation (*AVOIDEMIT, GROW, SITETNC, LEAK*, if included, and possible losses within the *L* term) can represent biogenic $CO_2$ fluxes that have a temporal dimension longer than an annual cycle for certain feedstocks and, thus, may require application of an accounting method for these temporal effects. The *GROW* term, for example, represents the projected change in biogenic carbon fluxes from feedstock growth in a given area over a given time period.[1] The *SITETNC* term reflects estimated site-induced changes in above- and belowground carbon that typically occur over a multiyear period due to a direct land use or land use management change that triggers changes in carbon stocks. Similarly, the *AVOIDEMIT* term accounts for the avoidance of estimated biogenic emissions that could have occurred on the feedstock landscape without biogenic

[1] Note that under the retrospective reference point baseline approach in a regional application, *GROW* is calculated as recent growth in the region where the feedstock is produced and not in terms of future regrowth over time.

feedstock removal (e.g., avoided decomposition, which also may occur over a year, multiple years, or decades depending on the feedstock) or per an alternative management strategy (e.g., waste-derived feedstocks). *LEAK* represents leakage effects that can occur from feedstock production, including indirect land use changes that could affect landscape $CO_2$ fluxes for years into the future. Feedstock losses captured in the *L* term may be used to reflect decomposition of feedstocks in storage or other processing along the supply chain.

Each approach to time discussed below integrates future multiyear carbon flux values (i.e., carbon emissions and/or sequestration that occur over multiple years) into an annual accounting framework (meaning net emissions are reported/calculated annually) for illustrative purposes. Note that the need to integrate future multiyear fluxes is necessary only when a specific application of the framework allows for or requires consideration of counterfactual or future emissions fluxes related to biogenic feedstock production activities. Again, it is not necessary to integrate these forward-looking temporal elements when values for accounting terms are derived through a retrospective reference point baseline approach.

The three potential approaches for incorporating multiyear carbon fluxes into the framework are presented in this section. These concepts are for illustrative purposes and do not present an exhaustive list of how temporal aspects could be treated in a framework application. These illustrative temporal accounting approaches are (1) front loading; (2) year-to-year carryover; and (3) annualized carryover. Another approach, discounting, is discussed in a separate section below. The frontloading approach sums all future estimated net emissions associated with biogenic feedstock production and accounts for them in the time period the biogenic feedstock is used. Under the year-to-year carryover approach, emissions are tracked over time and recorded as a cumulative amount as they occur over time. Under the annualized carryover option, estimated cumulative emissions fluxes are annualized over a specific time period (which can be the time frame in which the emissions impacts are expected to occur or some other determined time frame).

The basic advantages and disadvantages associated with each of these options are discussed below. It is important to note that none of these three approaches involve discounting as presented here. This means that net biogenic $CO_2$ fluxes that occur many years in the future are treated identically as net emissions that occur in the present in all methods discussed below. However, discounting could be utilized in conjunction with any of the three approaches outlined below (the last section of this appendix discusses discounting). Lastly, the methods below include some estimation of future conditions and related emissions fluxes, which may over- or underestimate future emissions fluxes relative to actual emission fluxes trajectories that come to pass.

## 4.1. Front-Loading

With the front-loading approach, consideration is given to all the biogenic carbon fluxes that will occur over some period of time (which could be the estimated emissions horizon or some other specified period such as, for example, 20 years, 30 years, or 100 years) as a result of a particular biogenic feedstock production activity in the current time period (for example, a land use change or residue removal). Then, these emissions fluxes can be summed over time for a cumulative estimate. These fluxes are then accounted for in the current period, or period when the feedstock is used (or reported), in units of $CO_2e$ per ton of feedstock. In this way, the total carbon fluxes associated with a particular unit of feedstock production are accounted for up front, before the estimated future emissions/sequestration associated with that unit of feedstock actually occur.

Under the front-loading approach, multiyear net biogenic carbon fluxes are accounted for over a specific time frame but attributed to a single (annual or other defined reporting) time period. The approach captures all of the present and future estimated net emissions associated with growth, harvest, decay, and/or land use changes related to the biogenic feedstock production, processing, and use. Also, economic discounting could be incorporated into the front-loading approach if it is determined that future carbon fluxes should not be treated the same as current fluxes, or if discounting is appropriate in a specific policy or program application of the framework.

Figure B-2 illustrates the calculations of $FCE_t$ under the front-loading approach in the context of logging residues. For a 100-year accounting period, the front-loaded $FCE_t$ is the sum of annual $FCE_t$ values over 100 years. In this case, the front-loaded $FCE_t$ equals 0.99.

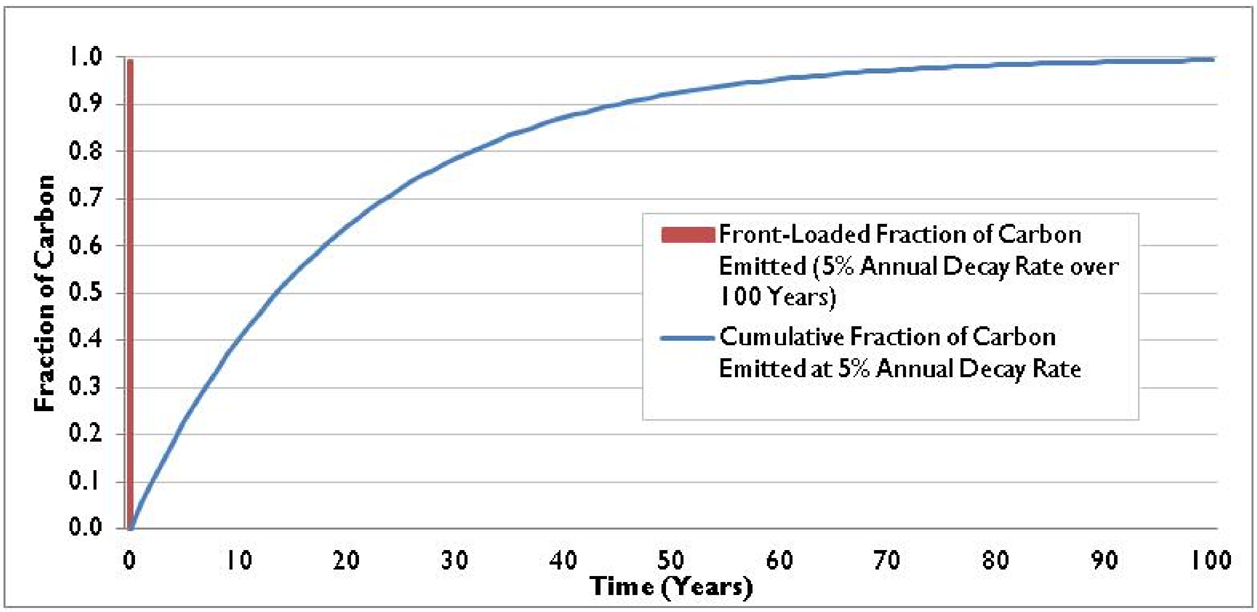


**Figure B-2. Cumulative FCE and Front-loaded FCE with a 5% Loss per Year Assumption over 100 Years[2].**

There may be policy applications or other framework applications in which the assessment horizon is shorter than the emissions horizon. For instance, the emissions horizon for certain feedstock production effects is 75 years, but the time frame for analysis is only 50 years. In such a circumstance, all the estimated future net effects may not be included in an analysis using this approach.

This basic method for incorporating temporal dynamics is relatively straightforward in that all or a portion of the estimated future net biogenic $CO_2$ emissions fluxes are accounted for in a single time

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[2] The sum of the annual FCE values over 100 Years is the front-loaded FCE over a 100-year accounting period.

step. However, there are inherent uncertainties related to future socioeconomic and biophysical projections and related trajectories of estimated net emissions fluxes related to the biogenic feedstock production and use. Also, if all estimated future emissions effects are captured in the current time period or when the biogenic feedstock is utilized, for some feedstocks this could be a relatively large assessment factor, which could discourage use of that biogenic feedstock.

## 4.2. Year-to-Year Carryover

In the year-to-year carryover accounting method presented here, the biogenic $CO_2$ fluxes associated with a unit of feedstock production in the current period are accounted for in the year in which the fluxes actually occur. For example, land use change that occurs during the production of this year's biogenic feedstock might generate a small increase in soil carbon sequestration each year for the subsequent 20 or 30 years. In this accounting approach, the accounting for the subsequent annual increment of change in emissions occurs in the year of the emissions change.

In the year-to-year carryover accounting approach, net emissions from feedstock production for a given year are reported in the same year that those emissions occur. Any net carbon fluxes carried over from feedstock utilization in previous years are also included. For example, if a feedstock removed from a site in year $t$ triggers fluxes of emissions to and from the atmosphere over subsequent $n$ years, the magnitude of the fluxes is projected $n$ years into the future. The fluxes would then be accounted for in the future, in the year ($t + 1$ year, $t + 2$ years, $t + 3$ years … up to $t + n$ years) in which they actually occur. Under the year-to-year carryover accounting approach, the emissions horizon is the same as the assessment horizon. Thus, an entity may be accounting in a given year for carbon fluxes associated with biogenic feedstocks used over multiple prior years (the number of years depends on the time frame chosen).

The carryover approach may increase the complexity of accounting requirements that would need to be implemented by stationary sources and program administrators. Under the year-to-year carryover approach, multiple terms in the framework may change from one year to the next, thereby complicating the calculations. Also, economic discounting could be incorporated into year-to-year carryover if future carbon fluxes should not be treated the same as current fluxes or if discounting is appropriate in a specific policy or program application of the framework.

Figure B-3 illustrates the annual $FCE_t$ year to year over a 100-year time frame using assumptions of 5, 10, and 25% emissions per year in the case of logging residues. The annual FCE is calculated by subtracting each year's $FCR_t$ value from the previous year's $FCR_t$ value. As an example using a 5% loss per year, in Year 1, 95% of the carbon is remaining and is subtracted from the prior year (100%), which gives 0.05 as the annual $FCE_t$ in Year 1. The representative values depicted in Figure B-3 illustrate that the annual $FCE_t$ in a particular year depends on the actual time profile (i.e., decay rate) of the emission pulse.

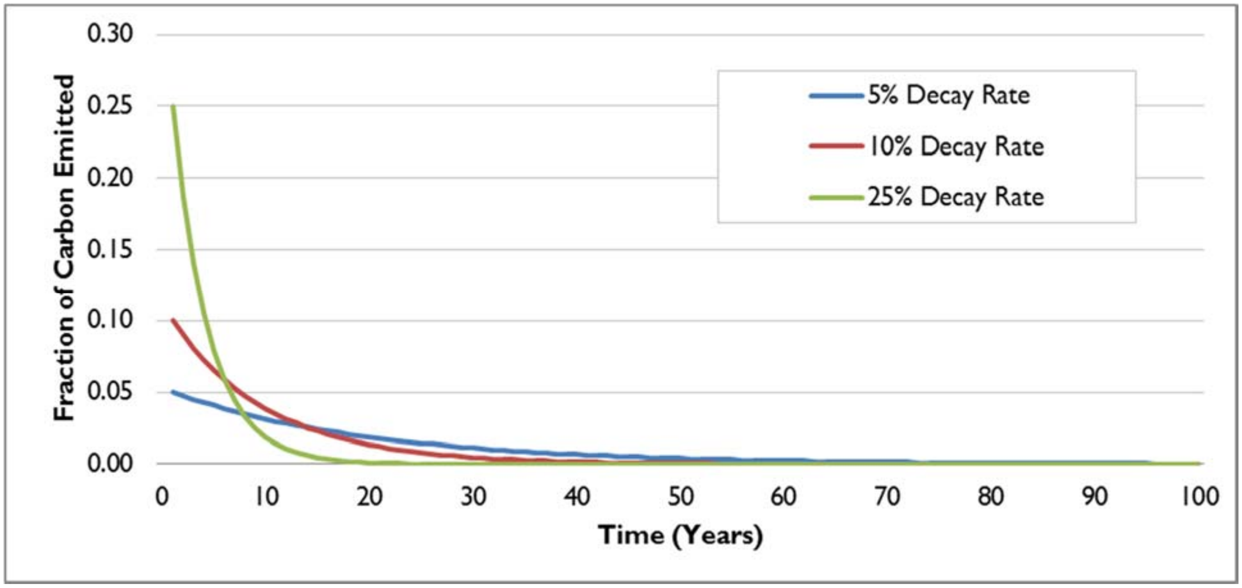


**Figure B-3. Year-to-Year Annual Fraction of Carbon Emitted (*$FCE_t$*) Depends on the Decay Rate.**

This method could allow for future estimated biogenic $CO_2$ fluxes related to the use of a feedstock to be reflected in the values for framework equation terms as they occur on the landscape rather than during the year of feedstock use. Also, this method permits updates to future trajectories of estimated emissions fluxes related to the biogenic feedstock production and use in case initial estimated trajectories prove to differ from actual emissions flux trajectories. However, the values of framework equation terms for a given year's feedstock use may change over subsequent years, which may cause market and investment uncertainty (the *BAF* can be applied only to the annual emissions from a stationary source in a given year, which can vary).[3] As a result, adjustment of future-year stationary source biogenic $CO_2$ emissions may not capture and represent the actual net emissions impact (on a tonnage basis) of future-year carbon fluxes related to previous-year feedstock consumption.

## 4.3. Annualized Carryover

The annualized carryover approach accounts for cumulative emissions over the emissions horizon and then divides those emissions equally over the assessment horizon. Thus, values for future estimated annual net emissions are equal across the assessment horizon and are determined by the annualized value. Depending on the dynamics of the biogenic $CO_2$ processes on the landscape, annualized carryover may over- or underestimate the fluxes at the start of the accounting period compared with year-to-year carryover accounting. The illustrative examples of annualized carryover in this appendix do not include economic discounting. However, economic discounting could be incorporated into this approach in applications of the framework where future biogenic $CO_2$ fluxes were not be treated the same as current fluxes. It is possible that a specific policy or

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[3] If both the *BAF* and emissions varied each year, then these two factors introduce uncertainty into the annual emissions estimate, making it difficult for a stationary source to have stability for investments.

program application of the framework would discount future biogenic $CO_2$ fluxes (discussed in Section 5).

Under the annualized carryover approach, the values of future estimated annual net emissions for each year's feedstock production are the same. The time-related values for relevant *BAF* equation terms would remain the same over time (or until recalculated based on new reference data) and provide simplicity for application of the *BAF* equation. However, because future net emissions effects related to a current year's consumption of feedstock are accounted for in future years, applied accounting complications could arise. For example, as with year-to-year carryover accounting, future fluxes related to previous years' feedstock consumption would need to be applied in each year when calculating a stationary source's *BAF* related to the use of a feedstock. If the stationary source changes ownership or operating status, properly transferring the accrued future emissions accounting values related to past feedstock consumption may prove complex.

To illustrate these dynamics, Table B-2 presents the annualized $FCE_t$ over a 100-year emissions horizon for a representative multiyear carbon flux related to forest residue decay, with different percentage carbon loss assumptions and different assessment horizons. To calculate annualized $FCE_t$ for a 100-year emissions pulse, cumulative emissions up to 100 years were divided by 20-, 30-, and 100-year time periods, respectively (e.g., annualized $FCE_t$ for a 5% decay rate over a 20-year assessment horizon is 0.99 divided by 20, which equals 0.05).

**Table B-2. 100-Year Emissions Annualized over 20-, 30-, and 100-Year Assessment Horizons.**

| Loss/Year (decay rate) | Annualized FCE (100-year emissions) | | |
|---|---|---|---|
| | Time Period (t) | | |
| | 20 Years | 30 Years | 100 Years |
| **5%** | 0.05 | 0.03 | 0.01 |
| **10%** | 0.05 | 0.03 | 0.01 |
| **25%** | 0.05 | 0.03 | 0.01 |

Table B-3 presents a truncated annualizing approach where the emissions horizon is truncated at 20, 30, and 100 years. The cumulative emissions after 20, 30, and 100 years are then divided equally over the same time periods. Under the truncated approach, not all of the estimated emissions are captured, and the assessment horizon is the same as the truncated emissions horizon (20, 30, and 100 years in this case). These time periods were chosen to represent different assessment horizons (e.g., facility lifetimes) that could be applied in practice. For example, the annualized $FCE_t$ for truncated emissions at 20 years for a 2% carbon decay rate is 0.33 divided by 20, which equals 0.02.

**Table B-3. 20-Year, 30-Year, and 100-Year Emissions Annualized over 20-, 30-, 100-Year Time Periods, Respectively.**

| Loss/Year | Annualized FCE [truncated emissions] | | |
|---|---|---|---|
| | Time Period | | |
| | 20 Years | 30 Years | 100 Years |
| **5%** | 0.03 | 0.03 | 0.01 |
| **10%** | 0.04 | 0.03 | 0.01 |
| **25%** | 0.05 | 0.03 | 0.01 |

The representative values in Table B-2 illustrate that in determining appropriate emission annualized values, it is important to consider both the emissions horizon for the feedstock effects as well as the assessment horizon for the reporting of those emissions. Specifically, annualized emissions increase as the emissions horizon increases; for example, under a 5% decay rate the non-truncated annualized $FCE_t$ for a 100-year emissions horizon and 20-year assessment horizon (0.05) is greater than the truncated annualized $FCE_t$ for a 20-year emissions horizon and 20-year assessment horizon (0.03). However, as the assessment horizon increases, annualized emissions decrease: for example, under a 5% decay rate the non-truncated annualized $FCE_t$ for a 100-year emissions horizon and 100-year assessment horizon (0.01) is less than the non-truncated annualized $FCE_t$ for a 100-year emissions horizon and 20-year assessment horizon (0.05).

This method for accounting for time allows for inclusion of all emissions fluxes over the emissions horizon within the assessment horizon. Also, similar to the year-to-year carryover approach, this method can allow updates to future trajectories of estimated emissions fluxes related to biogenic feedstock production activities and use in case initial trajectories prove to differ from actual emissions flux trajectories. However, similar to the year-to-year approach, framework equation term values for a given year's feedstock use may change over subsequent years, which may cause market and investment uncertainty. The *BAF* can be applied to the annual emissions from a stationary source in a given year, which can vary.[4] As a result, adjustment of future-year stationary source $CO_2$ emissions may not capture and represent the actual net emissions impact (on a tonnage basis) of future-year carbon fluxes related to previous-year feedstock consumption.

## 4.4. Temporal Scale of the Illustrative Future Anticipated Baseline Approach in the Technical Appendices

When using a future anticipated baseline, integrating time into the assessment of forward-looking phenomena is inherent in the approach, and decisions about temporal dynamics may affect the outcomes (as discussed in the previous subsection). The future anticipated baseline approach as generally discussed in this report could conceptually apply whatever future time horizon is necessary for the specific program or policy analysis at hand. This report does not apply the framework to specific policies or programs and thus has no specific temporal parameters such as

[4] If both the *BAF* and emissions varied each year, then these two factors introduce uncertainty into the annual emissions estimate, making it difficult for a stationary source to have stability for investments.

an assessment horizon or time of reporting. For illustrative purposes in the technical future anticipated baseline appendices of this report (Appendices J, K, and L), the year-to-year carryover is applied using a 50-year simulation horizon. This assessment time scale is long enough to capture significant carbon dynamics of longer rotation feedstock species, land use and land use management changes, and soil carbon pools. Conversely, it is short enough to detect significant biogenic $CO_2$ fluxes related to biogenic feedstock production and harvest. The year-to-year carryover approach is used to show how estimated future net biogenic $CO_2$ emissions fluxes could change over time and to provide insights about the potential future impacts of biogenic feedstock production, processing, and use. In addition to annual accounting using the year-to-year carryover approach, one can also use this approach to evaluate cumulative emissions for a specific time horizon. Additional discussion of periodic (flux based) and cumulative landscape emissions projections using the year-to-year carryover approach can be found in Appendices K and L.

# 5. Discounting and Its Relevance to the Framework

Broadly speaking, there is a value to time. For example, benefits and costs are typically valued higher if they are experienced sooner (OMB Cir A-94). This value of time is usually discussed as a "discount" of what the future holds. Discounting is regularly applied in finance and economics, where it represents the time value of money, and quantitative values can generally be assigned. Discounting allows for assessment of the future value in today's terms (i.e., the *net present value*). To compute net present value, it is necessary to discount future benefits and costs. The discount rate is the interest rate used in calculating the present value of expected yearly benefits and costs (OMB Cir A-94).

For example, money invested today will accrue interest, and the quantity of money will grow over time according to the interest rate. Similarly, a debt will increase over time according to the interest rate. Money received today has more value than the same amount of money received in the future. If the interest rate is known, the net present value of future costs (e.g., the monetary value of building and maintaining seawalls) can be calculated, as can the future value of benefits (e.g., the monetary value of homes and tourism on the seashore). In other words, the net present value of future costs and benefits can be calculated by multiplying the costs and benefits in each future year by a discount factor, then summing all values over the lifetime of an investment, policy, or decision.

Discounting the value of damages associated with GHG emissions, which span multiple generations, is particularly complex and raises difficult and controversial questions of science, economics, philosophy, and law. The U.S. federal government reviewed the literature on intergenerational discounting several years ago when developing estimates of the social cost of carbon, i.e., the monetized value of damages associated with a marginal change in $CO_2$ emissions. The federal government found that although it is well understood that the discount rate has a large influence on the current value of future damages from GHG emissions, there is no consensus about what rates to use in this context.

Recognizing the lack of consensus about an appropriate intergenerational discount rate and uncertainty regarding how interest rates might change over time, the federal government selected three rates to span a plausible range of certainty-equivalent constant discount rates: 2.5, 3, and 5%

per year. In sum, average returns on longer-term investments were used to inform selection of certainty-equivalent discount rates. The federal government viewed this approach as defensible and transparent given its consistency with current benefit-cost analysis principles as well as OMB's guidelines for such analysis as embodied in OMB Circular A-4. The Technical Support Document, *Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866*, discusses this analysis in detail (Interagency Working Group, 2010).

The federal government has continued to research alternative approaches for intergenerational discounting. In particular, a group of world-recognized experts convened at an EPA-funded workshop in 2011[5] to explore what principles should be used to determine the rates at which to discount the costs and benefits of regulatory programs when costs and benefits extend over very long horizons. The charge questions that were the subject of the workshop discussion focused on three main areas: (1) whether and in what context it is appropriate to apply a Ramsey discounting framework in an intergenerational setting; (2) whether and how to directly estimate discount rates over long time horizons; and (3) how to apply discounting in a regulation where some costs and benefits accrue intra-generationally while others accrue inter-generationally. Notably, the group reached consensus that there are compelling arguments for using a declining discount rate schedule, though determined that practical questions remain regarding how to establish and implement such a schedule (Arrow et al., 2013).

Discounting is more challenging when applied to nonmonetary quantities where there is not a clear interest rate, thereby making it difficult to quantitatively equate present and future events. If the discount rate is known in the context of avoiding future climate change impacts, the net present value of future costs (e.g., the monetary value of damages associated with climate change impacts) can be calculated, as can the future value of benefits (e.g., the monetary value of avoided damages or avoided GHG emissions.) Also, if carbon emissions have monetary value as determined through a carbon tax, a cap-and-trade system, an emissions limit or permit system, or through the structure of the damages caused, then quantitatively discounting the value of emissions is more straightforward. However, discounting becomes more challenging when the quantitative links between physical emissions and costs or benefits are less clear.

The traditional role of discounting is to compare the costs and benefits of quantities (such as money or the monetary value of $CO_2$ emissions) that occur at different periods in time. The higher the discount rate, the lower the present value of the future unit (money, carbon etc.) in the future. This means that a high discount rate implies a strong time preference, such that events in the future, for example, are given far less value than those occurring today. Failure to discount future events assumes a discount rate of 0 and implies no time preference; that is, a 0 discount rate assumes that future events have the same value as current events. For carbon accounting, the fundamental issue is whether carbon emissions (or sequestration today) are valued the same as carbon emissions (or sequestration in the future), and how the valuation of time is factored into carbon accounting. For example, if one ton of $CO_2$ is emitted this year and one ton of carbon is sequestered 20 or 100 years from now, the treatment or valuation of time will determine if these events are of equal and

[5] Link to workshop summary: http://rff.org/Events/Pages/Intergenerational-Discounting-Workshop.aspx

opposite value so that the net effect is 0 or not. It is clear that time is important, but the challenge lies in how to deal with this preference quantitatively.

## 5.1. Time Preference in $CO_2$ Emissions

As mentioned above, one of the current challenges in carbon accounting is the time value of carbon emissions (or sequestration). Do emissions at some time in the future have the same value as emissions now? Does the time path of emissions and sequestration matter? Is there value in delaying emissions? Is there value in temporary storage of emissions if they will be released later? The importance of the time value of carbon has been recognized for many years (e.g., Richards, 1997), but there continues to be much debate on how to deal quantitatively with time and what the "appropriate" discount factor is in the context of monetizing future GHG emissions. A recent advisory group to the California Air Resources Board struggled with this topic without reaching consensus but did provide the consensus statement that "the timing of emissions [is] important and, as a general goal, policy should differentiate based on timing where possible" (Martin, Kloverpris, Kline, Mueller, & O'Hare, 2011, p.48). The group also concluded that there is "no intellectually supportable escape from the universally demonstrated judgment of society that consequences occurring at different times must be valued with reference to the time of occurrence," but the group acknowledged the difficulty of determining appropriate discount rates (Martin et al., 2011, p.27). Similarly, an EPA (2010a) publication on economic analyses discusses approaches for dealing with time without ending up with a quantitative conclusion but recommends that analyses "display the time paths of benefits and costs as they are projected to occur over the time horizon of the policy…"

The prevailing view is that physical carbon flows should not be discounted as a function of time but that—where carbon flows have economic value—the monetary value of the flows should be discounted. As O'Hare et al. (2009) wrote in a paper on their view of the proper accounting for time in biofuels analyses, "the discounting model applies to costs and benefits, not to physical phenomena that generate them, unless their economic value is otherwise stable over time" (p. 3) and "before such economic analysis can be meaningfully pursued the relationship between the physical and economic quantities must be established" (p. 4). If carbon emissions were currently subject to taxation, for example, the tax rate would be the economic value of reducing (or avoiding) emissions and possibly used as a discount rate in net present value calculations. The concept of applying a discount to a physical measure, however, is difficult to rationalize: a ton of carbon is a ton of carbon, and differences arise only from its equated economic value.

Any program or policy that considers effects of carbon emissions over time will need to decide on the applicability of valuing these emissions and, if done monetarily, how to discount them. One recent example of this decision-making process can be found in the Renewable Fuel Standard Program (RFS2) Regulatory Impact Analysis (EPA, 2010b). When considering how to measure the lifecycle GHG emissions from a given type of renewable fuel relative to a 2005 petroleum baseline, two important elements were considered in terms of how to estimate the stream of emissions and benefits over time: (1) the time period considered and (2) the discount rate applied to future emissions. Although a range of options was considered in the proposed rule, for the final rule EPA chose a 30-year time period and a 0% discount rate. Although a relatively short time period of 30 years was chosen because it was similar to the life span of a biofuels-producing facility, a discount rate of 0% was chosen "due to the many issues associated with applying an economic concept to a physical parameter" (p. 423). This is primarily because the Energy Independence and Security Act (EISA) of 2007 did not establish any monetary valuation of carbon emissions for the RFS2 program, as well as the "lack of consensus as to the appropriate discount rate to apply to GHG lifecycle emissions streams through time" (EPA, 2010b, p. 423).

The peer review report *Methods and Approaches to Account for Lifecycle Greenhouse Gas Emissions from Biofuels Production over Time* (EPA, 2009) is particularly direct in its opposition to discounting physical emissions, stating that "all reviewers noted in some way that a discount rate should only be applied to a monetary unit, rather than a physical unit such as a carbon emission." Similarly, "proper discounting ... can only be conducted on value (i.e., damages, not physical quantities such as emissions)" (p. B-2) and "discount rates are only justifiable when applied to monetary impacts, not physical impacts" (p. 13). Further, "economic discounting cannot logically be applied to physical quantities such as GHG emissions, only to economic quantities such as climate change damages" (p. B-4). Similarly, Martin et al. (2011) wrote that "in the absence of agreement on…values, discount rates become meaningless" (p. 27) and that when considering discounting, "a prerequisite is to begin with a monetized value to discount" (p. 26). Other sources, such as the Interagency Review on Social Cost of Carbon (Interagency Working Group, 2010) and Johnson and Hope (2012), do not address the concept of discounting physical emissions and focus on damages, costs and benefits, or other concepts of monetized value.

Although the literature is generally opposed to the concept of discounting physical emissions, some sources do discuss related instances where the strategy may be applicable. First, as noted by a few respondents in the peer review report mentioned above (EPA, 2009), discounting physical emissions may be appropriate if these emissions are used as a direct proxy for damages. Discounting is "justifiable if physical emissions were being used as a proxy for economic damages associated with warming" (p. 15) and only in this case are discount rates used for physical carbon units "analogous to monetary discount rates" (p. 22).

A number of recent efforts have attempted to describe a time-dependent damage function for emissions, that is, efforts to link emissions to atmospheric concentrations and subsequently to the climatic effects (damages) of increasing concentrations. This approach encompasses more than a time preference, because it can include recognition of the dynamics of changing marginal damages over time (i.e., the notion that the climate impact of one ton of $CO_2$ emissions today is not equal to the impact of one ton of emissions in the future because of factors such as the persistence of GHGs in the atmosphere, options for mitigation, or damages that are a function of the total level of atmospheric $CO_2$ at the time). Whereas traditional time preference should result in a decreasing importance of future emissions, equating emissions with damages could result in increasing importance of future emissions if the damage function is increasing faster than the rate of time preference (see, for example, Richards, 1997). As characterized by Marshall (2009), "Ideally, a GHG accounting method … should explicitly analyze the expected damage associated with flows over time. The corresponding monetary units associated with this damage can then be discounted to determine how the impacts of future flows compare to those of the present." Fargione wrote that "if

EPA is not willing to make assumptions about the relationship between emissions and damages, then they should not use any discounting" (EPA, 2009, p. B-4).

Papers by O'Hare et al. (2009) and Cherubini et al. (2011), for example, calculate cumulative radiative forcing (described by O'Hare et al. as "a physically plausible proxy for the total damage to the planet from the $CO_2$ emissions") or GWPbio (defined by Cherubini et al. as "the effective climate impact") in efforts to describe a damage cost that reflects the time path of $CO_2$ emissions. Similarly, Kendall et al. (2009) propose a "time correction factor" to "properly account for the timing of … greenhouse gas emissions in the biofuels life cycle" (see also Alissa Kendall & Price, 2012). Levasseur et al. (2010) describe a dynamic life-cycle analysis that considers the time value of emissions. Conceptually, discounting marginal damages is related to traditional discounting in that it makes assumptions about changing values over time, but in this case, the "value" is expressed in terms of the impact on climate.

Ultimately at least three factors enter into considering the time dependence of the value of carbon flows: (1) the monetary values potentially captured in cost-benefit analyses (as discussed above); (2) the existence of irreversibilities or tipping points (see, for example, Kolstad, 1994); and (3) the role of learning (see, for example, Kolstad, 1993). On tipping points, Marshall (2009, p.9) wrote, "the potential for irreversible change is one of the significant determinants of the expected damage function for GHG emissions that must be considered in determining how to compare current to future emissions, and is one of the most convincing arguments for the need to make some sort of distinction between current and future … emissions." Kolstad (1994) includes the investment capital of mitigation measures as an irreversibility. On the role of learning, Kolstad (1993) notes the role of uncertainty in the relative value of current and future emissions and concludes that "accelerated learning tends to reduce current period optimal emissions." That is, rapid reductions in uncertainty tend to reduce, but not eliminate, expenditures to reduce current emissions as uncertainty is being resolved. Dornburg and Marland (2008) raise many of these issues in the context of the value of temporary carbon sequestration or of delaying emissions.

Uncertainty becomes a dominant factor in attempting to discount future emissions (or sequestration) when significant time intervals are involved in lifecycle analyses or the impacts of land use change. Despite recognition of the importance of dealing with the time value of $CO_2$ emissions, there is great uncertainty in the appropriate value of a discount rate. This uncertainty is due to uncertainty about the future, uncertainty about the correct relationship between emissions and damages, and the potentially long times involved in consideration of climate change impacts. It is clear that application of constant discount rates is not appropriate over long time periods (e.g., intergenerational times) (see, for example, EPA, 2009; Schelling, 1995). There is the suggestion that for consideration of long time periods it may be appropriate to use discount rates that decrease with time (see, for example, Guo, Hepburn, Tol, & Anthoff, 2006). Note that the imposition of any time horizons (as done with traditional measures of global warming potential) to limit consideration of effects after a specific period of time implicitly assumes that the discount rate increases to 100% and that impacts after that time are not counted at all.

Ultimately, O'Hare writes (personal communication, 2012), "at least in the short and medium term, something like compound discounting at a rate in the 3–7% range is necessary to rational decision making about any actions with consequences that occur in the future. This discounting must be applied to something like the social cost and not mere quantities of discharge." Richards (1997) suggested that "at a minimum, carbon discount rates should be tested for values equal to the social discount rate and zero." In 2009, Richards (in EPA, 2009) suggested discount rates of 2%, 3%, and 5%. The specific discount rate chosen depends on the circumstances. Although nearly all individuals possess a time preference, the strength of this preference can vary greatly and with it the corresponding discount rate. In the realm of policy making and finance, the selected discount rate is often simply the market interest rate, which generally fluctuates between 2% and 7%. Considering the long time horizons associated with climate change and climate change policy, small changes to the discount rate can have very large consequences. A widely cited report on the costs and benefits of climate mitigation strategies and published responses that criticize its use of very low discount rates illustrate the large impact of discount rates over long time periods (see Nordhaus, 2007; Stern, 2006).

Note that the decision to ignore time is in effect a decision to assume that the value of emissions is not affected by the time path of emissions and that the appropriate discount rate is 0. Marland et al. (2010), in the context of the carbon stored in durable wood products, showed that where discounting of carbon flows is implemented, it is very important to represent the time path of $CO_2$ emissions as accurately as possible.

There is much discussion and uncertainty about appropriate rates for compounded discounting, but at the same time there is a widespread consensus that the time value of carbon emissions is important. Specifically, as Richards wrote in 1997, "the time value of carbon is an important issue that requires an explicit decision." Writing in 2009, Richards added "if it doesn't matter when it is done, it doesn't matter whether it is done" (EPA, 2009, p. F-2).

## 5.2. Discounting Summary

The production and use of biogenic feedstocks for energy can in some circumstances have emission implications extending well into the future. Questions then arise about whether and how to value emissions fluxes that occur over time in present terms. Although there is no single, scientifically correct treatment of time, the choice of treatment may have significant impacts on the results of an accounting framework application. It is important to consider possible treatments of time and the implications of different treatments in terms of the respective strategies chosen for long-term and short-term emission accounting.

The prevailing view in the technical literature is that there is a value of time that can have important ramifications for prospective accounting and analysis, that it ought to be considered explicitly, and that time preference is traditionally viewed as related only to monetary or other values and is not inherent in physical measures of carbon emissions. Aside from certain financial transactions where there is an explicit discount rate (the interest rate), it can be difficult to determine an appropriate discount rate for any given circumstance, including accounting for GHG emissions over time. The debate continues about how to value (i.e., what discount rates to choose) when evaluating the future value of biogenic $CO_2$ emissions, where the impacts on the global carbon cycle may occur over very long periods of time and the impact of small changes in discount rate can be very large. The scientific literature does not provide guidance on selecting one appropriate discount rate but does suggest using multiple values to illustrate the great importance of time.

The decision on how to treat the time value of biogenic $CO_2$ emissions (or sequestration) will likely fall to policies or programs like a carbon tax, a cap-and-trade system, or other legal decisions that deal with society's willingness to consider the inherent risks of a changing climate. The decision to not discount the value of emissions over time is an effective decision to select a discount rate of 0. For the purposes of accounting for biogenic $CO_2$ emissions from stationary sources, the framework application in this report focuses primarily on the physical flows of biogenic $CO_2$ and, in the forward-looking context, the comparison of different potential flows across alternative future scenarios. Applications of the framework could incorporate discount rates into calculations of the biogenic assessment factor as appropriate for that specific application.

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# Appendix C: Spatial Scale

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## 1. Introduction

This appendix discusses the importance of spatial scale choice when applying the assessment framework. Spatial scale selection can affect the results of any analysis evaluating GHG emissions and sequestration, regardless of whether that analysis applies a retrospective reference point baseline approach or a future anticipated baseline approach (Galik and Abt, 2012). A range of options for choosing an appropriate spatial scale is explored, along with a discussion of the advantages and disadvantages of each option. This appendix then lays out the technical underpinnings for the use of specific regions for calculating illustrative biogenic assessment factor (*BAF*) equation term values using the retrospective reference point and future anticipated baseline approaches. These regional constructs include: the Resources Planning Act[1] regions (8 regions) are used for the retrospective reference point forest-derived feedstock examples, and the agro-forestry regions used in the U.S. Forest and Agriculture Sector Optimization Model with Greenhouse Gases (FASOM-GHG) (11 regions) are applied for retrospective reference point agricultural feedstock examples and future anticipated baseline agriculture and forestry feedstock examples. Additional information on reference point baseline and future anticipated baseline methods can be found in Appendices H and K. Spatial scale considerations in the composition and management of waste-derived feedstocks are briefly discussed here in Section 2.3.4 and in detail in Appendix N.

# 2. Considerations and Implications of Spatial Scale Choice

Different spatial scales offer different levels of precision in terms of estimates, can affect depth and breadth of measurement, dictate the availability and verification of data, and limit modeling options. The size of an assessment area can also determine the ability of an assessment system to reflect carbon dynamics in the biogenic feedstock source area and any inter-regional trade of biogenic feedstocks. The choice of spatial scale can allow for broad aggregation of or, conversely, evaluation of differences between, characteristics of the land base (e.g., ownership type, management regimes, soil types), biophysical characteristics of the biogenic feedstock (e.g., species and growth and harvest rates), and feedstock production and market dynamics.[2]

Therefore, it is crucial that any application of the assessment framework consider these trade-offs and implications on results when identifying the most appropriate spatial scale (or scales if more than one scale is appropriate) for use in a particular program or policy. In general, there is no single scientifically correct option or specific method for determining the "appropriate" spatial scale for all analyses: the appropriate spatial scale differs depending on the specific goals and parameters of a specific policy or program application of the framework. The issues related to spatial scale can differ with feedstock type, biophysical and economic factors, and the circumstances for each program that needs to assess biogenic feedstock production and use. Thus, the choice of spatial scale is primarily a function of the stated objective of the specific program or project being developed.

The source of the biogenic feedstock is an important consideration because the biophysical attributes of the biogenic feedstock and land on which it is produced are used to derive input values for use in the framework (reflected as values within the *BAF* equation in the main report Part 2). The biophysical attributes of different biogenic feedstocks can vary between geographic locations because of a number of environmental factors and net primary productivity of the landscape (Beringer et al., 2011). Therefore, unless the global landscape were entirely homogenous, it would be inaccurate to assume the biophysical as well as feedstock production dynamics in one part of the country or world are the same as those in another without evaluation.

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[1] For more information on the USDA Forest Service Resources Planning Act (RPA) Assessment, see www.fs.fed.us/research/rpa/.

[2] Leakage, represented by the *LEAK* term (see Appendix F), can also be influenced by the spatial scale chosen in a specific policy analysis. For additional information on leakage, refer to Appendix E.

Before discussing the range of different spatial scales and related tradeoffs and implications of each, the subsections below first discuss how land base characteristics and geographic location, data availability, and data accuracy can affect the choice of spatial scale.

## 2.1. Location and Land Base Characteristics

Location of feedstock production dictates important biophysical factors such as temperature, soil type, precipitation, elevation, species type and mix, and growth rates (Eagle et al., 2010). As such, the geographic location of the landscape not only determines, in part, what type of feedstocks can be produced but also the various biogenic $CO_2$ fluxes associated with feedstock production.

Ownership type also plays a role in determining what feedstocks are grown, how often they are harvested, and related GHG emissions fluxes. For example, although the ratio of forest growth to removals (of woody biomass) at the national scale is roughly 1.72 currently (Smith et al., 2009), it varies substantially with geographic region, species, and ownership. For example, the ratio of forest growth to harvest for private forests in the conterminous United States is 1.3, while the same ratio on public lands is 5.3 (DOE, 2011). An assessment area that includes a large proportion of publicly owned land would therefore be more likely to have lower levels of harvest (and higher levels of growth) than a similar area with more private land ownership (DOE, 2005, 2011). A more detailed discussion of working lands is covered in Appendix H.

Given the transaction data collected and processed when a biogenic feedstock enters a stationary source, it is often possible to determine the precise location of the feedstock harvest site, though it may only be possible to know the broad geographic origin. For example, entities operating primarily on long-term procurement contracts will likely use the same feedstock production sites year after year, and the geographic location of those sites can be known. In such cases, measurement and analysis of production-related biogenic fluxes at a localized scale are possible. On the other hand, for stationary sources operating using aggregated feedstocks (e.g., agricultural residues from multiple landowners piled together at centralized site) or feedstocks that require storage and may become mixed (e.g., forest logging and milling residues), it may be difficult to know the precise origin of the feedstock, so only the broad geographic region could be identified. Also, for some feedstocks, production sites may vary from year to year (e.g., logging residues from harvests that may not return to the same location for decades or crop rotations and fallow cycles).

## 2.2. Data Availability and Accuracy

The choice of spatial scale can be greatly influenced by the availability and accuracy of data and the precision with which one can model feedstock production and market dynamics. When a stationary source purchases biogenic material for energy production, it is possible to measure every ton of material that is purchased or brought into the stationary source, or subsequently used in a particular process at the stationary source (e.g., using measurement equipment such as scales and monitors). However, when estimating the biogenic resource in a production source area, it is necessary to use sampling approaches, which are inherently less precise than complete measurements due to sampling and measurement errors. For example, to estimate woody biomass in the forests of a region, trees on inventory plots (samples) are measured periodically (FAO, 1997;

USDA, 2014). Tree measurements (e.g., species, diameter, height) are used in conjunction with mathematical models to estimate biomass per tree and then statistically expanded to obtain estimates of biomass per unit area of forest (FAO, 1997; USDA, 2014 ). Remote sensing approaches (e.g., satellite imagery, aerial photography) are also used to estimate the area of forest cover within a region (FAO, 1997; USDA, 2014).

The level of data accuracy varies with choice of spatial scale. When carbon stocks are estimated at a larger spatial scale (e.g., national, regional) through statistical sampling, the increase in sample size provides more precision (i.e., smaller sampling errors). For smaller land areas, the estimates will be less reliable due to a lack of statistical power associated with small sample size (Westfall et al., 2013). Estimates at these smaller scales must then be derived from other sources such as special inventories or surveys (i.e., thorough inventories conducted as part of a forest management plan).

In addition to primary data collection for retrospective analysis of landscape emissions, geo-referenced land cover and forest inventory data often serve as a primary input to economic models that can be used to project landscape biogenic emissions relative to an anticipated baseline. Thus, models that aggregate land use data to a larger region will reduce the uncertainty associated with those primary model inputs.

#### 2.2.1. Cross-boundary Flows

Another difficulty introduced by defining geographic boundaries for analysis is assessment for transfers across political boundaries (e.g., cross-state or international trade). For example, it is common for wood-using mills in one state to purchase wood from across state or regional boundaries (Teeter et al., 2006). As a result, the emissions from biogenic feedstock consumption for energy production may occur in a different region than the sequestration in the forest-derived feedstock production area. In an assessment framework, transportation across accounting boundaries introduces complexity in that feedstocks of the same type (e.g., trees) acquired from different areas or regions may be accounted for separately as they may have different biophysical attributes (e.g., species, growth rates). Thus, entities using biogenic feedstocks, or another party designated with this responsibility by a program/policy, would need to anticipate and/or monitor the source region for all feedstocks a facility uses to account for regional differences. The data collection and modeling complexities will increase with the number of regions defined in a geographically divided assessment framework.

Further, it may not be possible to determine the specific origin of all biogenic feedstocks. In the context of forest-derived biomass, even if the specific site is known, source locations would change annually because of the long-term nature of forest harvesting cycles. For agricultural feedstocks, it may be possible to know the specific locations that supply biomass to a procuring entity. In other cases, aggregators or suppliers may purchase material from a variety of sources, and knowledge of specific origins of feedstock may be lost. When the biogenic feedstock production location is known, it is possible to collect very detailed site-specific data, although this may be costly to collect and verify.

In addition to inter-regional considerations, international feedstock trade flows are important to acknowledge as well. International feedstock production and the imports of those feedstocks can significantly affect overall U.S. biogenic feedstock resource availability and demand pressures on those resources. The pricing and flow of feedstocks and related commodities have the potential to influence domestic supply chains and land use activities. The report acknowledges the significance, but does not include assessment, of international biologically based feedstock production and the role of imports and exports (i.e., the impacts of U.S. feedstock production on international trade flows and resource allocation). Deciding whether to include and therefore craft a means to account for imported and exported biogenic feedstocks would be a decision specific to application of this framework in the context of a particular policy or program requirements and objectives.

## 2.3. Range of Potential Spatial Scales and Related Implications and Tradeoffs

For purposes of this framework, several spatial scale options were considered: stand/field, fuelshed, state, regional, and national scales. The ordering here is generally in the direction of increasing size; however, there could be instances in which a fuelshed (or woodshed) area may be larger than, for example, an individual state (e.g., a small state such as Rhode Island). Furthermore, some scales may approximate an aggregate of other scales, such as multiple states combining to form one region.

### 2.3.1. Stand or Fuelshed

The finest spatial scale would be at the specific site of the biogenic feedstock origin (agricultural field, forest stand, etc.). The linkage between feedstock source area carbon dynamics and the net biogenic emissions from an entity using biomass is most direct at finer spatial scales. Accounting at the stand or field level directly links emissions and sequestration on the landscape producing a biogenic feedstock, and the impact of each entity's biogenic feedstock use on the biogenic production site carbon fluxes could be determined. However, an assessment using the reference point baseline approach at these small scales can be challenging because data would need to be collected for every site from which a stationary source procures feedstocks (e.g., feedstock tracking, record keeping), and these data must accompany the movement of the feedstocks around the country. An assessment that uses an anticipated baseline approach would also be difficult, but one could model production systems rather than tracking each production plot.

Next may be an aggregate of areas from which feedstock may be procured for use at a specific entity: the fuelshed.[3] When the location of feedstock production sites is known, the fuelshed can also be known because it would be the aggregate of sites from which feedstocks originate. In the case of unknown source locations, one might be able to generalize a fuelshed into a region encompassing local and likely sources. For example, several analyses have used a circular fuelshed with either a straight-line or road-distance radius to model the impact of increased forest-related feedstocks relative to business-as-usual conditions (50 miles straight-line: Galik and Abt [2012]; 30

[3] Fuelshed is defined as an aggregate of areas from which feedstock may be drawn for a specific facility.

miles road distance: Brinkman and Munsell, [2012]). Thus, fuelsheds are specific to stationary sources procuring biogenic feedstocks, but fuelshed areas for multiple facilities could overlap, and this could change over time as supply and market dynamics change, capital depreciates, and new facilities are built.

An approach at a comparable scale to fuelsheds might be a fixed geographic region that approximates the area of a fuelshed. For example, Galik and Abt (2012) note that 50-mile radius fuelsheds approximate the area of USDA Forest Service Forest Inventory and Analysis (FIA)[4] survey units, which are fixed regions (aggregates of counties) defined to provide forest inventory information at specified precision (USDA Forest Service, 2014). A 50-mile radius circle encompasses about 7,850 square miles (slightly over 5 million acres). This is approximately equal in size to each of the five FIA units within the state of Virginia (Rose, 2009). It is also approximately equal to the area of New Jersey or Massachusetts, or the total area of the three smallest states (Delaware, Connecticut, and Rhode Island) combined. Specification of predefined fuelshed-sized regions enables consistent estimation of biomass production and harvest within a region, but also means that some entities may need to acquire feedstocks from multiple regions.

Again, assessment at small scales like the fuelshed level can directly link landscape emissions and sequestration to the use of biogenic feedstock of a specific stationary source, but also necessitates feedstock tracking and other documentation, especially for the retrospective reference point approach.

### 2.3.2. State

An advantage of using a state-level approach is that they often coincide with other administrative or reporting units. Forest harvests and agricultural yield data can be tracked by state (for tax reporting purposes, for example). State boundaries might be logical when states may implement different policies and regulations pertaining to feedstock production as well as commercial trade. However, certain small states (e.g., Rhode Island) may not be large enough to offer adequate or accurate data on biogenic carbon stocks (i.e., forest growth and removals), thus rendering retrospective and future anticipated modeling unreliable because the associated sampling errors are likely too large or model inputs would not be reliable (Crocker et al., 2011). Furthermore, state lines are political boundaries and do not take into account similar landscape types from one state to the next. State lines can divide landscapes that should be considered as a whole. As discussed earlier in this appendix, another potential difficulty with defining spatial scale with a political boundary is assessment for biogenic feedstock transfers across such boundaries because states may have different laws and regulations.

### 2.3.3. Regional

Establishing a regional spatial scale could aggregate multiple states into one primary region of assessment. Here, the regional scale of assessment is large enough that accurate data are available (i.e., adequate statistical power), but still small enough to capture important differences in land

[4] For more information on FIA, consult www.fia.fs.fed.us.

base and therefore feedstock characteristics, such as growth and removal rates, decomposition rates, and species mix (Westfall et al., 2013). In other words, regions achieve a balance between preferred statistical precision of larger scale assessments and ability to capture important land base (biogeochemical and ownership types) and market drivers of smaller spatial scales. Regional assessment allows for important distinctions between drivers of changes in land-based biogenic carbon sequestration and resource supply and demand that, using a reference point baseline, could potentially be masked at the national level. However, regional assessment potentially ignores state- or site-level impacts as well as indirect impacts in other regions (which would be inherently captured by a national approach). Also, determining regional boundaries might be related to market characteristics, with multistate regions forming coherent markets for biogenic feedstocks.[5]

Galik and Abt (2012) provide a thorough evaluation of the impact of spatial scale on the GHG balance of biomass energy production from forest sources. They considered assessment scales from individual sites to fuelsheds to the state level (for the state of Virginia) and projected carbon dynamics for a 25-year time frame relative to a baseline scenario. Their conclusion was that "those assessment scales that do not include possible market effects attributable to increased biomass demand, including changes in forest area, forest management intensity, and traditional industry production, generally produce less favorable GHG balances than those that do." They further concluded that the larger spatial assessment scales (in this context, states and regions) "most closely approximate the actual GHG emission implications" for the scenarios and locations they modeled. However, it is important to note that in some cases the regional scale, like the national scale, can also mask important fluxes in landscape emissions.[6]

Regions could be defined on the basis of homogeneity of biophysical characteristics such as, in the case of forest-derived feedstocks, species types, growth rates, and climate. Regional boundaries must be drawn carefully to ensure the region is large enough to offer adequate data accuracy and availability, yet small enough to better reflect landscape biogenic carbon dynamics. One difficulty with choosing this spatial scale is that each region can encompass multiple states with different laws and regulations. For example, states with strong renewable energy incentives (including renewable portfolio standards or state incentives) and high relative biomass use could drive

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[5] An example of a fixed regional framework is the EPA Emissions & Generation Resource Integrated Database (eGRID) region structure. EGRID is used for calculating GHG emissions related to electricity generation. Subregions nest within regions defined by North American Electric Reliability Corporation (NERC). Regions vary widely in size from small portions of an individual state to areas encompassing portions of seven large states. For more information on NERC and eGRID regions, consult http://www.epa.gov/egrid.

[6] Depending on the spatial scale considered, changes in forest carbon stock can be dramatically different, as illustrated by the impact of hurricane Hugo on South Carolina's (SC) forest resources. In 1989, Hugo hit SC and caused extensive damage to the state's forests. The hurricane reduced the inventory of softwood (e.g., pine) growing stock by 21% or 1 billion cubic feet (Sheffield and Thompson, 1992), which is equivalent to more than 2 years of the previous average forest harvest across the entire state (Tansey, 1986). After the hurricane, the removals of softwood timber in the state exceeded the net growth by 43% (Conner, 1993), whereas before the hurricane net growth exceeded removals by 2% (Tansey, 1986). However, in the subsequent assessment of forest resources (Haynes et al., 1995), southern softwood net growth exceeded harvests. Thus, the deficit situation in SC resulting from the hurricane impact was not observed in the larger region of the south and applying regional southern assumptions regarding balance between growth and removals to SC could have led to additional pressure on the resource.

landscape biogenic feedstock removals and associated emissions fluxes for an aggregated region. Landscape emissions impacts in neighboring states in the same region could be modest, but a regional assessment could reveal large landscape emissions changes due to policy actions in one state.

### 2.3.4. National

The next largest spatial scale possible for estimation and reporting would be national. Although a global assessment scale is certainly possible, the highest level of spatial aggregation evaluated in this report is national. The key advantages of a national-level assessment are that it captures market interactions, including domestic leakage effects, and offers high-level insights concerning general emissions fluxes from U.S. carbon stocks in forests and agricultural landscapes. The market interactions component is especially critical, especially for the anticipated baseline approach. A regional assessment of a biogenic feedstock demand shock may not capture emissions changes outside of the assessment region as markets adjust to the shock and production expands or contracts elsewhere. A national assessment using an anticipated baseline modeling approach would capture these interactions and indirect emissions impacts. Furthermore, evaluating landscape emissions in response to a national policy could justify a national assessment scale (Latta et al., 2013).

At the national scale, observing or projecting emissions fluxes from managed terrestrial systems (i.e., from U.S. forests and agricultural lands) can be accomplished using published datasets such as the U.S. GHG Inventory and/or models designed to project emissions from land management activities. At this assessment scale, however, quantifying the relationship between the actions of an entity using biogenic feedstocks (or a group of such entities) (i.e., biogenic feedstock demand) and the carbon dynamics of the feedstock production site (which is defined nationally) (i.e., biogenic feedstock supply) could be difficult, especially for certain feedstocks. Assessing such causality at this scale is difficult as it is hard to differentiate between this driver (biogenic feedstock demand) and other influences on the national landscape (e.g., urbanization, natural disturbances). Also, reporting changes in biogenic $CO_2$ fluxes at the national scale could mask important regional differences in landscape and feedstock characteristics such as growth rates, species composition, and other environmental conditions, especially when applying a reference point baseline approach.

For example, if one is interested in carbon stock changes associated with a particular forest harvest, reporting and considering the effects of the harvest at a national scale would likely reveal little or no measurable impact on overall carbon stocks at the national level. However, by normalizing the impacts (e.g., $CO_2e$ per ton of feedstock harvested), the national level results can be informative and account for certain impacts that could be lost in a regional-scale analysis (e.g., inter-regional, domestic leakage effects). When using a retrospective approach one might need to establish a causal statistical relationship between the harvest under consideration and resulting emissions changes elsewhere. Ultimately, carbon stocks may be declining in some areas but increasing at a higher rate in other areas, regardless of whether a reference point or future anticipated baseline approach was applied and regardless of biogenic feedstock demand for energy purposes. Reporting changes in carbon stock at the national scale would mask important regional differences in terms of harvest and growth rates, as well as species composition, and climate. The result of a national scale assessment is that the evaluation of one harvest activity could have a very minor or statistically indistinguishable impact on overall national carbon stocks.

Were this same forest harvest reported on a fuelshed scale (the area required to provide continuous forest-derived biogenic feedstock to a specific end user) instead of a national scale, it likely would have a measurable impact because of the smaller area under consideration. However, this impact would potentially ignore other market adjustments and landscape impacts at the state or regional scale. The actual harvest itself is the same in both scenarios, but the measured impact would be different because of the choice of assessment spatial scale.

Similarly, waste-derived materials also may have some regional variability, including the composition of waste (which can vary from community to community within a region) and regional climate factors that affect methane ($CH_4$) oxidation via cover soils at managed landfills (Bogner et al., 2007; EPA, 2009; Spokas and Bogner, 2011). However, there is a lack of literature describing the degree to which composition of municipal solid waste (MSW) can vary from region to region, and thus this analysis uses a national average composition based on EPA data through 2012 in the illustrative calculations in Appendix N (EPA, 2014). Although composition of MSW may vary from region to region, this mainly contributes to potential generation *amounts* of $CO_2$ and $CH_4$ in a given landfill, whereas the goal of the framework methodology for waste-derived feedstocks is ultimately concerned with *how* the $CO_2$ and $CH_4$ from MSW are treated and used in one activity versus another. From this perspective, $CO_2$ and $CH_4$ from MSW can be treated similarly across the United States.

## 2.4. Spatial Scale in the Framework

A spatial scale should be small enough to recognize changes (e.g., carbon stock changes, emissions fluxes), drivers, and trends and large enough to offer accurate data and be capable of dealing with large stochastic events such as storms. It should have the ability to recognize cross-boundary flows. Too large an area and important local or regional trends could be masked; too small an area and limited data will preclude accurate estimation or would overestimate or underestimate the net landscape emissions impacts by ignoring changes in land management at a regional scale. The spatial scale should be determined by a trade-off between the statistical precision and data availability for larger regions, against the local specificity and accurate depiction of biophysical attributes of smaller regions.

Ultimately, the choice will depend on the specific context and program, and it may be possible to use different or nested spatial scales within the same set of analyses.[7] This framework explores the regional scale further in the sections below to derive proof-of-concept values.

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[7] This framework can be customized so individual entities using biogenic feedstocks can derive and input entity-specific values into the framework's equation to calculate an individualized *BAF* (see the main report Part 3 for more on customized feedstock approaches). However, in some policy or program applications or for some entities, this customized approach will not be appropriate or feasible so the framework can be applied at different scales.

### 2.4.1. Assessment at the Regional Scale

The *location* of regional boundaries should reflect land base characteristics and the spatial distribution of biogenic feedstock characteristics such as species, rates of productivity, similarity of management practices, ownership patterns, and market attributes. Regional boundaries can coincide with other administrative or reporting units, because this may increase the likelihood that other relevant data or model outcomes would be summarized for the regions. Because it is further likely that forest harvests would be tracked by state (for tax reporting purposes, for example), the use of state boundaries as regions, where possible, may be advantageous.

The *size* of the regions should be determined by a trade-off between the statistical and modeling precision offered by larger regions with improved biophysical information and local specificity of smaller regions. However, the practical implementation of an assessment framework must also be a consideration: it is recognized that at larger spatial scales, implementation becomes simpler.

The actual regional delineations applied to the reference point and future anticipated baseline supporting appendices apply slightly different regional scales, as discussed below.

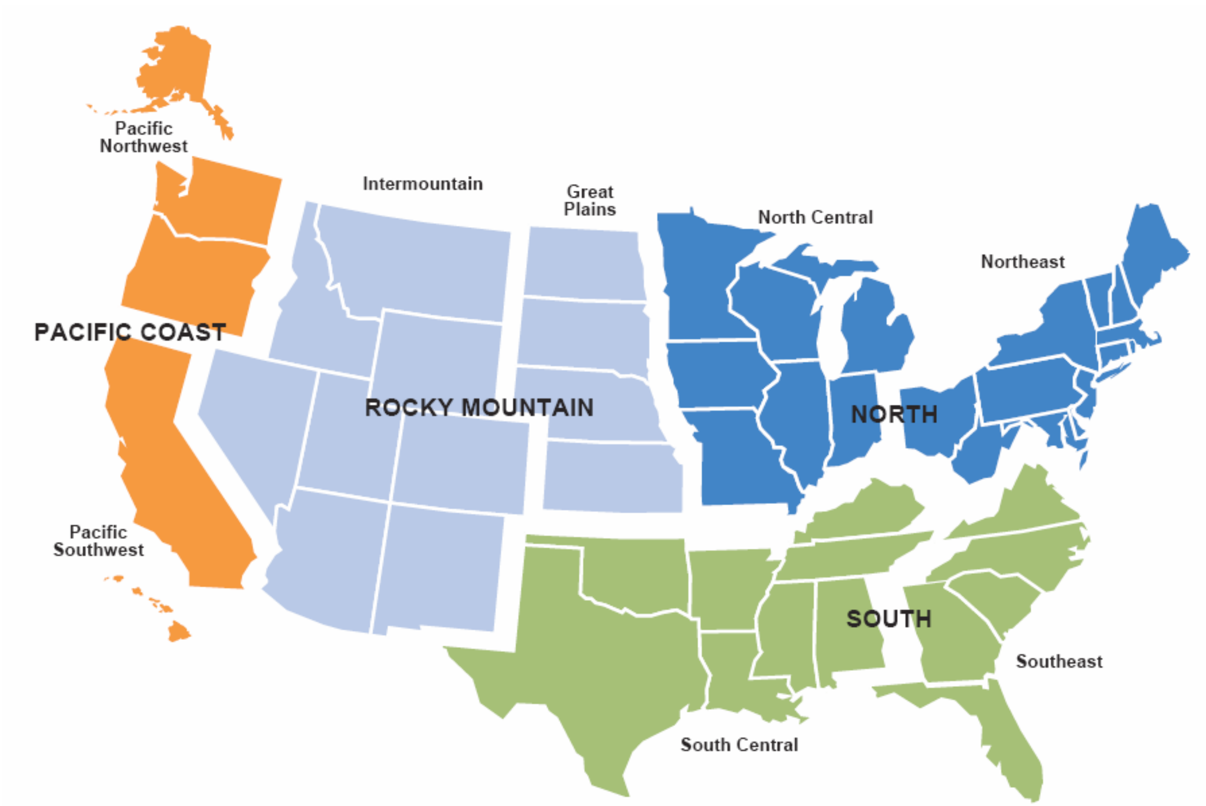


**Figure C-1. RPA Regions (USDA Forest Service, 2012).**

#### 2.4.1.1. RPA Regions

For the retrospective reference point baseline approach illustrative examples for forest-derived feedstocks provided in subsequent appendices, the regions follow the region boundaries developed by the USDA Forest Service for the Resources Planning Act (RPA), resulting in eight regions as shown in Figure C-1.

The RPA regions are based on publicly available data on forest resource stock dynamics (inventory, growth, harvest) generated by the FIA program of USDA Forest Service. These regions are designed to reflect the spatial distribution of forest characteristics (forest types, rates of productivity, management practices) and follow state boundaries to increase the likelihood that other relevant data are reported at the same administrative level. The RPA regions form the basis for a wide array of reports on forest resources conditions, markets, and trends (see USDA Forest Service, 2012).

For forest-based operations, one important source of publicly available data on forest stock dynamics (inventory, growth, harvest) comes from the FIA. FIA collects and provides information on hundreds of thousands of sample plots nationwide on all types of forest ownerships. FIA data can be used to assess availability of forest-derived feedstocks and estimate production of harvest residues (Conner and Johnson, 2011; Johnson, 2001). These data can be used to estimate several terms in the *BAF* equation using the reference point baseline approach.

For forest-derived feedstocks, the smallest spatial scale at which FIA data are reliable may be somewhat larger than a fuelshed. It is possible to use FIA data to narrow down the forestland base within prospective regions to a working forest for each region and then compute variables such as gross growth, removals, and excess growth, along with their sampling errors. Sampling errors for basic estimates of overall biomass may be within a few percent at this scale. However, sampling errors on other variables of interest—such as growth and harvest—will be much higher. For example, the state of New Jersey is about 7,500 square miles in size, approximately the size of a 50-mile radius circle that might approximate a fuelshed. In a recent report (Crocker et al., 2011), the sampling error for the volume of New Jersey's growing stock was 4.6%, but sampling errors for growth and removals were 9.62% and 29.5%, respectively.

Therefore, although the precision of the basic volume estimate may be acceptable at fuelshed scales, growth and removal metrics related to the balance of carbon emissions and sequestration will be less reliable, and for that reason larger spatial scales are preferred. At the RPA regional level, the spatial scale provides estimates within acceptable uncertainty ranges.

#### 2.4.1.2. FASOM-GHG Regions

The regional delineation used for the illustrative *BAF* equation applications for agriculture-derived feedstocks under the retrospective reference point baseline approach and for forest-derived and agriculture-derived feedstocks under the future anticipated baseline approach (Appendices H through M) is the delineation as used within the U.S. FASOM-GHG. These 11 regions are shown in Figure C-2.

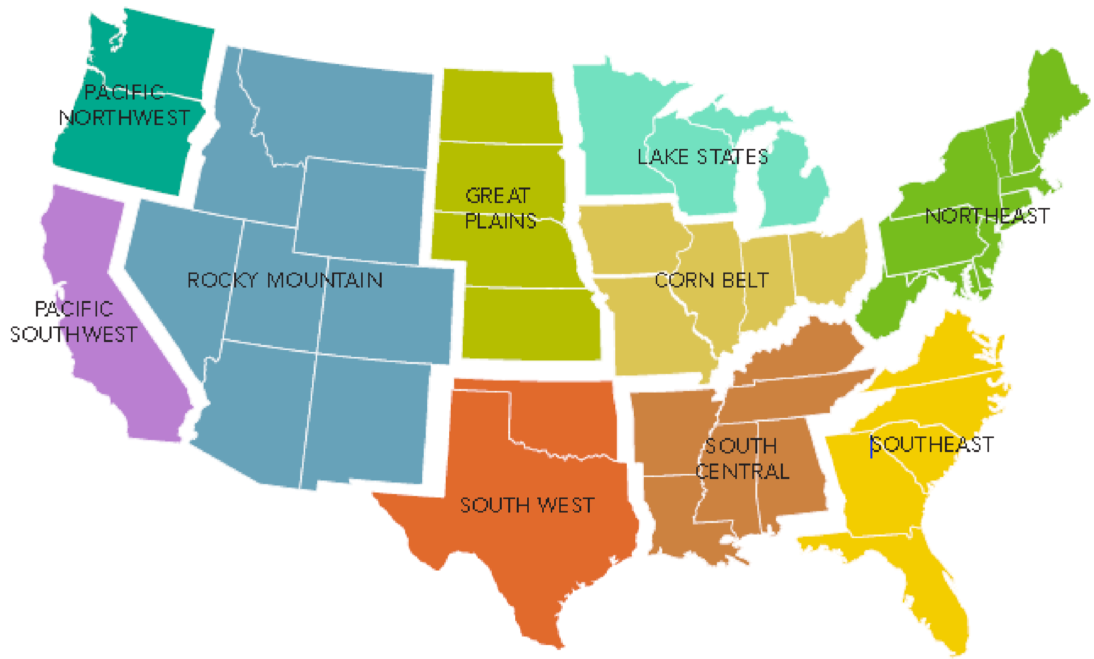


**Figure C-2. FASOM-GHG Regions.**

The FASOM-GHG regions are based on the underlying datasets used in the FASOM-GHG model.[8] Many of the datasets are resolved at the state and/or county level and aggregated up to the various FASOM-GHG regions for reporting. Forest-sector data are based on a number of relevant datasets, including the FIA. As noted, the FASOM-GHG regions are similar to the RPA regions in general size and geographical location. Specifically, the FASOM-GHG regions reflect areas that exhibit similar land characteristics, crop types, existing forest resources, forest/crop yields, forest/agricultural management alternatives, soil types, rainfall, and climate patterns (see Beach and McCarl, 2010).

# 3. Conclusion

The RPA and FASOM-GHG regions are examples of spatial scales that address some of the tradeoffs previously discussed in this appendix. The regions are small enough to recognize trends and changes in growth and removals, yet large enough to offer widely available data and adequate statistical power. Furthermore, the RPA and FASOM-GHG regions are well established in the literature and not as complex as alternative regional delineations (e.g., the eco-regions previously developed by EPA for ecological applications [see Bryce et al., 1999]). The regions also largely follow state boundaries, which allows for easier reporting and greater recognition of cross-boundary flows.

That said, although the RPA and FASOM-GHG regions are used in this report to road test the framework, they are selected for illustrative purposes only. Ultimately, any final choice of regional

[8] Additional information on the FASOM-GHG model and its application for the technical appendices of this report can be found in Appendix L.

delineation is a decision for policy makers and should reflect the requirements for a particular program.

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# Appendix D: Feedstock Categorization and Definitions

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# 1. Introduction

The purpose of this appendix is to describe the categorization of biogenic feedstocks used in the assessment framework. A biogenic feedstock can be defined as any organic material originating from modern or contemporarily grown plants, animals, or microorganisms, excluding material embedded in geological formations or fossilized, that is used for combustion or product processes or otherwise decomposes at a stationary source. A multitude of feedstocks meet this definition, though some feedstocks are more commonly used for bioenergy than others. Feedstocks differ in physical properties; origin (including local climate and biogeochemical attributes); species; growth rates; management (from planting to harvesting); and whether they are deliberately raised as an energy feedstock or if they can be used for other purposes (e.g., human or animal consumption), are reclaimed wastes from other processes, or are salvaged following extreme events such as hurricanes or insect outbreaks. Duration of typical growth or decay periods also can differentiate feedstocks. Annual crops, for example, might be accounted for differently than perennial crops, and both might be accounted for differently than waste-derived feedstocks. Furthermore, a feedstock in continuous supply may need to be accounted for differently than a feedstock available only occasionally (e.g., short growing seasons, feedstocks that result from fire or insect infestation).

This appendix first lays out the broad feedstock categorization used in the framework. It then discusses the various feedstock attributes of commonly used biogenic feedstocks and why certain feedstocks are grouped together. This appendix also generally discusses other feedstock categories, including secondary use feedstocks, imports, and emerging markets. The list of feedstocks included in this categorization is not exhaustive and may need modification per specific policy applications and/or as new feedstocks come into the biogenic feedstock market. The broad feedstock categorizations in this appendix are not intended to represent specific regulatory definitions that currently exist or that may need to be developed as part of policy applications of the framework.

## 2. Summary of Biogenic Feedstock Categories Used

To account for differing feedstock characteristics, the framework separates biogenic feedstocks that might be used in a stationary source into three basic categories:

1. Forest-derived feedstocks: biomass derived from natural forests, tree plantations, and wood products production processes;
2. Agriculture-derived feedstocks: biomass derived from agricultural operations; and
3. Waste-derived feedstocks: biomass derived from any source of animal, industrial, or municipal waste.

The framework uses these categories because they are large enough to capture the important differences among feedstocks in terms of their biophysical attributes but small enough to be more manageable and understandable for application in a stationary sources context. Table D-1 includes examples of biogenic feedstocks under these common categories that have been used commercially, or could be used commercially in the near future, for bioenergy purposes.

**Table D-1. Biogenic Feedstocks.**

| Forest-derived Feedstocks | Agriculture-derived Feedstocks | Waste-derived Feedstocks |
|---|---|---|
| ***Roundwood:***<br>Pulpwood, saw logs<br><br>***Logging Residue:***<br>Branches and limbs, debris | ***Conventional Agricultural Crops:***<br>Camelina, corn, canola, sorghum, soybeans, sugarcane, wheat<br><br>***Agricultural Crop Residues:***<br>Barley straw, corn stover, oat straw, rice straw, wheat straw<br><br>***Dedicated Energy Crops:***<br>Miscanthus, napier grass, switchgrass, short rotation woody crops (e.g., hybrid poplar, poplar, willow, eucalyptus) | ***Municipal Solid Waste:***<br>Urban wood waste, yard trimmings, food waste from industrial processes, kitchen scraps<br><br>***Animal Wastes:***<br>Livestock manure, litter, manure wastewater<br><br>***Wastewater*** |
| ***Industrial Products and Processing By-products:***<br>• No current alternative market uses, such as pulping liquor<br>• Has current alternative market uses, such as mill residues (bark, peeler shaving, sawdust); ethanol; pellets | ***Industrial Products and Processing By-products:***<br>• No current alternative market uses, such as: shells, husks, and cobs<br>• Has current alternative market uses, such as animal fats, oils, and greases; distillers grains; ethanol; biodiesel | |

# 3. Biogenic Feedstock Characteristics

The feedstock categories are based on the key characteristics of feedstocks themselves as well as the feedstock source conditions that may lead to different net biogenic carbon-based emission profiles and thus merit different treatment under the framework. These characteristics generally include similarities and differences between feedstock growth and decay cycles, typical management and land use patterns associated with feedstock production, potential alternate market and carbon fate pathways, and other factors. More specifically, these characteristics include the following:

- *Time scale over which feedstock carbon sequestration and emissions occur.* For some feedstocks, carbon sequestration into the feedstock can occur over a short time (i.e., a year or less). For other feedstocks, sequestration occurs over a much longer time (i.e., decades to hundreds of years). Although emissions to the atmosphere occur instantaneously during combustion, some emissions-associated feedstock losses (e.g., decay) or alternative pathways may take place over days, weeks, months, or even years during storage and handling.
- *Alternate fate pathways (assumptions about "What would have happened otherwise?").* Baseline assumptions can involve consideration of the end-of-life emissions profile of each feedstock were it not used at the stationary source for energy. The baseline assumptions vary according to the feedstock type and could vary if there are other possible market uses. For example, some feedstocks may be left undisturbed to decompose if not used for energy, thereby emitting both $CO_2$ and $CH_4$, which is avoided when the feedstocks are combusted. If not used for energy, some feedstocks would otherwise have to be disposed of (landfilled or other means). Conversely, some feedstocks may have been used in other markets if they were not being used at a stationary source for bioenergy production.
- *Land use/land-use management changes.* The cultivation and use of certain biogenic feedstocks can create market competition that stimulates a shift in land use or land use management changes. Changes in land use and related management activities can generate emissions that contribute to the net atmospheric contribution from using the feedstock at a stationary source.
- *Leakage:* The use of some feedstocks for bioenergy may have GHG emissions effects outside of the biogenic feedstock production assessment boundary caused by the biogenic feedstock production activities (e.g., replacement of diverted crop, livestock, or forest products due to a change in land use from conventional products to biogenic feedstocks). The directionality and magnitude of these leakage effects may vary significantly according to feedstock type, location, and other factors. Further discussion can be found in Appendix E.
- *Storage and handling losses.* Various steps involved in converting a biogenic feedstock into a bioenergy product may involve losses of the biogenic carbon during transportation, storage, and handling. These feedstock losses vary according to the feedstock type.

The following subsections discuss how different feedstock categories could be considered within the framework according to their different characteristics.

## 3.1. Forest-derived Feedstocks

Forest-derived feedstocks currently constitute one of the largest sources of bioenergy in the United States (EIA, 2011). The majority of that energy production is currently derived from wood products processing. However, increased demand for bioenergy can result in higher prices being paid for bioenergy feedstock, and in doing so, increase competition with existing forest products markets, especially pulpwood (Becker et al., 2009; Galik et al., 2009; Lundmark, 2006). Increasing demand for pulpwood can have cascading effects on sawtimber markets, as few stems are left to grow into sawtimber size classes (Abt & Abt, 2013). The interaction between bioenergy harvests and broader timber market effects is somewhat dependent on the rate at which and the cost at which residues can be recovered. Abt & Abt (2013), for example, show that the assumed rate of logging residue recovery has a substantial influence on timber market response, and by extension, possible future changes to forest land management.

In this section, characteristics of roundwood, logging residue, and forest-derived industrial products and processing by-product feedstocks are presented.

### 3.1.1. Roundwood

Roundwood biomass includes trees of commercial size, species, and quality from a forest or plantation in an area with commercial markets. Roundwood is most often sent to sawmills or pulp and paper mills, though it is occasionally used for energy purposes (as clean chips, for example) at dedicated or cofiring electricity generating unit (EGU) facilities.

- *Time scale over which feedstock carbon sequestration and emissions occur.* Roundwood feedstocks typically have a longer harvest cycle than agriculture-derived feedstocks like traditional crops or dedicated energy crops. Example harvest cycles for roundwood pulp and sawtimber production are about 11 to 15 years for pulpwood and 25 years for sawtimber grown on plantations in the southern United States and 45+ years for sawtimber in the Pacific Northwest. Note that the overall average age of the U.S. forest inventory is older than these values because of the inclusion of less actively managed forest area.
- *Alternate fate pathways:* Roundwood, if used for industrial purposes other than energy, could lead to long-term carbon sequestration. For example, if the wood were used for furniture, buildings, or pulp and paper, its carbon would be sequestered for longer than if it is burned immediately for energy purposes (though pulp and paper products would generally have shorter sequestration time frames than more durable products) (Skog, 2008). Additionally, roundwood use at stationary sources for the sole purposes of bioenergy production could detract from roundwood use in other markets.
- *Land use/land use management changes:* Roundwood biomass can have several markets competing for the same raw material. For instance, pulp and biomass-to-energy markets can compete for the same tree sections. As such, changes in demand for roundwood biomass, whether for bioenergy or other uses, can lead to changes in production, potentially

causing direct land use or land use management changes. Such changes, including, for example, shortening rotation ages/increasing harvest frequency, can in turn cause the landscape to have different GHG emissions profiles and equilibrium states.

- *Leakage:* Increasing use of roundwood for energy production would likely also have ramifications throughout related commodity markets, causing leakage effects such as indirect land use change. For example, leakage could take the form of additional land outside of accounting boundaries converting from a previous use to accommodate displaced market demand (e.g., shifting cropland or pastureland to forestland, which could result in higher carbon sequestration on those newly forested lands, but potential carbon emissions due to conversion of other lands elsewhere to cropland).
- *Storage and handling losses:* Roundwood biomass is often harvested, preprocessed, transported, and used within a matter of days or weeks, which limits storage losses. However, depending on location, there may be longer storage needs at certain times during the year, and some degradation of woody biomass and associated dry matter loss could occur during storage and handling. The degree to which dry matter loss occurs depends largely on moisture content, where woody materials with high moisture levels are more likely to be colonized by fungi and mold, which can cause dry matter losses. In addition, the longer biomass is stored, the greater the dry matter loss, other things being equal. Accounting for forest-derived feedstocks like roundwood should cover losses in storage and material handling to provide a complete link between feedstock available at the source location and that used in the stationary source.

### 3.1.2. Logging Residue

Logging residues include biomass derived from harvest operations including treetops and non-merchantable sections of the stem, branches, and bark left on the ground after logging. If not left to decompose or open burned on site at the logging operation site, logging residues are often sent to sawmills or pulp and paper mills, though they are also used for energy purposes, either at EGUs or to fuel internal processes at sawmills and pulp and paper mills.[1]

- *Time scale over which feedstock carbon sequestration and emissions occur.* If left in the forest, logging residues may be either burned or left to decay over a period that can range from days to years, depending on the size and nature of the woody material and the surrounding environmental conditions (e.g., moisture, soil type, exposure to light) (Turner et al., 1993; Turner et al., 1995). Materials such as leaves of deciduous trees will decompose within a couple of years, while conifer needles will often take several years. In general, the wetter the biomass on the forest floor, the faster these residues will break down. If not burned, non-merchantable large woody material would decay slowly in the forest, and its carbon

[1] Traditional harvests (removing tree boles and leaving tops and limbs on site) in some instances may transition to more intensive practices such as whole-tree harvesting/chipping. In this presentation of the framework, whole-tree use is not attributed exclusively to the roundwood feedstock, because the practice also includes the harvest of what traditionally would have been left as residue. Thus, a whole-tree harvest can be viewed as removing two feedstocks: roundwood and logging residue. Thus, under the framework, whole trees could be divided into both roundwood and logging residue.

content in the forest can be estimated from sampling surveys. It can take several decades for pine logs with high resin content to fully decompose.

- *Alternate fate pathways:* If left in the forest, logging residues may be either burned or left on site to decay over a period that can range from days to years (as discussed above), which can have different net biogenic contributions to the atmosphere. Under current biomass market prices in most regions, logging residues are often not collected, and procurement of residue does not trigger the harvest operation (DOE, 2011).
- *Land use/land use management changes:* The type of harvest operation (e.g., whole tree versus non-whole tree harvest), stand and timber structure, and soil conditions play significant roles in the abundance and merchantability of logging residues. For instance, hardwoods in general yield higher percentages in non-timber biomass than softwoods. If soils are wet, logging residue material may be used to stabilize skid trails resulting in no surplus for feedstock supply. Extracting biomass for energy production often requires the simultaneous harvest of more valuable wood (timber, pulpwood) to justify the cost of collecting the residue material. Under current market conditions, increased demand for logging residues for bioenergy production is unlikely to expand the harvest area (i.e., land use change) though it could change the intensity of residue collection operations (i.e., a land use management change). If more logging residues are collected as biogenic feedstocks for energy production, this management change could impact the soil carbon contributions to the harvest landscape and, conversely, remove the volume of woody matter decaying on the forest floor or being burned on site.
- *Leakage:* Under current market conditions, there are no commercial alternative markets for logging residues and thus few pathways for increased logging residue removal inspire leakage effects such as indirect land use change. However, if the demand for logging residues increases substantially (e.g., as markets for bioenergy feedstocks develop), this could alter current practices to the extent that leakage could potentially occur.
- *Storage and handling losses*. As noted above, to the extent that forests are harvested and used on a fairly continuous year-round basis with only days, possibly weeks, between harvest and use, there will tend to be relatively little storage and handling loss. However, the longer storage is required, the greater dry matter loss will occur, other things being equal. Within forest-derived feedstock types, logging residues are more likely to experience feedstock losses during transport, storage, and handling than roundwood because of the smaller size of the feedstock pieces. Nonetheless, processing losses of forest-derived biomass are generally expected to be minimal.

### 3.1.3. Forest-derived Industrial Products and Processing By-products

Usual practices within the forest industry generate a wide variety of forest industrial products and processing by-products. These by-products include liquids such as black liquor from the pulping process and mill residues such as bark, shavings, sawdust, sanderdust, hog fuel, and unusable bole components (due to knots, holes, etc.).[2] Consideration of forest-derived industrial processing

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[2] Forest products are characterized by a joint production function, as any products are produced from a single tree. Forest product industrial entities will try to optimize production to maximize the amount of high-value products

products and by-products should include assessment of whether these materials have current alternative market uses to bioenergy or not. Most residues from wood processing facilities are currently used for onsite energy production or sold for other forest products (e.g., particleboard). Deviating by-products that do have current market uses to additional energy production instead of their traditional use could have potential impacts on those traditional markets. For example, markets for sawdust, shavings, and chips from sawmills are well established. Sawdust and shavings may be used in composite wood products such as particleboard, medium density fiberboard (MDF), or pellets, shavings may be sold to farmers or pet owners for animal bedding, and bark may be sold for use as mulch or fuel. Very small amounts of mill residue go unused (USDA, 2007; U.S. Department of Energy, 2011). In addition, chips are often sold to pulp mills (USDA, 2007). Bark, slabs, edges, and other material may be burned on site at forest product mills for heat and energy production. Thus, when mill residues that were used in other markets are diverted into energy production, it may be an indication that leakage effects are possible (i.e., if sawdust goes to a biomass energy entity rather than a pulp mill, the pulp mill will need to make up the shortfall, possibly by increasing pulpwood harvests).

An example of a forest industrial processing by-product with no current alternative market use is spent pulping liquor. Spent pulping liquor (e.g., black liquor from the kraft pulping process) contains nearly half the original energy content of the wood and is not currently sold on the market, because it is typically combusted within the pulp mill chemical recovery process for purposes of reclaiming pulping chemicals and producing energy. If not combusted for chemical recovery and energy, black liquor-producing entities would need to dispose of the material (e.g., treatment in wastewater treatment systems, decay in lagoons, combustion without energy). When evaluating black liquor combustion on site for energy versus possible alternates fates (disposal and potential $CH_4$ and $CO_2$ emissions from decay), black liquor combusted for on-site energy is expected to have less net atmospheric biogenic $CO_2$ contributions. Also, because black liquor production is contingent on paper production and related paper market demand and prices, it is therefore unlikely that changes in demand for or prices of black liquor would lead to changes in paper production and related land use, harvest, or forest management decisions (e.g., no effect on landscape attributes). This may also be the case for other industrial processing by-products with no current alternative market uses. More information and analysis on black liquor can be found in Addendum A to this appendix.

- *Time scale over which feedstock carbon sequestration and emissions occur:* For this feedstock category, the analysis of the time scale for feedstock-related carbon fluxes will depend on the feedstock and landscape attributes and/or alternate paths associated with the feedstock (e.g., the feedstock production site and the alternate fate pathways [discussed below]).

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(e.g., saw lumber) and minimize the amounts of low-value products (e.g., pulp, black liquor). Although there is some responsiveness to relative price movements (e.g., higher demand and prices for wood pellets may lead to an increased proportion of scrap going to this use and a decreased proportion going to particleboard), the elasticity of transformation between outputs may be very inelastic, and even with a negative price some low-value products would still necessarily be produced as a by-product of the production of high-value products (e.g., sawdust, black liquor).

- *Alternate fate pathways:* The alternate fate of forest-derived industrial products and processing by-products can vary widely per feedstock (those with and without alternative market uses) and per stationary source process. In the case of feedstocks with alternative market uses, these feedstocks typically would not be used for energy if there is a higher-value use (e.g., as raw material for pulping or composite wood products), and these feedstocks would pass through the stationary source through means other than the stack. For those feedstocks with no other current market uses, the alternate fate pathways could include use for energy (e.g., as boiler fuel) and disposal (e.g., through non-energy-related burning, landfilling, on-site storage), which might include decay causing $CH_4$ and $CO_2$ emissions.
- *Land use/land use management changes:* Under current market conditions, there is no evidence of land use or land use management changes related to producing forest-derived industrial products and by-products.
- *Leakage:* If demand for forest-derived industrial products and processing by-products increases for energy use, leakage could occur if these feedstocks currently have alternative market uses.
- *Storage and handling losses:* With some feedstocks in this category that require storage, the longer a feedstock is stored, the greater dry matter loss will occur, other things being equal. Within forest-derived feedstock types, industrial products and process by-products are more likely to experience feedstock losses during storage and handling than roundwood because of the smaller size of the feedstock pieces (Thornqvist and Jirjis, 1990; Jirjis, 1995; Afzal, 2010).

Furthermore, some products from the forestry sector are purposefully produced for energy production. These products include pellets or other fuels produced from woody biomass (these are covered in the secondary use feedstocks section below).

## 3.2. Agriculture-derived Feedstocks

Although the majority of biogenic feedstocks that have been used for energy generation in the United States to date are derived from forest materials, there is potential for large-scale use of agricultural feedstocks as well. Traditional agricultural crops that have historically been grown for food, feed, and fiber could be used as bioenergy feedstocks, as could crop residues, dedicated energy crops, or industrial products and processing by-products. This section describes characteristics of these agricultural feedstocks, separating them into three categories: (1) crops grown primarily for bioenergy use, whether conventional or dedicated energy crops; (2) crop residues; and (3) agricultural-derived industrial products and processing by-products.

### 3.2.1. Conventional Agricultural Crops

The conventional agricultural crops category includes feedstocks from crops traditionally grown for food, feed, textile, or other uses, such as corn and soybeans. These crops can be converted at stationary sources into conventional starch-based fuels, electricity, biodiesel, and cellulosic fuels. Use of these feedstocks in bioenergy production could result in changes in their production, price, and trade and potentially result in direct and/or indirect land use change. Crops for which only the processing by-products from a multiproduct processing activity (e.g., soybean oil, rice hulls) are used for bioenergy are covered under the agriculture industrial processing by-products subcategory below.

- *Time scale over which feedstock carbon sequestration and emissions occur:* Growth and harvest of, and related sequestration and emissions from, conventional crops generally occur at time scales of a few months to a year. Even though the net atmospheric biogenic contribution from the growth and harvest of the feedstock itself is in balance, other factors such as land use management and land use changes and related soil carbon effects can affect the overall assessment outcome.
- *Alternate fate pathways:* If conventional crops are not used for bioenergy, they would be used for other purposes, such as food, animal feed, or fiber or the production of liquid biofuels.
- *Land use/land use management changes:* There can be direct land use change effects if the demand for agricultural crops for bioenergy causes changes in land use and land use management. Changes in demand for agricultural biomass can lead to changes in cropping patterns, and production practices (e.g., intensification) can lead to changes in GHG emissions (e.g., impacts on soil carbon levels).
- *Leakage:* Leakage effects related to conventional crops for energy purposes can be substantial. The new or diverted production of feedstocks can affect commodity markets and thus lead to changes in production that alter land uses and land use management outside of the bioenergy feedstock supply chain (Murray et al., 2004; Searchinger et al., 2009; EPA, 2010a). For example, if forested land is converted to crop production for energy uses, the carbon storage occurring on the landscape changes in both standing biomass and soil carbon pools. Also, other forested lands outside of the bioenergy supply chain could become managed or be managed differently to meet the market product demand displaced when the original forestland was converted to crops for energy use.
- *Storage and handling losses*. Agricultural feedstocks generally need to be processed before they can be used for energy, which can lead to low levels of decomposition or physical feedstock losses. Additionally, because of their seasonal nature, conventional agricultural biomass needs to be stored to provide a year-round supply of energy. Thus, agricultural biomass may experience more feedstock losses than forest biomass, on average, simply because it typically needs to be stored longer. Dry matter losses during storage inside buildings are expected to be less than 5% (Collins et al., 1997; Huhnke, 2006; Shinners et al., 2007). Outside storage could lead to substantially greater losses because of the exposure to weather and increased losses to pests.

### 3.2.2. Agricultural Crop Residues

Agricultural crop residues, such as corn stover, wheat straw, and rice straw, can be collected for conversion or combustion at stationary sources to generate electricity and cellulosic fuels. These residues are traditionally tilled into the soil, providing nutrients for the next planted crop. However, crop residue management changes, such as the removal of residues that would otherwise remain on the field or be open burned, result in impacts on soil conditions (e.g., increased erosion) as well as soil carbon levels (e.g., lower carbon inputs to the soil). In addition, in some instances (e.g., the removal of corn stover) there may also be a loss of nitrogen in the soil. In such cases, to address this loss of soil nitrogen following the removal of corn stover, additional fertilizer may be added to ensure continuing yields of the main corn crop. Thus, additional $N_2O$ emissions may be incurred as a result of residue removal.

- *Time scale over which feedstock carbon sequestration and emissions occur:* Crop residues are generated on an annual basis when crops are harvested, with subsequent decomposition taking place over a period of months or years depending on production practices and environmental conditions.
- *Alternate fate pathways:* Agricultural residues, like forest residues, would decay, emitting $CO_2$ and $CH_4$, and make small contributions to soil carbon if they are not removed for bioenergy or other uses. Removing residues may increase the return of carbon to the atmosphere from the residues in the short term and reduce the amount of carbon stored in the soils over a longer term.
- *Land use/land use management changes:* In the case of changes only to land use management, there may potentially be effects from removing agricultural residues from the landscape and using agricultural processing by-products, because land use management changes can affect soil GHG fluxes.
- *Leakage:* In addition, to the extent that a market develops for crop residues, this additional coproduct of crop production may increase returns to production of agricultural crops with marketable residues and induce land movements from other uses to crop production with residues.
- *Storage and handling losses*. Accounting for agricultural feedstocks such as residues and by-products should cover any losses in storage and material handling. Although these losses may be small compared with feedstock use (less than 10%), these losses in the supply chain are required to link what is being used in the stationary source process to what is grown at the feedstock source location.

### 3.2.3. Agriculture Industrial Products and Processing By-products

Similar to those produced in the forestry sector, agriculture industrial products and processing by-products are considered in two subcategories: those with current alternative market uses and those without. Again, if these industrial product and processing by-product feedstocks are used to produce bioenergy instead of traditional market uses, this alternative use for energy can potentially disrupt the traditional markets and have related land use and/or land use management changes.

Examples of agriculture industrial processing by-products with a current alternative market use include animal fats, oils, and greases that come from livestock production and distillers grains that are produced during the grain ethanol production process. Animal fats, oils, and greases are used in a variety of markets, including the manufacture of beauty products, pet foods, and many other goods. Distillers grains are used as animal feed, often serving as a substitute for corn and soybean meal in feedlots.

Examples of agriculture industrial processing by-products with no current alternative market use (outside of renewable fuel production) include shells, husks, and cobs.

However, some agriculture-derived products are produced for energy purposes but not necessarily for use at stationary sources. These include ethanol and biodiesel produced from a variety of agriculture sources: grains, oilseeds, cellulosic material, sugar-based crops, and more. These products tend to be produced for use in the mobile source sector, but their use in a stationary source remains a possibility.

- *Time scale over which feedstock carbon sequestration and emissions occur:* For this feedstock category, the time scale over which sequestration occurs for the primary crop is typically less than a year because the biogenic feedstock is derived from the production of annual crops. For emissions from the industrial products and by-products, the time period over which emissions would take place will depend on the specific feedstock being considered and the alternate fate pathways.
- *Alternate fate pathways:* The alternate fate of agriculture industrial products and processing by-products varies across feedstocks and by stationary source process. Some feedstocks have active alternative markets, whereas others do not currently have alternative market uses. In the case of feedstocks currently being used in alternative uses (e.g., distillers grains, oils), they would typically not be used in energy production as long as there are higher-valued uses. For feedstocks without current alternative market uses, the alternate fate pathways could include use for energy. Materials such as corn shells, husks, and cobs may be spread back on the field as the combine harvests the grain, which helps maintain soil quality. Thus, using these materials in alternate ways may necessitate the addition of more fertilizer or other soil amendments to maintain soil quality. A given alternate fate pathway for these feedstocks may potentially result in $CO_2$ and $CH_4$ emissions from decay.
- *Land use/land use management changes:* Under current market conditions, using agricultural industrial products and processing by-products that are currently being used in making animal feed, cosmetics, or alternative fuels not used at stationary sources is likely to lead to market impacts. Land use may change as demand for agricultural commodities that generate industrial products and by-products that can be used for energy production increases. Diverting the use of by-products that are not currently used in other markets is less likely to have effects on land use, though there may be effects on land productivity and input use due to nutrient removal, as discussed above. It is also possible that creating a new demand for corn by-products (e.g., shells, husks, and cobs), for instance, would lead to more land moving into corn if the additional revenue available from corn by-products becomes sufficiently high.
- *Leakage:* If there is sufficient demand for agricultural industrial products and processing by-products for use in energy production, there could be leakage because these products are diverted from their current market uses.
- *Storage and handling losses:* It is possible that there would be some physical losses or decomposition of some feedstocks, with losses tending to increase with length of storage, other things being equal.

### 3.2.4. Dedicated Energy Crops

Dedicated energy crops, including switchgrass, miscanthus, energy sorghum, and short-rotation woody crops (e.g., poplar, willow), can be converted at stationary sources into heat and power, cellulosic fuels, and biodiesel. Direct and indirect land use change can occur as a result of dedicated feedstock production because existing forestlands, croplands, or grasslands would likely need to be converted to grow these feedstocks since they are not currently produced commercially at large scales in the United States.

- *Time scale over which feedstock carbon sequestration and emissions occur:* Growth and harvest of dedicated energy crops generally occur at time scales of a year or a few years, with short-rotation woody crops having the longest growth cycle in this subcategory (3 to 20 years, depending on site and management conditions). For some energy crops like sweet sorghum, sequestration and emissions related to a single rotation typically occur over the course of a few months to a year. For these energy crops on rotations similar to traditional agricultural crops, the net atmospheric biogenic contribution from the growth and harvest of the feedstock itself is considered in balance, though other factors such as land use management and land use changes and related soil carbon impacts can affect the overall net contribution outcome. For energy crops with rotations longer than a year, the rotation ages and harvest regimes (and thus resulting sequestration and emissions related to growth and harvest) depend largely on specific species and site and management conditions.[3] Many energy crops also have extensive root systems that are left during bole/limb harvests to regenerate for multiple cycles. These root systems and minimal/no tillage practices can lead to high levels of soil carbon sequestration over time. In some circumstances, dedicated feedstocks may have substantial direct land use and/or leakage impacts related to cultivation if its production has displaced other land use types (especially in the case of displaced forest), which may have implications for landscape carbon equilibrium over time.
- *Alternate fate pathways:* If not used for bioenergy, there would be no dedicated planting of energy crops aside from those cultivated currently for biofuel production, which is not yet conducted on large scales commercially.
- *Land use/land use management changes:* There can be direct land use change effects if the demand for dedicated energy crops causes changes in land use and land use management. Numerous alternative energy crops could be grown on land currently being used for production of other forestry or agricultural commodities. Changes in demand for energy crops could displace traditional crops and forest products, and these changes in cropping patterns and production practices can cause substantial GHG emissions changes. For example, land may be converted from traditional crops such as corn or soybeans to energy crops, which would lead to increases in soil carbon sequestration as well as other carbon

[3] Site requirements, regeneration potential, growth and yield estimates, pests, and fertilization regime all affect species growth and influence management decisions (in addition to cost for the latter). Management regimes for energy crops can vary widely, from active management to low-intensive/unmanaged regimes, which causes widely different growth and harvest-related emissions.

pools. Land converted from forests to energy crops can increase GHG emissions from the landscape initially.

- *Leakage:* Leakage effects related to dedicated energy crop production can be substantial. The new or diverted production of feedstocks can affect commodity markets and thus lead to changes in production that alter land uses and land use management outside of the energy feedstock supply chain (Murray et al., 2004; Searchinger et al., 2009; EPA, 2010a). For example, if corn-producing cropland is converted to a short-rotation woody crop for energy production, this change displaces corn from the marketplace and the corn could be replaced by production elsewhere. Even though there may be more carbon sequestration occurring on the production landscape (due to more carbon stored in root systems and soil carbon pools), there is potential significant leakage and related emissions effects elsewhere, particularly if forested or grassed lands are brought into crops.
- *Storage and handling losses*. Dedicated energy feedstocks generally need to be processed before they can be used for energy, which can lead to low levels of decomposition or physical feedstock losses. Additionally, because some energy crops have shorter rotations and thus a seasonal nature, these feedstocks may need to be stored to provide a year-round supply of energy. Therefore, energy crop biomass may experience more feedstock losses than forest biomass, on average. Dry matter losses during storage inside buildings are expected to be less than 5% (Collins et al., 1997; Huhnke, 2006; Shinners et al., 2007). Outside storage could lead to substantially greater losses because of the exposure to weather and increased losses to pests.

## 3.3. Waste-derived Feedstocks

A critical difference between waste and other biologically based material is related to the connection to the land providing the material. The biologically based material in waste is removed from land for other certain end uses (e.g., for manufacture of consumer and industrial products such as newspaper, food, and construction), after which it is disposed of. Given that the treatment of waste itself does not drive the management of the growth and harvesting of biomass, it is more difficult to quantify a connection between the consumption of waste-derived feedstocks at stationary sources and the landscape from which the biogenic component of the feedstock was originally produced.

The treatment of waste-derived feedstocks at a waste management system emits carbon as $CO_2$ (and $CH_4$) that would have otherwise been returned to the atmosphere as predominantly $CO_2$ from natural decay of waste, regardless of the management or status of the land providing the biological material. The human management of the waste materials affects only the timing or location of these GHG emissions.

In addition to biogenic $CO_2$ emissions, waste management systems can emit large quantities of $CH_4$ if they manage wastes under anaerobic conditions. Methodologies for estimating and accounting for $CH_4$ from waste management are available and widely used in many GHG accounting programs. Many waste systems already account for $CH_4$ using methodologies from the EPA Greenhouse Gas Reporting Program (GHGRP). The decision to consider avoided $CH_4$ emissions in an analysis should be made in the context of the type of baseline that is most appropriate given the policy context.

This framework considers the comparison of $CO_2$ emissions from waste management sources to the biogenic $CO_2e$ emissions implications associated with decomposition of the same waste in other types of managed systems. For example, an assessment of waste materials diverted from a landfill to an incinerator for energy production could consider the biogenic $CO_2$ emissions that occur at one point in time (at the incinerator) against the avoided $CO_2$ and $CH_4$ generated over decades through decomposition in the landfill and also avoided carbon storage in the landfill (EPA, 2010b, 2013; IPCC, 2006), or it could consider the biogenic $CO_2$ emissions against the $CO_2$ that would be emitted through the natural decay of the original biomass.

Feedstocks listed here are those that would have been produced in the waste sector regardless of any potential use for bioenergy production. In this way, they are not comparable to various residue feedstocks listed elsewhere in this document.

### 3.3.1. Municipal Solid Waste (MSW)

MSW includes waste generated by residential, commercial, and institutional entities. It contains a variety of biogenic materials, the composition of which varies by region, season, and long-term trends in waste generation. The average national composition of MSW in 2011 was estimated by EPA (2013) in Municipal Solid Waste Generation, Recycling, and Disposal in the United States Detailed Tables and Figures for 2011. The biogenic fractions of MSW include paper, food waste, yard waste, wood, diapers, natural fiber textiles, and natural rubber. MSW is defined slightly differently by different states. Both the overall carbon content and the ratio of fossil carbon to biogenic carbon of MSW vary widely. About one-third of MSW generated is recycled (including composting). Of the MSW that is not recycled, about 80% of MSW is disposed of through landfilling, and 20% of MSW is treated using combustion (EPA, 2013). Disposal of MSW in landfills causes some of the biogenic materials to be converted to methane; well over half of the methane ($CH_4$) generated in U.S. MSW landfills is captured for combustion (EPA, 2013).

- *Time scale over which feedstock carbon sequestration and emissions occur:* MSW waste disposed of in a landfill will degrade slowly over one to several decades. Food waste typically degrades more quickly than other MSW components like wood and paper wastes. The time scale of MSW degradation is also affected by the climate, with faster degradation occurring in moist, warm climates. It is generally estimated that half of the biogenic carbon disposed of in an MSW landfill will not degrade and will remain sequestered in the landfill. The time scale for composting MSW is on the order of 1 month. The time scale for MSW combustion is very short, on the order of minutes.
- *Alternate fate pathways:* Landfill gas generated as a result of the degradation of MSW in a landfill will generally contain about 50% methane and 50% $CO_2$ (by volume, dry basis). For landfills with no gas collection system, this landfill gas will be emitted to the atmosphere. For landfills with gas collection and combustion systems, a significant portion of the methane in the landfill gas can be converted to $CO_2$ prior to release into the atmosphere. Composting can be used as an alternative to MSW landfilling, particularly for food and yard wastes. Degradation of MSW in composts is primarily aerobic, but methane emissions occur as a result of anaerobic pockets that develop within the compost pile. Direct combustion of

MSW is another alternate pathway; essentially all of the carbon in MSW is converted and released as $CO_2$,

- *Land use/land use management changes*: Not applicable.
- *Leakage:* Not applicable.
- *Storage and handling losses*. Loss of MSW due to storage or transport of the MSW prior to receipt at the centralized treatment location is not considered. For example, only that portion of MSW as received at the landfill is considered in the methodology. All emissions from "storage" of the waste at an MSW landfill are considered part of the emissions source, whether the emissions are captured for combustion or uncaptured and released to the atmosphere. Storage of the waste at an MSW combustor facility is assumed to have negligible emissions because of the relatively short on-site storage times and is not specifically considered in the methodology.

### 3.3.2. Animal Agriculture Wastes

Livestock manure, litter, and manure wastewater are typically treated in a manure management system that stabilizes and/or stores wastes in one or more of the following system components: uncovered anaerobic lagoons, liquid/slurry systems with and without covers (including but not limited to ponds and tanks), storage pits, digesters, solid manure storage, dry lots (including feedlots), high-rise houses for poultry production (poultry without litter), poultry production with litter, deep bedding systems for cattle and swine, manure composting, and aerobic treatment units. International convention considers all carbon in animal agriculture waste to be biogenic (IPCC, 2006). Decomposition of the manure can occur through anaerobic or aerobic decomposition. Some manure management systems combust $CH_4$ from anaerobic treatment.

- *Time scale over which feedstock carbon sequestration and emissions occur:* Animal agriculture waste degradation typically occurs over several weeks to several months.
- *Alternate fate pathways:* Animal agriculture waste degradation typically generates predominately $CO_2$ (for aerobic systems) or a mixture of $CO_2$ and $CH_4$ (for anaerobic, facultative, or anoxic systems), depending on the type of manure management system used. When anaerobic treatment systems are used, the $CH_4$ in the biogas can be collected and combusted to $CO_2$. For dry animal agriculture wastes, direct combustion of the manure is an alternative treatment option, which will convert and release the biogenic carbon as $CO_2$ emissions.
- *Land use/land use management changes*: Not applicable.
- *Leakage:* Not applicable.
- *Storage and handling losses:* Most manure treatment occurs on site, and the methodology considers all emissions from the storage or treatment of the manure, whether the emissions are captured for combustion or uncaptured and released to the atmosphere. When centralized treatment is used, loss or degradation of animal agriculture wastes due to storage or transport prior to receipt at the centralized treatment location is not specifically considered; the methodology primarily considers only the quantity of waste as received at the centralized treatment location. Storage of the manure prior to direct manure

combustion is assumed to have negligible emissions due to the relatively short on-site storage times and is not specifically considered in the methodology.

#### 3.3.3. Wastewater

Wastewater is typically treated through processes that treat or remove pollutants and contaminants, such as soluble organic matter, suspended solids, pathogenic organisms, and chemical contaminants, from wastewater prior to its reuse or discharge from the facility. Sources include municipal and industrial wastewater treatment facilities. International convention considers all $CO_2$ generated as a result of aerobic wastewater treatment to be biogenic (IPCC, 2006). Some wastewater treatment facilities use anaerobic processes, which results in $CH_4$ emissions; some facilities capture and combust $CH_4$ derived from anaerobic digestion.

- *Time scale over which feedstock carbon sequestration and emissions occur:* Wastewater treatment processes degrade organic matter over a time scale of a few hours to a few days, although some large surface impoundments can have residence times as long as several months.
- *Alternate fate pathways:* Aerobic wastewater treatment processes convert the degradable organic matter almost entirely to $CO_2$; anaerobic, wastewater treatment processes convert approximately 50% or more of the organic carbon to $CH_4$; and facultative or anoxic wastewater treatment processes produce $CO_2$ predominately, but also produce appreciable amounts of $CH_4$. Gas generated in anaerobic wastewater treatment systems can be collected and combusted to convert the $CH_4$ in the biogas to $CO_2$ prior to release into the atmosphere.
- *Land use/land use management changes*: Not applicable.
- *Leakage:* Not applicable.
- *Storage and handling losses:* Most wastewater treatment operations occur on site, and the methodology considers all emissions from the storage or treatment of the wastewater, whether the emissions are captured for combustion or uncaptured and released to the atmosphere. When centralized treatment is used, loss or degradation of degradable carbon in the wastewater due to storage or transport prior to receipt at the centralized treatment location is not specifically considered; the methodology primarily considers only the quantity of wastewater as received at the centralized treatment location.

# 4. Other Possible Feedstock Categories

## 4.1. Secondary Use Feedstocks

Secondary use feedstocks are feedstocks that leave the stationary source where the original biogenic feedstock material is transformed into a product or by-product (i.e., primary stationary source) that is used for energy production at a different stationary source (i.e., secondary stationary source). For example, if a secondary stationary source chooses to use for energy an agriculture- or forest-derived industrial processing product such as distillers grains or woody residuals from a primary stationary source, the net biogenic emissions value (landscape and process attributes from the original feedstock at the primary source) for the biogenic feedstock carbon of that material could be used at the secondary entity. Hypothetically, the same treatment could be used in the case of energy products such as pellets or ethanol used for energy at a secondary stationary source.

Secondary use feedstocks could be addressed in many different ways, and different policy applications of the framework may necessitate certain treatments. One consideration when applying the framework in specific policy contexts is how to avoid possible instances of double-counting landscape and process attribute values as they relate to biogenic materials used or processed at the primary stationary source and transferred for use at a secondary stationary source. For example, counting total biogenic feedstock attributes values at both the primary and secondary stationary sources could result in double-counting these values. For a more detailed example, see Appendix F. Evaluation of all the different possible policy treatments for secondary feedstocks is outside the scope of this report.

## 4.2. Feedstock Imports and Exports

International feedstock production and the importation of those feedstocks can significantly affect overall biogenic feedstock resource availability in the United States and demand pressures on those resources. The pricing and flow of feedstocks and related commodities have the potential to significantly affect domestic supply chains, and, conversely, U.S. biogenic feedstock production can affect international commodity markets and land use activities. The framework could either include or exclude international biogenic feedstocks, depending on policy requirements and international agreements related to GHG emissions accounting. However, decisions as to the inclusion/exclusion of international feedstocks and the treatment of such feedstocks if included would depend on the specific application of the framework. The general framework description and illustrative examples given in this report do not address the export of U.S. feedstocks or the import of feedstocks produced abroad.

## 4.3. New, Unconventional, or Otherwise Unanticipated Feedstocks

The framework is designed to be flexible so it can be modified as needed to be applicable to nearly all domestic biogenic feedstocks currently in use or under consideration for bioenergy production. However, new and unconventional or otherwise unanticipated feedstocks may emerge over time. Feedstock categorization in the framework should be broad enough that any feedstocks not already included can fit into the predefined categories if possible and if not, new categories could be made. The current feedstock categories are structured with this in mind, though some emerging feedstocks (e.g., algae) will require additional parameters that could be added on an as-needed basis.

For purposes of this report, biogenic feedstocks have been classified broadly into the feedstock categories identified above based on the physical attributes those feedstocks possess. This categorization does not represent a formal or legal definition, nor does it intend to replace any existing legal definitions. For example, under existing regulations, certain feedstocks are already regulated using specific definitions. Thus, although tire-derived fuel might fall under the waste-derived feedstock categorization in the framework presented here because of its attributes relative to biogenic $CO_2$ accounting, it is classified as a fuel under other policy applications. The same might be true for other feedstocks.

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# 6. Addendum: Spent Pulping Liquor—Overview of Processes and Possible Alternate Fates

## 6.1. Joint Production of Industrial Products and By-products

Some industries may produce by-products useful for bioenergy generation in the process of making their primary market products. For example, manufacturers may generate agricultural by-products such as shells, husks, and cobs that have little or no market value and may be disposed of if not used for energy. A second example is kraft pulp mills, which generate black liquor that is burned on site to recover chemicals for reuse in the pulping process as well as to produce energy. Other than for on-site use within the pulp mill (i.e., captive use within the mill), black liquor has no commercially feasible alternative use. Unlike by-products in other industries, black liquor is not only limited in commercially viable alternative uses, but it also has limited disposal options (Gaudreault et al., 2012).

Forest products, in general, are characterized by a joint production function, because many products can be produced from a single tree. Firms strive to optimize production to maximize the amount of high-value products (e.g., saw lumber, paper) and minimize the amounts of low-value products (e.g., black liquor). Although there is some responsiveness to relative price movements (e.g., higher demand and prices for wood pellets may lead to an increased proportion of scrap going to this use and a decreased proportion going to particleboard), the elasticity of transformation between outputs may be very inelastic, and even with a negative price, some low-value products would still necessarily be produced as a by-product of the production of high-value products (e.g., sawdust, black liquor).

When estimating the landscape biogenic emissions outcome from biomass production and usage, the SAB Panel advocated the use of a future anticipated baseline approach that would capture additionality—"i.e., the extent to which forest carbon stocks would have been growing or declining over time in the absence of harvest for bioenergy." Capturing additionality requires that the framework model "a 'business as usual' scenario along some time scale and compare that carbon trajectory with a scenario of increased demand for biomass" (Swackhamer and Khanna, 2011). Because of the joint-production nature of the forest products industry, care needs to be taken in how a "scenario of increased demand for biomass" is created.

In a partial equilibrium framework, there are different options for simulating an increase in demand for a biogenic feedstock relative to a "business as usual" case. One option is to introduce into, or shock, a model with *X* additional tons of production of a specific biogenic feedstock, which will likely result in increased production of all the other products that are jointly produced with that bioenergy feedstock. The landscape effects associated with increased production of the specific biogenic feedstock would be conflated with the effects of the increased production of other jointly produced products. Another option is to shock a model with an increased price for the specific biogenic feedstock of interest (mechanically, this could be achieved in the model with a subsidy for the specific feedstock). If the feedstock of interest is jointly produced as a by-product from the production of other products, the increase in price for the specific feedstock is not likely to increase production of or increase procurement of raw materials to generate that by-products-based biogenic feedstock. For black liquor, it would be expected that a model run that increased the demand for and price of black liquor explicitly would result in very little change in forest harvest and land use decisions, because even a large increase in the price for black liquor would have a small impact on the value of the overall product mix.

For residues and materials diverted from the waste stream, the SAB Panel endorsed considering their alternate fates (e.g., some forest residues may be burned if not used for bioenergy, waste-derived materials might be otherwise landfilled) and information about decay (e.g., using decay functions to evaluate ecosystem carbon storage in forest residues not burned for energy or disposal). Furthermore, in the case of waste-derived materials, the SAB stated that "after calculating decay rates and considering alternate fates, including avoided methane emissions, the agency may wish to declare certain categories of feedstocks with relatively low impacts as having a very low [biogenic accounting factor], or setting [biogenic accounting factors] equal to 0 or possibly negative values in the case where methane emissions are avoided" (Swackhamer and Khanna, 2011). In terms of net atmospheric contributions of biogenic $CO_2e$[4] emissions, as shown in the following sections of this addendum, it is more beneficial that black liquor is burned for on-site energy use and chemical recovery because black liquor has no other commercially viable alternative uses and no practical disposal options. It is also unlikely that changes in demand for or prices of black liquor would lead to changes in land use, harvest, or forest management decisions because black liquor is not produced for its value alone but is only produced as a by-products of manufacturing high-value pulp for use in papermaking.

The purpose of this addendum is to provide background information on the chemical pulping process leading to generation of black liquor and to explore hypothetical alternate fates of black liquor in the context of biogenic $CO_2e$ accounting and avoided emissions. The information in this appendix, including example calculations of alternate fate-related biogenic emissions, supports that a 0 or negative assessment factor for black liquor may be reasonable. This finding is based on the joint function production rationale presented above; the related expectation that there would not be any perverse outcomes with respect to land use, harvest, or forest management decisions; and an analysis of hypothetical potential alternate fates and related avoided emissions.

### 6.2. Overview of Pulping Processes

The pulp and paper industry consists of facilities that manufacture pulp and/or paper. Pulp is the fibrous raw material for papermaking (Smook, 2002). Pulp is manufactured using either chemical or non-chemical pulping processes. Paper can be manufactured at mills that also produce virgin pulp or at mills that do not produce pulp but instead purchase pulp or use recycled fiber to manufacture paper. Some mills produce only market pulp to sell to other mills and do not manufacture paper.

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[4] This addendum generally uses $CO_2e$ in the context of biogenic emissions because both $CO_2$ and $CH_4$ are specifically discussed.

Different processes are used for pulp production. Kraft pulping is by far the most common pulping process used by plants in the U.S. for virgin fiber. The kraft pulping process produced approximately 86% of all U.S. pulp tonnage in 2010. Non-chemical pulping processes are also used in the U.S., including mechanical pulping and secondary (recycled) fiber pulping. Non-chemical processes do not produce spent pulping liquor. Approximate percentages of U.S. pulp production for other processes in 2010 were sulfite pulping (1%), semi-chemical pulping (6%), and mechanical pulping (8%) (RISI, 2011). Thus, the remainder of this addendum focuses on the kraft pulping process.

Chemical (i.e., kraft, soda, and sulfite) pulping involves “cooking” of raw materials (e.g., wood chips) using aqueous chemical solutions and elevated temperature and pressure to extract pulp fibers. The kraft pulping process uses an alkaline cooking liquor (known as “white liquor”) of sodium hydroxide (NaOH) and sodium sulfide ($Na_2S$) to digest wood, while the similar soda process uses only NaOH to digest the wood. The cooking liquor in the sulfite pulping process is an acidic mixture of sulfurous acid ($HSO_3$) and bisulfite ion ($HSO_3^-$). The bases used in cooking liquor preparation are typically calcium, magnesium, and sodium. Semi-chemical pulping uses a combination of chemical and mechanical (i.e., grinding) energy to extract pulp fibers. The chemical portion (e.g., cooking liquors, process equipment) of the pulping process and pulp washing steps are very similar to kraft and sulfite processes.

For economic and environmental reasons, chemical and semi-chemical pulp mills employ chemical recovery processes to reclaim spent cooking chemicals. Typically, a combustion unit (e.g., recovery furnace) is used to recover the cooking chemicals from spent cooking solutions (or liquors). Kraft and soda mills have an additional chemical recovery process in which a lime kiln is used to regenerate a portion of the chemical cooking solution. In addition to spent pulping liquor, other by-products such as turpentine, soap, and tall oil may be produced during the kraft pulping process and are typically sold commercially.

## 6.3. Generation of Pulp and Black Liquor

Wood is the predominant raw material used for pulp and papermaking in the U.S. Following wood input procurement and handling operations, digestion of wood chips is the first step in kraft pulping. As shown in Figure D-1, the kraft digester has two primary products: pulp and black liquor. About half of the wood input to the kraft pulping process (digester) is dissolved into the solution of spent pulping chemicals to form weak black liquor. The weak black liquor is separated from the pulp by washing and is concentrated in evaporators and sent to the kraft recovery furnace where inorganic pulping chemicals are recovered for reuse and dissolved organics (e.g., lignin) are burned for fuel to make steam and electricity.

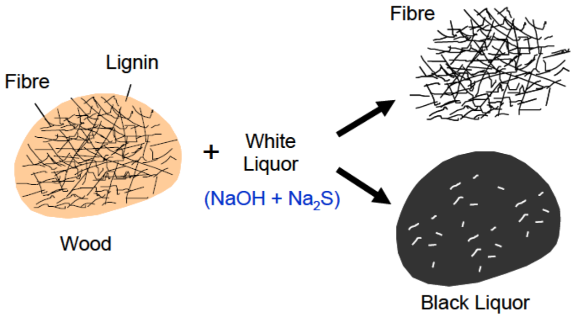


**Figure D-1. Products of the Kraft Mill Digester (Tran and Vakkilainnen, 2008).**

The amount of wood input to the pulping process that becomes wood fiber in the digester is referred to as pulp yield. The remainder of the wood can generally be assumed to partition to the black liquor.

The yield of pulp (expressed as a percentage by weight on a moisture-free basis) obtained from a given species of wood is influenced by the severity of the pulping process used. A yield figure of 50% means that 1 ton of dry wood yields 0.5 tons of dry pulp.[5] Mechanical pulping processes have yields of 90 to 95%, while chemical processes have yields of 40 to 55%, and semi-chemical processes have yields of 50 to 80%. Typical pulp yields for the kraft process are 40 to 50% (Smook, 2002). The lowest yield is obtained from drastic chemical digestion that gives pulp consisting of nearly pure cellulose fibers. The yield of pulp obtained by digesting wood also depends on the chemical composition of the wood. Because the principal chemical components of the normal roundwood of most species do not vary much in amount, the percentage yield of pulp obtained by a given process does not vary greatly from one species to another. The chemical composition of wood, and consequently the yield of chemical pulp, generally varies more between softwoods (coniferous woods) and hardwoods (broadleaf woods) as classes than between the individual woods within these classes (USDA, 1980).

Black liquor solids (BLS) refers to the dry weight of the solids in the black liquor that enters the chemical recovery furnace. BLS contain the spent cooking chemicals (inorganics) and dissolved organics. The BLS exiting the washer in the weak black liquor are sent for further processing in the kraft recovery system, which regenerates the inorganic pulping chemicals, burns the dissolved organics for energy production, and sometimes results in the recovery of other organic by-products. Weak black liquor exiting the washing process typically contains 13 to 17% BLS (Smook, 2002). Weak black liquor is evaporated and concentrated to 65 to 80% solids prior to burning in the recovery furnace.

A nominal value of 1.6 tons of BLS per air-dried short ton of pulp (ADTP) is often used to represent BLS production, though this value ranges from approximately 1.3 to 1.9 ton BLS/ton pulp (1,300 to

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[5] Note that commercial pulp production rates are usually referred to in "air dried" units, which are generally considered to have a moisture content of 10%. "Moisture-free" or "bone-dried" mass refers to pulp or wood at 0% moisture.

1,900 kg solids/metric ton of pulp) (NCASI, 2011). The heating value of black liquor ranges from 5,400 to 6,600 btu/lb BLS (NCASI, 2011).

The chemical composition of black liquor varies based on its solids content. The solids alone comprise a complex mixture of both inorganic and organic constituents. The inorganic constituents in black liquor are derived from the cooking liquor and comprise various sodium and sulfur compounds, including NaOH, sodium sulfate ($Na_2SO_4$, also known as "salt cake"), $Na_2S$, sodium thiosulfate ($Na_2S_2O_3$), sodium carbonate ($Na_2CO_3$), and sodium chloride (NaCl). Collectively, inorganic salts constitute between 18 and 25% of the solids in black liquor.

The organic compounds found in black liquor are derived from wood. They are either (1) natural wood extractives (or their reaction products) that are released as a result of the pulping process or (2) materials formed through the reactions of the pulping chemicals with the lignin or cellulose components of wood. Therefore, the compounds can be classified as lignin derived, cellulose derived, or extractives derived. Typical content ranges in kraft liquor are:

- Lignin derived (39 to 54%); primarily consisting of polyaromatic macromolecules with lesser amounts of molecular weight alcohols, aldehydes, and simple phenolic compounds such as phenol, p-methyl phenol, catechol, and guaiacol;
- Cellulose derived (25 to 35%); primarily a mixture of carboxylic acids such as formic, acetic, glycolic, lactic, and glucoisosaccharinic; and
- Extractive derived (3 to 5%); primarily resin acids and fatty acids that are converted to salts at the high pH of the mixture.

In sum, spent pulping liquor can have hundreds of constituents (AF&PA, 2001). The exact composition depends on wood type, the concentration of the components in the white liquor used to digest the wood chips, and the actual process parameters. Information on different estimates of the elemental composition of black liquor is shown in Table D-2.

**Table D-2. Elemental Composition of Black Liquor (Weight % of BLS).**

| Element | TAPPI (Clay, 2008) | GA Tech (IPST, Undated) | EPA (EPA, 1997 ) |
|---|---|---|---|
| Carbon, C | 35 | 34–39 | 35.2 |
| Hydrogen, H | 3.3 | 3–5 | 3.6 |
| Oxygen, O | 35.7 | 33–38 | 35.2 |
| Sodium, Na | 19.7 | 17–25 | 19.2 |
| Sulfur, S | 4 | 3–7 | 4.8 |
| Potassium, K | 1.6 | 0.1–2 | 1.0 |
| Chloride, Cl | Unspecified | 0.2–2 | 0.1 |
| Nitrogen, N | Unspecified | 0.04–2 | Unspecified |
| Others, including non-process elements (Ca, Al, Si, Fe) | <1 | 0.1–0.3 | 0.2 |
| Carbonate, $CO_3$ | 8 | Included in C and O above | |
| Sulfate, $SO_4$ | 3 | Included in S and O above | |

## 6.4. Biogenic $CO_2$ Emissions from Kraft Black Liquor Chemical Recovery

To assess biogenic $CO_2$ emissions associated with the kraft pulping process, it is helpful to understand how biogenic $CO_2$ emissions are currently estimated for kraft pulp mills. Carbon originating in the wood input into the pulping process primarily apportions to the pulp product and black liquor, as shown in Figure D-1 above.

The kraft chemical recovery process can be depicted by two interconnected loops, the sodium loop and calcium loop, as shown in Figure D-2. In the kraft pulping and chemical recovery process, biomass carbon from the wood is dissolved and either emitted as biomass $CO_2$ from the recovery furnace or captured in sodium carbonate ($Na_2CO_3$) in the smelt discharged from the bottom of the recovery furnace. In the process of converting the $Na_2CO_3$ into new pulping chemicals, this biogenic carbon (i.e., the carbonate ion) is transferred to calcium carbonate ($CaCO_3$). In the lime kiln, the $CaCO_3$ is converted to calcium oxide (i.e., CaO or lime, a material used in the chemical recovery process) and biogenic $CO_2$, which is released to the atmosphere (EPA, 2009).

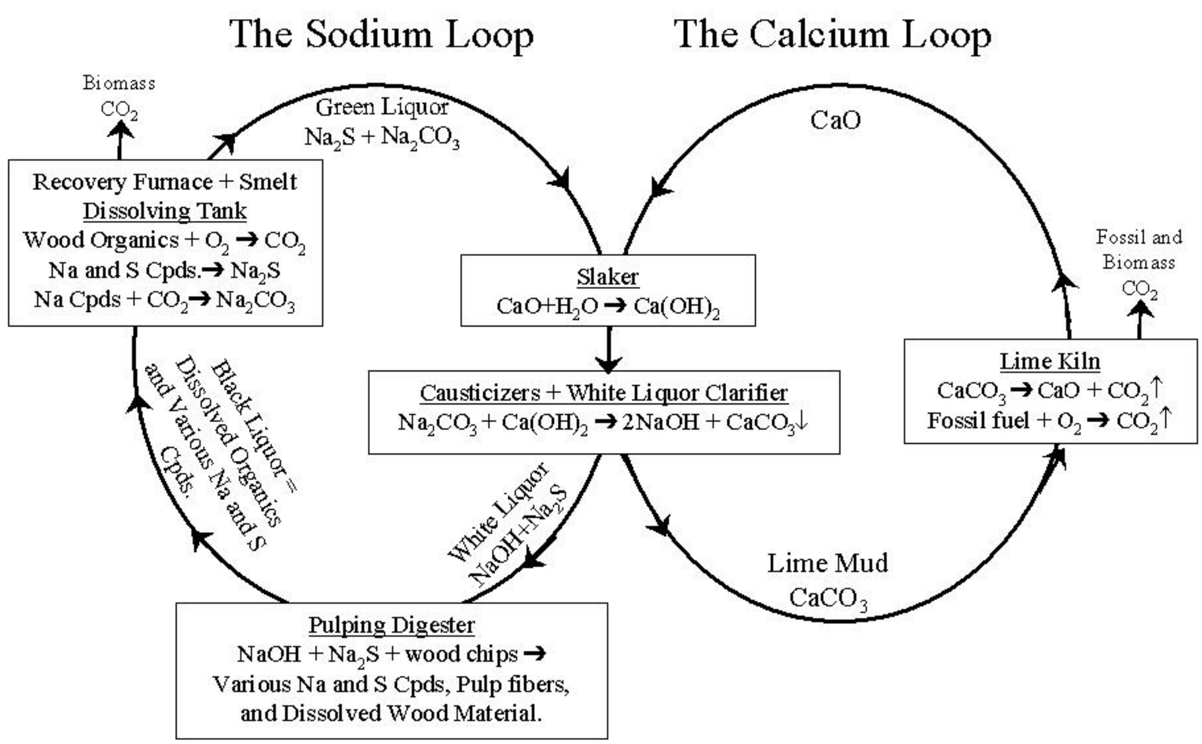


**Figure D-2. Simplified Representation of the Kraft Pulping and Chemical Recovery System (EPA, 2009).**

The majority of the wood-derived carbon within the black liquor either:

- Exits the pulping process as biogenic $CO_2$ emissions from the recovery furnace stack; or
- Reacts with sodium compounds to form $Na_2CO_3$ in the smelt that exits the bottom of the recovery furnace as smelt.

The carbonate in the smelt makes its way through the chemical recovery loop to the lime kiln (becoming $CaCO_3$ along the way). In the lime kiln, the $CaCO_3$ is converted to CaO, emitting the biogenic $CO_2$ originating from the wood in the black liquor. The red text in the simplified diagram below shows what happens to the wood-derived carbon (fossil $CO_2$ emissions from the recovery furnace and lime kiln are not shown).

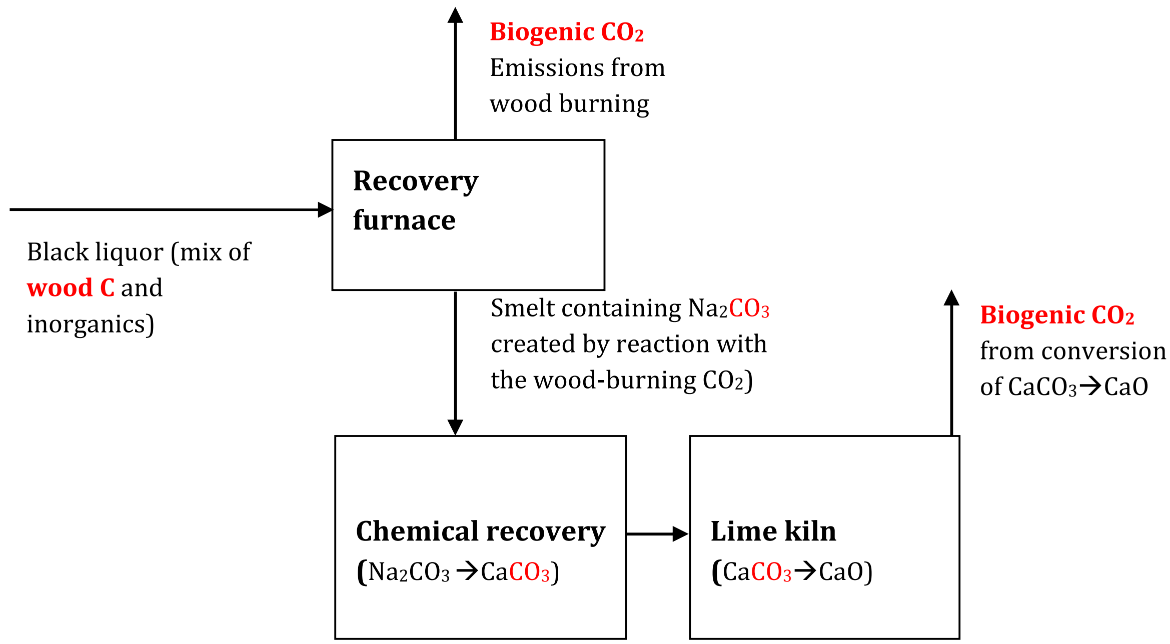


**Figure D-3. Simplified Diagram Showing How Biogenic $CO_2$ Emissions Are Emitted from Both Kraft Recovery Furnace and Lime Kiln.**

The emission factors used to estimate biogenic $CO_2$ emissions from the black liquor chemical recovery process are based on the carbon content of black liquor and therefore account for biogenic $CO_2$ emissions from both the recovery furnace and lime kiln. Thus, rather than depicting the biogenic $CO_2$ accounting boundary around the recovery furnace as the sole biogenic $CO_2$ emissions unit, it would be more consistent with the current biogenic $CO_2$ emissions estimation practice (including that required under the U.S. EPA's Greenhouse Gas Reporting Program) to consider the biogenic $CO_2$ emissions unit as the entire pulping process.

Fossil-fuel-related $CO_2$ emissions are estimated separately for the recovery furnace and lime kiln. The only other non-biogenic carbon introduced into the process is from carbonated makeup chemicals, for which $CO_2$ emissions are estimated using a mass balance approach. Fossil-fuel and makeup chemical $CO_2$ emissions estimates are independent of the biogenic $CO_2$ accounting method presented in this report and need not be discussed further in this document.

## 6.5. Alternate Pathways for Black Liquor

Consideration of alternate fate pathways for black liquor (as opposed to reuse within the kraft process) is purely hypothetical for U.S. mills because U.S. mills have taken steps to maximize recovery of black liquor. If black liquor is not reused within the pulping process and instead is disposed of, disposal methods might involve incineration without the benefit of energy recovery or discharge of the black liquor in liquid form (e.g., weak black liquor) into a wastewater treatment system or lagoon. The benefits of recovering black liquor include:

- Energy value;
- Avoided cost of replacement chemicals, primarily equivalent saltcake;
- Reduction in biological oxygen demand load on the effluent treatment system; and
- Reduction in color and chemical oxygen demand (COD) discharge in the treated effluent (EPA, 1997).

For many reasons U.S. mills have opted to use chemical recovery furnaces as opposed to disposing of black liquor. Depending on its volume and concentration, if released into the environment, black liquor can be odorous, toxic to aquatic life, and cause a dark caramel color in water (EPA, 1997). Black liquor contains sulfur from the kraft pulping process, which results in malodorous total reduced sulfur emissions detectable via olfactory senses in very low concentrations. Although the cellulosic constituents in black liquor may be biodegradable, lignin is very difficult to biodegrade, leaving a portion of the COD and much of the dark brown color to be discharged to receiving waters following wastewater treatment. EPA has established Best Management Practice regulations to protect the environment from the negative consequences of spent pulping liquor spills (40 CFR 430.03, 430.28, and 430.58). These rules were promulgated in 1998 as part of the effluent guidelines and standards for the pulp, paper, and paperboard source category (40 CFR Part 430) developed under the Clean Water Act.

In the hypothetical alternate fate pathway examples below, equations from Appendix N of this document were used to estimate emissions from treatment of black liquor sent for incineration without energy recovery or wastewater treatment. The black liquor is the biogenic feedstock in this example (as opposed to the wood input to the pulping process) because the black liquor (which is a complex mixture of organic and inorganic chemicals) differs from the wood input to the pulping process both chemically and physically and is generated within the pulping process. As explained in Appendix N, the following terms can be dropped from the biogenic assessment factor (*BAF*) equation when conducting an alternate fate analysis: Net feedstock growth on the production landscape (*GROW*); total net change in production site non-feedstock carbon pools (*SITETNC*); leakage associated with feedstock production (*LEAK*); the feedstock carbon losses during storage, transport and processing ($L$); and the feedstock carbon embodied in products ($P$). As a result, the assessment framework equation as applied to estimating biogenic $CO_2$ emissions from the alternate fate of black liquor feedstocks can be simplified to:

***BAF = AVOIDEMIT***

*AVOIDEMIT* represents the avoided biogenic emissions that could have occurred per an alternative management strategy instead of the feedstock's use in bioenergy production, relative to the feedstock's use for bioenergy production. The *AVOIDEMIT* term, as applied to the black liquor biogenic feedstock, is expressed as:

$$\boldsymbol{AVOIDEMIT = 1 - \frac{\textbf{CO}_2\textbf{e emissions from treatment alternative to combustion}}{\textbf{CO}_2\textbf{e emissions from combustion treatment}}}$$

The *AVOIDEMIT* term must be calculated for the specific feedstock being managed relative to a specific, alternative practice. A positive *AVOIDEMIT* value implies that use of the feedstock for bioenergy production contributes more emissions to the atmosphere than would have occurred under the alternative management strategy. A 0 value implies that both practices are equivalent in terms of how much emissions they contribute to the atmosphere. A negative value implies that using the feedstock for bioenergy production contributes less emissions to the atmosphere than the alternative management practice.

The $CO_2$e emissions in the *AVOIDEMIT* equation above include both $CO_2$ and methane ($CH_4$) resulting from each feedstock management alternative. The following subsections present calculations of $CO_2$e emissions for the two hypothetical alternative management approaches.

### 6.5.1. Alternate Fate Scenario 1: Black Liquor Incineration without Energy Recovery

Table D-3 presents emission factors and calculated emissions for incineration of black liquor normalized per ADMT of pulp produced. The emissions from incinerating black liquor without energy recovery would be the same as the emissions from burning the material in a recovery furnace. The inorganic chemicals in the smelt produced from burning the black liquor would remain and could be recovered in either case. No emissions would be avoided by treating the black liquor through incineration, and the stack emissions would be about the same as combustion without energy recovery. Therefore, the resulting hypothetical example BAF is 0.

**Table D-3. *AVOIDEMIT* for Scenario 1 Where Black Liquor Is Incinerated Without Energy Recovery.**

| | | |
|---|---|---|
| **Emissions Factors:** | | |
| **$CO_2$** | 94.4 | kg/MMBtu HHV (40 CFR 98, subpart AA) |
| **$CH_4$** | 0.0019 | kg/MMBtu HHV (40 CFR 98, subpart AA) |
| **Process Parameters:** | | |
| | 1.6 | Ton BLS/ADTP (NCASI, 2011) |
| | 0.9072 | ADMT pulp/ADTP (conversion factor for metric tons: short tons) |
| | 1.8 | Ton BLS/ADMT pulp (calculated) |
| | 6600 | Btu/lb BLS based on NCASI, 2011 range of 5,400 to 6,600 Btu/lb for BLS |
| | 13.2 | MMBtu/ton BLS (calculated) |
| **$CO_2$** | 2.20 | Metric tons $CO_2$/ADMT pulp (calculated) |
| **$CH_4$** | 4.42E-05 | Metric tons $CH_4$/ADMT pulp (calculated) |
| **$CH_4$ GWP** | 25 | (40 CFR 98, subpart AA) |
| **$CO_2$e** | 2.199 | Metric tons $CO_2$e/ADMT pulp if the black liquor is burned |
| ***AVOIDEMIT* numerator** | 2.199 | Emissions from treatment alternative (incineration without energy recovery) |
| ***AVOIDEMIT* denominator** | 2.199 | Emissions from burning black liquor for chemical and energy recovery, metric tons/ADMT pulp |
| ***BAF = AVOIDEMIT*** | 0 | = 1− (emissions from treatment alternative) / (emissions from burning black liquor for chemical and energy recovery) |

### 6.5.2. Alternate Fate Scenario 2: Black Liquor Decomposition in a Wastewater Treatment System

The equations related to $CO_2$ and $CH_4$ emissions from wastewater treatment in Section 6.1 of Appendix N were used to evaluate two hypothetical wastewater treatment scenarios for the disposal of black liquor: (1) treatment in a deep anaerobic lagoon; and (2) treatment under aerobic conditions. Under both alternate pathway conditions (anaerobic and aerobic), emissions from treatment of black liquor as wastewater exceeded those from the current practice of burning black liquor for chemical and energy recovery. The resulting hypothetical example BAF varied from slightly negative (−0.09) to −1.2 depending on the wastewater management method, indicating that using the black liquor feedstock for bioenergy production contributes less biogenic emissions to the atmosphere than the alternative wastewater management practice. Table D-4 presents the equation terms and calculated values for avoided emissions from wastewater treatment.

**Table D-4. *AVOIDEMIT* for Scenario 2 Where Black Liquor Is Disposed Through Wastewater Treatment.**

| Equation Term | Description | Anaerobic Deep Lagoon Treatment | Aerated Treatment Process with Anoxic Areas |
|---|---|---|---|
| **Input Values** | | | |
| $Q_{ww}$ | Wastewater influent flow rate | 9.7 $m^3$/ADMT pulp[1] | 9.7 $m^3$/ADMT pulp[1] |
| **OD** | Oxygen demand of influent | 92,700 mg/l[2] | 92,700 mg/l[2] |
| **EffOD** | Oxygen demand removal efficiency of the biological treatment unit | 0.75[3] | 0.75[3] |
| $MCF_{ww}$ | $CH_4$ correction factor (from Appendix N, Table N-13) | 0.8 | 0.3 |
| $BG_{CH4}$ | Fraction of C as CH4 in generated biogas (default is 0.65) | 0.65 | 0.65 |
| $\lambda$ | Sludge biomass yield (from Appendix N, Table N-13) | 0 | 0.45 |
| $GWP_{CH4}$ | Methane global warming potential | 25 | 25 |
| **Calculated Values** | | | |
| $CO_{2WW}$ | $CO_2$ emission rate (metric tons $CO_2$/ADMT pulp) | 0.45 | 0.41 |
| $CH_{4WW}$ | $CH_4$ emission rate (metric tons $CH_4$/ADMT pulp | 0.18 | 0.036 |
| $CO_{2S}$ | $CO_2$ emission rate (metric tons $CO_2$/ADMT pulp) | Not applicable ($\lambda = 0$) | 0.34 |
| $CH_{4S}$ | $CH_4$ emission rate (metric tons $CH_4$/ADMT pulp | Not applicable ($\lambda = 0$) | 0.03 |
| ***AVOIDEMIT* numerator** | $= CO_{2WW}+CO_{2S} + (CH_{4WW}+CH_{4S})*GWP_{CH4}$ | 4.8 | 2.4 |
| ***AVOIDEMIT* denominator** | Emissions from burning black liquor for chemical and energy recovery, metric tons/ADMT pulp | 2.2 | 2.2 |

| Equation Term | Description | Anaerobic Deep Lagoon Treatment | Aerated Treatment Process with Anoxic Areas |
|---|---|---|---|
| ***BAF = AVOIDEMIT*** | = 1− (emissions from treatment alternative) / (emissions from burning black liquor for chemical and energy recovery) | −1.2 | −0.09 |

[1] Based on typical industry parameters of 1.6 tons BLS/ADTP and assuming that weak black liquor is 15% solids (i.e. 0.15 ton BLS/ton liquor) with a density of 1.1 g/cm$^3$. Thus: 9.7 m$^3$/ADMT pulp = (1.6 tons BLS/ADTP) x (ADTP/0.9072 ADMT) x (ton liquor/0.15 ton BLS) x (907185 g/ton liquor) x (cm$^3$/1.1 g) x (m$^3$/100$^3$cm$^3$)

[2] Chemical oxygen demand for weak black liquor derived from pine (Yang, 2003).

[3] A relatively low oxygen demand removal efficiency of 75% was used considering the low biodegradability of the lignin component in the wastewater stream.

### 6.5.3. Discussion

The two scenarios presented above illustrate that current black liquor management practices (burning for energy and chemical recovery in a recovery furnace) result in the same or less emissions than the hypothetical alternate fate for black liquor (disposal via incineration or wastewater treatment). Black liquor is produced as a by-product from the pulping process and, under current market conditions, has no commercially viable alternatives other than use for chemical and energy recovery within the pulping process. Because black liquor is jointly produced with other forest industry products (pulp production), black liquor management practices and increased demand and/or price for black liquor are likely to result in no to little change in land use, harvest, and forest management decisions. The avoided emissions associated with disposal of black liquor as compared to the current management practice (burning for energy and chemical recovery in a recovery furnace) resulted in hypothetical example BAFs ranging from different negative values to 0, depending on the treatment method. Because of the joint production function rationale as well as the hypothetical alternate fate example calculations conducted above, an estimated BAF of 0 can be considered for black liquor and is consistent with the approach suggested by the SAB Panel for materials diverted from the waste stream (Swackhamer and Khanna, 2011).

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# Appendix E: Discussion of Leakage Literature

## Table of Contents



## 1. Introduction

The purpose of this appendix is to review the literature that discusses greenhouse gas (GHG) emissions leakage, which is an indirect source of biogenic $CO_2$ emissions associated with the production, processing, and use of biogenic material at stationary sources. Specifically, this appendix examines a range of studies that, using a variety of modeling approaches and scenario designs, evaluate how and why leakage occurs. There is a particular focus on indirect land use change (ILUC), which is an important form of leakage to consider when assessing biogenic $CO_2$ emissions from stationary sources. This examination is intended to help identify important factors that could be considered when assessing leakage and different methods that have been used to calculate emissions leakage in other contexts. In the event that policy- or program-specific applications of the framework necessitate calculations of leakage, the analysis here could inform such a process. This appendix considers both international and domestic (interregional) leakage for completeness.

Recognizing that leakage associated with bioenergy feedstock production can occur due to market and land use change induced by displaced feedstock or feedstock substitute production, the framework equation presented in this report includes a leakage term.[1] However, a specific quantification methodology recommendation is not provided in this report: the determination to estimate and include leakage in applications of the framework, as well as the methods to calculate it, will be policy- or program-specific.

The potential importance of leakage can vary across feedstocks and production circumstances. It is important to recognize that biogenic assessment factor (*BAF*) results may differ considerably for some feedstocks depending on treatment of leakage, but quantifying it is complex, as discussed in the literature review below.

The remainder of this appendix discusses the primary factors that typically contribute to leakage, provides an overview of relevant literature, and offers examples of different leakage analyses that have been conducted in different policy contexts and with different goals, assumptions, and parameters.

## 2. Background

Policies and programs typically have limited spheres of direct influence or scope and therefore may result in changes in activities outside their scope that can contribute to the net impacts of the action. Leakage is an indirect consequence of policies or behaviors that can and occur in many different contexts. Leakage effects could be positive (e.g., benefits of local tourism extending beyond the region or technological innovation spreading from one firm to others) or negative (e.g., reduced deforestation in one region is at least partially offset by increased deforestation in other regions as output prices rise). In the context of environmental policy, one of the key areas in which leakage has been examined in recent years is displacement of GHG-emitting activities to areas and/or sectors that are not covered by a policy, or program (Barker et al., 2007; Weber and Peters, 2009; Chen, 2009). There are different definitions of carbon leakage in the literature, but the International Panel on Climate Change (IPCC) Special Report on Land Use, Land-Use Change, and Forestry defines carbon leakage as "the indirect impact that a targeted LULUCF activity in a certain place at a certain time has on carbon storage at another place or time" (IPCC, 2000).

In the context of the framework and its focus on biogenic carbon and $CO_2$[2] emissions fluxes, leakage represents any biogenic $CO_2$ flux changes outside of a biogenic feedstock production assessment scope that can be attributed to the production activities (e.g., replacement of diverted crop, livestock, or forest products on other lands due to a change in land use from conventional commodity production to biogenic feedstock production for energy conversion).

If the assessment scope of the policy or program was global, then there would be no leakage because all emissions would be inherently captured within the assessment scope. In practice,

[1] The *LEAK* term could be incorporated into the retrospective reference point and future anticipated baselines in different ways. For further information on the retrospective reference point baseline, see Appendix H. For further information on the future anticipated baseline, see Appendix J.
[2] The framework could potentially be expanded to include additional GHGs as appropriate for a particular policy application.

however, project assessment scopes are typically more limited. In some cases, especially where policies result in substantial price effects, there may be changes in activities and emissions that take place outside the defined assessment scope. The reason is that activities in regulated sectors/regions tend to shift to, or have influences on, sectors/regions outside the regulatory and/or assessment framework, particularly if there is reduced product availability from the regulated sectors/regions. In that case, there will tend to be price increases that will induce expanded production in other related sectors and in other regions.

Although outside of a project's direct control, emissions that are shifted to another location or sector may have an important effect on the project's net GHG benefits. One of the more important sources of leakage for programs or policies affecting land use is the impact on carbon storage due to shifting land use. Where land is changing uses due to these indirect pressures, this specific leakage effect is commonly referred to as ILUC. Depending on the lands being converted and biogenic material being produced, ILUC can cause net changes in GHG emissions or sequestration. Because leakage from additional biogenic feedstock production can potentially be significant, the framework in may need to consider including leakage for certain policy or program applications. Inclusion of leakage estimates would account for changes in GHG emissions from ILUC or other sources of leakage that occur outside of the biogenic feedstock production assessment scope.

# 3. Factors that Can Contribute to Leakage

In the general context of leakage related to GHG emissions globally, for example, as long as all emissions sources are governed by the same rules, shifting emissions from one region to another is perfectly acceptable and indeed represents a more economically efficient outcome (Murray, 2008). If all regions are covered by the same policy or assessment system, no leakage would occur, because all emissions would be accounted for (assuming full enforcement) (Murray, 2008). However, few policies have a global scope, making leakage difficult to avoid. Whenever incentives for action differ across potentially affected entities, there will be a tendency to shift activities that result in emissions from more highly controlled entities to less controlled entities. In general, leakage can erode net carbon reductions because "the spatial scale of intervention is inferior to the full scale of the targeted problem" (Wunder, 2008, p.65).

The primary driver of leakage is economic—in globally integrated markets, increased demand for a biogenic feedstock for energy within the assessment area may lead to increased production of that type of biomass and/or other changes in land use patterns outside the assessment boundaries. This is because increased demand for a biogenic feedstock for energy production triggers higher overall demand for the biogenic feedstock, thereby leading to higher commodity prices for that feedstock and its substitutes. These commodity price increases can lead to a succession of land-use changes to produce more feedstock, including the conversion of forest and other high-carbon storage ecosystems to lower carbon storage systems and the release of carbon stored in soils and vegetation. However, depending on the feedstock and time frame considered, it is also possible for positive leakage to occur. For instance, higher prices for forest biomass could lead landowners to convert a large enough area of agricultural lands to forests that regional carbon stocks are increased relative to baseline conditions.

Leakage effects, including ILUC, can also occur when lands and/or biogenic materials previously used for other purposes are instead diverted to biogenic feedstock production due to competition and resource scarcity. However, the market demand for the original product still exists and with higher commodity prices there is still an incentive for supply of the original product to approach the original quantity demanded. This additional demand can be met through intensification of existing lands producing the original product materials elsewhere or extensification, which means bringing new lands into production.

Agricultural and forest commodities are frequently traded in markets that operate at a local, regional, national, or global scale. As a result of this integration, changes in the supply and demand of commodities in one part of the world may be translated into changes in market supply and demand of the same and related commodities in other parts of the world. Policies targeting land use for specific activities in one location can induce a broader reallocation of land use unless such shifts are specifically and effectively restricted by the policy (e.g., Wu, 2000; Wear and Murray, 2004; Murray, McCarl, and Lee, 2004).

Similarly, substitutability and competition with other biomass types may lead to production changes beyond the assessment area because of potential product substitution (Latta et al., 2013). Land can be used to produce a wide array of forestry and agricultural products. Land cover and land use are expected to vary over time as land is allocated to activities that yield the highest net present value based on information available when the land use allocation decision is made. In addition, many forestry and agricultural commodities have other commodities that are at least partially substitutable for them (e.g., livestock feed can be made using a variety of grains and oilseeds, including corn, wheat, rye, barley, oats, soybeans, and others used in various combinations that meet livestock nutritional requirements). As a result, commodity prices are generally correlated due to adjustments taking place on both supply and demand sides as both buyers and sellers adjust to changing relative prices. Thus, there may be an associated emissions shift from assessed regions to unassessed regions due to land use change and other production-related activities. Ignoring leakage can make emissions fluxes from biomass use appear larger or smaller than they actually are, thereby potentially undercutting program objectives (Murray, 2008).

When these land-use transitions occur outside the assessment region, related GHG emissions fluxes may not be accounted for. Some of the literature indicates that biogenic feedstock production projects reduce GHG emissions to the atmosphere only if the net growth of harvesting of the biomass for energy captures carbon above and beyond what would be sequestered anyway (i.e., if sequestration is additional).[3] In one study, foregone sequestration is considered the equivalent of additional emissions, and when these emissions are associated with activities producing biomass,

[3] "Additionality" is a criterion for assessing whether an activity has resulted in GHG emission reductions or removals relative to what would have occurred in its absence. This is generally a more complex criteria for land-based mitigation activities than for point-source or facility-based activities because of the inherent dispersed, heterogeneous, dynamic, and systems-based aspects of agricultural and forestry production, but there are viable strategies for addressing additionality in these sectors (Janzen et al., 2012).

the author argues they should be included in GHG accounting associated with the biomass production (Searchinger, 2008).

"Leakage potential can be high if no counteracting provisions are put in place" (Murray, 2008, p. 10); and the economic forces driving leakage are interdependent and "difficult to restrain" (Murray, 2008, p. 18). According to some studies, when considering leakage, estimates should reflect the following elements: connectedness of output and land markets, mobility of labor and capital, consumer flexibility, producer flexibility, availability of alternative lands for production, and ability of producers to change their emissions profile without modifying production (Wunder, 2008; Henders and Ostwald, 2012).

The change in total cultivated land associated with a change in demand for bioenergy feedstocks will depend on a number of parameters, but one of the most important factors is the land supply elasticity. In cases where land supply is relatively elastic, there will be relatively large increases in supply of a given land type when the returns to that type of land increase. Landowners will shift their land cover and crop mixes to provide more of the commodities that are in greater demand. If land supply is relatively inelastic, on the other hand, then there will be a smaller response to changes in demand for individual commodities. The more competitive and integrated land markets are across regions, the larger the extent of leakage expected when different regions face different incentives to mitigate emissions. However, when considering leakage across regions, even when considering only a single type of biomass, it should be noted that there is a difference between shifts in production activity or area and net emissions. For example, even within the United States, shifting forestry area from the Southeast (SE) to Pacific Northwest (PNW) would likely reduce net carbon emissions, whereas the reverse would result in significant positive domestic leakage due to the lower carbon density of SE forests (summing across carbon contained in aboveground biomass, belowground biomass, deadwood, forest floor, and soil organic carbon) (Heath et al., 2011).

The elasticity of demand for conventional commodities must also be considered. In cases where demand is inelastic, the quantity demanded will change by a smaller percentage than prices rise. This inelastic behavior will result in a greater amount of leakage than in markets with more elastic demand. Higher production costs resulting in lower production levels in regulated regions will result in a great deal of shifting of production to regions unaffected or less affected by policies because the overall market demand does not decline much in response to higher prices. When demand is highly elastic, policy impacts that result in higher production costs and increased market prices will result in less production moving to other regions because the equilibrium quantity demanded will decline by a greater percentage than price increases. In addition to the own-price elasticity of demand, cross-price elasticities of demand for substitutes and complements are also important to consider. Not only will increases in the market prices of directly affected commodities potentially lead to increased production of those commodities into less directly impacted regions, but they will also impact production of complement and substitute commodities in other regions. Another important point of consideration is that as demand for a commodity increases, producers may intensify production practices (e.g., increase fertilization rates, use of irrigation, improved crop varieties, and other yield enhancements) because higher output prices make it profitable to engage in more intensive production practices requiring greater input expenditures. Achieving higher yields through intensification would limit direct and indirect land-use change (Searchinger, 2008) but may lead to other GHG fluxes (e.g., increased $N_2O$ emissions from higher levels of nitrogen fertilization, higher $CO_2$ associated with greater fossil fuel use for irrigation). Thus, the net change in GHG emissions would depend on the relative changes in emissions across all relevant pools and intensification could either increase or decrease total emissions relative to extensification.

# 4. Overview of Relevant Literature

Although the concept of carbon emissions leakage in industrial sectors has been widely studied for over 20 years, the focus on leakage in land-using sectors has been more recent. There have been many studies of industrial carbon leakage ever since the United Nations Framework Convention on Climate Change was established in 1992, identifying differentiated responsibilities for reducing emissions, and the subsequent Kyoto Protocol was developed and agreed upon with emissions limits specified only for a set of developed countries. There is also extensive literature on the international trade and competitiveness under environmental policy going back to the 1980s, although this literature on "pollution havens" was not typically focused on global pollutants and often not explicitly focused on implications for total emissions as much as distributional impacts of polluting industries' potential relocation between states or countries.

Interest in leakage associated with land-using sectors has grown considerably in the last decade with the development and implementation of bioenergy policies and international policy interest in reducing emissions from deforestation and land degradation (REDD+). In this section, an overview of the literature on leakage associated with land use and potential relevance is provided, followed by brief sections summarizing some of the recent relevant literature focused on agriculture and forestry applications.

## 4.1. Literature Research on Leakage

A full accounting for leakage associated with land use and related GHGs is very complex because of the multiple affected markets, heterogeneity, dynamics, and numerous interactions. The literature offers an incomplete picture of leakage magnitude and what can be done to minimize negative leakage (Kim and Dale, 2011; Murray, 2008). Furthermore, the precise meaning of the term "leakage," both in terms of scale and scope, can fluctuate from study to study, making direct comparisons difficult.[4] Finally, few studies mirror the feedstock sub-delineations used in the framework report, thereby complicating evaluations of feedstock-specific applicability.

Because the primary bioenergy, REDD, and other forestry and agriculture policies of interest for leakage assessment have typically been implemented relatively recently, time series data for empirical analysis are limited. Typically, the policies being considered do not have direct historical precedent and would result in new markets being created, which results in changes in market and

[4] "Leakage" sometimes refers simply to indirect land-use change but can also be used along with carbon debt or market price impacts. If GHG emissions from all regions are accounted for in a consistent manner and reflected under a regulatory framework, then there could be indirect land-use change without carbon leakage (because any changes in emissions associated with indirect land-use change have been reflected in GHG accounting).

land use activities that fall outside past experience and limits the ability of statistical analyses of existing data to explain future outcomes. Data limitations, along with the complexity of adequately reflecting relevant factors influencing market outcomes and land dynamics, have resulted in there being a limited empirical literature on leakage in land-based sectors. Absent empirical data from representative case studies, leakage estimates have instead employed a variety of economic land use models covering the agricultural, forestry and other land use sectors, such as, the Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOM-GHG), the Food and Agricultural Policy Research Institute (FAPRI) model, and the Global Trade Analysis Project (GTAP) general equilibrium model. These economic models vary widely in terms of model type, inputs, and assumptions, as well as scope and scale in terms of output. In addition, many of the existing economic land use models do not fully account for all GHG emissions associated with the market activities being modeled. Therefore, in some applications, the changes in market activities and land use simulated using the models have been combined with emissions factors available in models such as the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model in order to estimate net changes in GHG emissions.

Such models can lend insights as to the possible directionality and/or general magnitude of leakage. However, leakage affects multiple markets and regions simultaneously, thereby increasing the complexity of model projections and making it difficult to isolate the causes and effects of market and land-use shifts and related leakage. The results are largely case-specific, and depend greatly on context and the assumptions of each particular study.

In countries where land use is highly regulated and controlled with few linkages to international markets, it may be easier to estimate (and control) leakage. When land use change occurs in an unplanned and unpredictable manner across numerous countries (e.g., indirect land-use change caused by bioenergy development), detecting leakage is particularly difficult and involves many different measurements and analyses to adequately represent and understand the land use and emission dynamics. Many studies project multiple scenarios using many different assumptions as sensitivity analyses to reflect parameter uncertainty. Because of this and the aforementioned variability in how leakage is defined and measured, there is considerable inconsistency surrounding leakage estimates (e.g., Plevin et al., 2010).

## 4.2. Leakage Literature: Agriculture

There has been a growing amount of attention and research effort devoted to the effects of policies affecting the demand for agricultural commodities on land use change, particularly in the context of increased demand for biofuels (e.g., ethanol and biomass-based diesel). Notably, EPA includes estimates of land use change due to increased demand of specific biofuels (e.g., corn ethanol) as part of the GHG accounting applied for Renewable Fuel Standard (RFS2) feedstock analyses (EPA, 2010), as described in more detail in Section 5.1.1. Partly because of the implementation of RFS2 and biofuels policies in the European Union, Brazil, and elsewhere in recent years, there has been a great deal of interest in indirect land use change and leakage associated with bioenergy policies.

The existing literature assessing potential leakage magnitude from corn ethanol production shows that estimates differ considerably across studies and within a study depending on underlying assumptions (Khanna et al., 2011; Khanna and Crago, 2012). Searchinger et al. (2008, 2009) and Fargione et al. (2008) are frequently cited studies among the first to challenge the net benefits associated with biofuels on the basis that indirect land use and GHG emissions from land use conversion were not being fully captured and to quantify the impacts. For instance, Searchinger et al. (2008) combines calculated changes in land use they estimate are necessary to meet required increases in U.S. corn ethanol volumes with emission factors from the GREET lifecycle model to project changes in net GHG emissions resulting from an increase in U.S. corn ethanol production. Searchinger et al. (2008) calculated that a 56 billion liter increase in ethanol production would divert corn from 12.8 million ha (hectares) of U.S. cropland. This would in turn bring 10.8 million ha of additional land globally into cultivation, including 2.8 million ha in Brazil, 2.3 million ha in China and India, and 2.2 million ha in the United States. The emissions associated with converting this land represent leakage (aside from direct land use change for corn production in the United States), but the true magnitude of the leakage depends on the nature of the land-use change (i.e., what type of land is being converted). They assumed the conversion of forest to cropland releases 604 to 1,146 metric tons $CO_2$e per ha, while the conversion of grassland/savannah to cropland releases 75 to 305 metric tons $CO_2$e per ha.

Regardless of the type of land converted, the payback period for these up-front emissions can be very long. Using an average of 351 metric tons $CO_2$e per converted ha, Searchinger et al. (2008) estimated "carbon neutrality" for corn-based ethanol only after 167 years. Using the standard 30-year time frame, corn-based ethanol results in a 93% increase in net emissions compared with business-as-usual gasoline consumption. This study also ran a sensitivity analysis that includes 20% increases in grain yields, land-use emissions reduced by half, and process efficiency gains of 40%. In that best-case scenario, net emissions approach those of gasoline after 30 years.

A key driver of the results in each of these studies is that expanded corn ethanol production in the United States will substantially reduce U.S. corn exports, leading to expanded foreign corn production. They assumed that foreign countries will convert natural vegetation to croplands, including tropical rainforests and other high carbon density areas, which results in very large carbon emissions (Searchinger et al., 2008, 2009; Fargione et al., 2008).

However, a number of subsequent studies have critiqued some of the assumptions used in Searchinger et al. (2008) (e.g., Wang and Haq, 2008) and many subsequent studies provide alternative estimates of ILUC and leakage. Some of the key considerations influencing ILUC include assumptions regarding price elasticities, yield improvements over time, the substitution rate of dried distillers grains for corn, the types of land that are converted, GHG calculations, and long-term land dynamics in response to changing incentives, among others.

For instance, Hertel et al. (2010) employed an approach similar to Searchinger (2008), but used different assumptions about the land that will be converted to agriculture. Importantly, Hertel et al. (2010) relied on the GTAP-BIO version of the GTAP model, a general equilibrium agricultural economic model that provides more land use data and economic details of land use than the more basic assumption about land requirements used by Searchinger (2008). Whereas Searchinger (2008) estimated most conversion will be either forest to cropland or grassland to cropland, Hertel et al. (2010) estimated that much of the cropland transition will actually stem from degraded, pasture, or other low-carbon lands, which substantially reduces emissions from indirect land use change. Studies such as Potter et al. (2007) and Aguilar et al. (2012) have estimated significant quantities of marginal lands in the United States that they argue could be converted to biomass production, which would substantially reduce pressure for conversion of higher carbon density forests and grasslands. However, it is also important to consider the economic incentives facing landowners. The fact that it is physically possible to grow bioenergy crops on marginal lands does not mean landowners will choose to do so if it is more profitable to grow them on more productive croplands.

Drabik and de Gorter (2011) also measured carbon leakage associated with biofuels (specifically, corn-based ethanol), but instead of land-use change they focused on the market response to corresponding changes in fuel prices. Specifically, if fuel prices decline as a result of increased ethanol production, then total fuel consumption will presumably increase (i.e., the rebound effect). As a result, 1 gallon of ethanol does not replace 1 gallon of gasoline. Instead, because of the price impact, Drabik and de Gorter (2011) showed 1 gallon of ethanol replacing only 0.35 to 0.5 gallons of gasoline. Thus, although average per gallon carbon intensity of fuel may decrease as a result of ethanol, overall fuel consumption may increase and could potentially overwhelm these reductions.

In another application examining potential domestic leakage of environmental policy, Wu (2000) analyzed domestic leakage related to the U.S. Conservation Reserve Program (CRP) and found that for every 100 acres of cropland retired under CRP in the central United States, about 20 acres of non-cropland were converted to cropland. This leakage effect was estimated to offset about 9% of the CRP water erosion benefits and 14% of the wind erosion benefits.

### 4.3. Leakage Literature: Forestry Sector

Similar to the agricultural sector, policies affecting forest use in one place are expected to impact forest management decisions elsewhere. In particular, there has been a great deal of interest in quantifying the extent to which forest conversation measures in one location induce greater timber harvesting elsewhere. Many carbon leakage studies have focused on the forestry sector.

Murray, McCarl, and Lee (2004) developed a conceptual model for analyzing market adjustments and carbon leakage. They also applied FASOM to empirically estimate leakage from different forest preservation strategies in the United States. They estimated leakage for U.S. carbon sequestration policies and found leakage rates varying from less than 10 to over 90% depending on policy specifications and region(s) of the country where the policy is implemented. Alig et al. (1997) also applied FASOM and found that carbon benefits from expanded U.S. afforestation would largely be offset by converting existing forestland to agriculture. Wear and Murray (2004) explored the effects of public forest conservation in the U.S. Pacific Northwest on forest production and markets in the United States and Canada. They found that a total of about 84% of reduced public harvest would be replaced by increased private harvest, and around 58% of reduced public harvest would be replaced within the United States and another 26% in Canada.

Sohngen and Brown (2004) examined leakage associated with a specific tropical forest conservation project in Bolivia. They developed a dynamic timber market optimization model and ran the model using differing assumptions about global policies, capital constraints, demand elasticity, and deadwood decomposition rates. Overall, they found leakage rates of 5 to 42% for this project-level assessment. Leakage is lowest when demand is more elastic and wood decomposition rates are faster.

There have also been a number of recent studies examining leakage associated with forestry policies at the global level. Gan and McCarl (2007) estimated international leakage with the GTAP general equilibrium model. They defined leakage in terms of forest production rather than emissions, but the changes in forest production from their model would clearly have implications for GHG emissions. They examined lumber, paper, and log markets and analytically derived the transnational leakage and applied GTAP to estimate leakage at a global level. The study concludes that leakage is related to assumptions about the elasticities of demand and supply of forestry products, lumber and wood products, and pulp and paper products across many countries. They also note that cooperation among countries tends to alleviate leakage. Under current global trade conditions, they estimated leakage rates ranging from 42 to 95% with leakage rates above 70% for the majority of regions. Because they are defining leakage in terms of forest products production, carbon leakage may be even greater if forest production is shifting to less efficient production regions and/or regions with higher carbon density. Sun and Sohngen (2009) use a global land-use forestry model to estimate global leakage emanating from three different forestry set-aside scenarios and find that leakage could be nearly 100% in the near term under one of the global policies examined. Sohngen and Sedjo (2000) showed the impact of increased demand on harvests and management of industrial forests in regions around the globe. Their model showed significant GHG emissions from boreal and temperate forests, but this impact was dampened by the rising influence of subtropical plantations. Overall, they found carbon leakage to be less than 16%.

In addition, several recent studies have identified the potential for increased bioenergy demand to raise biomass prices sufficiently to induce greater levels of afforestation and more intensive forest management to the extent that total carbon stocks are actually increased (Daigneault, Sohngen, and Sedjo, 2012; Sedjo and Tian, 2012; Sedjo, Sohngen, and Riddle, 2013). There may be a short-term reduction in forest carbon as harvesting increases but greater sequestration in the long term as forest area and carbon densities increase.

# 5. Examples of Leakage Analysis

Although the development and implementation of a land use analysis that adequately reflects leakage is a very complex endeavor, there are cases where leakage has been estimated and used in calculations of net GHG emissions profiles for both policy analyses and carbon accounting protocols. Several examples are presented below.

## 5.1. Policy Analyses that Include Leakage

There are a number of examples of policies and programs that take leakage into account. However, the methodological approach of each program is carefully tailored to meet the program or policy's requirements that the analysis is being designed to serve. As such, these methodologies may differ in several ways, including technical assumptions used, models (and types of models) used, scope (spatial and temporal), and many other factors.

### 5.1.1. Renewable Fuel Standard

The Energy Independence and Security Act of 2007 (EISA) specifies life-cycle GHG emissions reductions thresholds that renewable fuels must meet to qualify in different categories and defines lifecycle GHG emissions to include “significant indirect emissions such as significant emissions from land use change.” As a result, EPA’s analysis was conducted to capture emissions that may result from indirect land use changes in the United States and abroad. EPA’s analysis of the RFS2 program was conducted estimating the effects of shocks of national aggregate demand for individual feedstocks. Partial equilibrium models of the global agricultural sector (FAPRI) and the domestic forest and agricultural sectors (FASOM-GHG) were used to simulate the effects of expanded bioenergy production consistent with RFS2 requirements on land and commodity markets. The changes in market activities and land use generated using these models were combined with emissions factors from the GREET model, satellite data analysis of modeling of land use change emissions (Harris et al., 2008), and IPCC emissions factors to generate estimates of lifecycle emissions associated with renewable fuels production.

Both direct and indirect land use changes are included in the calculation of the net lifecycle GHG reductions provided by individual biofuels feedstocks, including land use adjustments within the U.S. and internationally. Total net changes in activities are presented in that study; there is no separation of direct and indirect impacts. For a biofuel pathway to qualify under a given RFS2 category, it must meet or exceed the GHG reduction threshold for that category based on the total net emissions associated with that pathway relative to the use of fossil fuels. Renewable fuels have a minimum target of 20% reduction; advanced fuels, including biomass-based diesel, must provide at least a 50% reduction; and cellulosic fuels must reduce emissions by at least 60%.

It is also important to note that the inclusion of leakage in the RFS2 analysis is the result of fulfilling statutory obligations as set forth in the EISA of 2007. The goals, methodology, tools, and assumptions used for EPA’s RFS2 analysis may not necessarily be suited for another policy analysis. Whether and how to reflect leakage in the context of a specific policy or program application of the biogenic assessment framework would need to be determined for particular applications. Each analysis must design a framework that best suits their particular goals and analytical requirements. For more information on EPA’s RFS2 final rulemaking and analysis, refer to the Regulatory Impact Analysis (EPA, 2010).

### 5.1.2. CARB Analysis

The intent of the low carbon fuel standard (LFCS) implemented in California is to reduce the GHG emissions intensity of fuels used in the state using a performance-based standard. In its rulemaking implementing the LCFS, the California Air Resources Board (CARB) noted that incomplete policy coverage could result in little change in emissions at the global level (CARB, 2009). The LCFS implemented by CARB attempts to control leakage by adopting very similar language on including indirect emissions from EISA (OAL, 2010), which should capture indirect emissions within fuel emissions intensity estimates. Under this policy, there is a GHG intensity target developed for transportation fuels based on this lifecycle assessment. Regulated parties are the transportation energy suppliers, who are allowed to trade credits, providing incentives for using fuels with lower net carbon emissions and stimulating investment in continued development of low-carbon fuels.

CARB has worked extensively with the GTAP model to assess international land use emissions within a global framework (e.g., Tyner, 2011). Changes in land use and activities estimated using this model are combined with emissions factors obtained from the GREET model to generate estimates of the net changes in GHG emissions associated with production of individual fuels. To calculate the carbon in baseline fossil fuels across their life cycle, CARB uses the Oil Production and Greenhouse Gas Estimator to calculate a value for Annual Crude Average Carbon Intensity.

### 5.1.3. EU RED Analysis

The EU Renewable Energy Directive (RED) mandates that 20% of all energy usage in the EU, including at least 10% of all energy in road transport fuels, must be produced from renewable sources by 2020. In addition, an amended fuel quality directive was implemented requiring that the road transport fuel mix in the EU should be at least 6% less carbon intensive than the diesel and gasoline baseline by 2020. The EU RED also states that under national biofuel support systems, "the contribution made by biofuels produced from wastes, residues, non-food cellulosic material, and ligno-cellulosic material shall be considered to be twice that made by other biofuels." This policy also specifies sustainability criteria, whereby biofuels must achieve a minimum reduction in GHG of 35% relative to fossil fuels in order to be eligible for support under EU renewable energy policies. Beginning January 1, 2017, the threshold rises to 50% reduction in GHG. Beginning January 1, 2018, any facilities starting production on or after January 1, 2017, must meet a minimum GHG reduction of 60%.

A lifecycle methodology is defined to calculate emissions from biofuels production for the purposes of calculating the net GHG reductions. The European Commission has provided default emissions factors for each biofuel production pathway that regulated entities can use for their calculations and reporting. Regulated entities also have an option to provide information about their specific production processes in order to calculate emissions that are specific to their process for use in place of the default values. The EU RED does not currently account for indirect land use change, though it does restrict production of biofuels on land that had high biodiversity status or high carbon content at any point on or after January 2008. Information about biofuel sustainability must be tracked using a mass balance chain of custody system. There have been recent proposals and considerable debate about adding specific indirect land use change emissions factors to this policy, but agreement has not yet been reached.

## 5.2. Treatment of Leakage in Existing Carbon Accounting Protocols

Several organizations have developed carbon accounting protocols for companies and entities looking to measure the impact of carbon reduction (i.e., offset) projects. In creating these protocols, developers have devised methods to incorporate leakage factors into their methodologies and are pragmatic attempts at best practices for use with existing carbon reduction projects. As with the literature discussion in Section 5.1, it is often difficult to determine the precise definition of leakage used in each protocol and there are not necessarily mechanisms to adjust carbon credits on the basis of leakage.

Galik, Mobley, and deB. Richter (2009) assessed seven different protocols along a number of dimensions, including leakage. Those protocols included the U.S. Department of Energy 1605(b) Technical Guidelines for Voluntary Reporting of Greenhouse Gases (Office of Policy and International Affairs, 2007); Georgia Forestry Commission (GFC) Carbon Sequestration Registry Project Protocol (Georgia Forestry Commission, 2007); Chicago Climate Exchange (CCX) Sustainably Managed Forests/Long-Lived Wood Products Protocols (Chicago Climate Exchange, 2007a, 2007b); California Climate Action Reserve (CAR) Forest Project Protocol (CAR, 2010); Voluntary Carbon Standard (VCS) Improved Forest Management Protocol (VCS, 2007a, 2007b); a protocol based on recommended concepts in Duke University's *Harnessing Farms and Forests in the Low-Carbon Economy* (HFF) publication; and a draft recommendation for active forest management offset projects proposed by the Maine Forest Service and others under RGGI (Maine Forest Service et al., 2008). They found that only the VCS and HFF accounting standards had quantified mechanisms for accounting for leakage at the time they conducted the assessment. Both VCS and HFF included all forest carbon pools assessed by Galik, Mobley, and deB. Richter (2009) and VCS generated values for leakage between 10 to 40% (with a base case of 10%) while HFF included leakage of 33.5 to 44.5% (base case of 43%).

In addition, there are accounting procedures developed by the UNFCCC Clean Development Mechanism (CDM). The CDM methodology for simplified baseline and monitoring methodologies for small scale biomass project activities (UNFCCC, 2014) identifies three potentially significant sources of emissions (>10 % of project emissions reductions) that are attributable to the project:

- Shifts of pre-project activities, including decreases in carbon stocks outside the area where the biomass is grown due to shifts in pre-project activities;
- Emissions related to the production of the biomass; and
- Competing uses for the biomass.

Those emissions may be considered project emissions if they arise from lands under the control of the project owners or sources of leakage.

CDM guidance suggests that shifts in pre-project activities are relevant where the lands would be used for other purposes (e.g., agricultural production) in the absence of the project. In cases where the land would not be used or where land use inside the project boundary does not change as a result of the project, the guidance is that leakage does not generally need to be included. That applies to extraction of biomass from existing forests, cultivation of biomass on abandoned lands, and for biomass residues or wastes because they assume the use of the residue or waste is unlikely to affect the generation of the residue or waste. For other types of biomass, the CDM guidance is to evaluate the potential displacement of activities or people using the following indicators:

- Percentage of families/households of the community involved in or affected by the project activity displaced (from within to outside of the project boundary) due to the project; and

- Percentage of total production of the main product (e.g., corn, beef) within the project boundary displaced due to the generation of renewable biomass.

If the values of these two indictors are both less than 10%, then leakage is assumed to be 0%. If the value of either indicator is >10% but <50%, then leakage is assumed to be equal to 15% of the difference between baseline and project emissions. If the value of either is >50%, then this simplified methodology is not applicable and a new procedure must be submitted for approval.

In terms of emissions from the production of biomass, the two categories of emission included are emissions from fertilizer application and project emissions from land clearing. It is assumed that all other emissions sources are likely to be smaller than 10% individually and therefore do not need to be included. The guidance suggests that land use change other than deforestation does not need to be included and the guidance indicates that the project developers should demonstrate the area where the biomass is grown is not a forest and has not been deforested within the last 10 years.

For competing uses of biomass, the guidance suggests evaluating if there is a surplus of biomass in the region of the project activity that is not currently utilized. If it is demonstrated (e.g., based on published literature, official reports) at the beginning of each crediting period that the quantity of available biomass in the region (e.g., 50 km radius) is at least 25% greater than the quantity of biomass utilized including the project activity, then this source of leakage can be assumed to be 0%. Otherwise, leakage should be estimated and deducted from project emissions reductions,

The recently updated Climate Action Reserve (CAR) protocol (Version 3.2.) uses default leakage factors to account for changes in activities outside the project boundary. They define a decision tree for project developers to use to determine the appropriate leakage factor. A standard discount of 24% is used for cropland converted to forest (CAR, 2010; Henders and Ostwald, 2012). For other land uses, leakage is defined as 0% for improved forest management projects on actively managed forestland for projects that increase harvesting. Improved forest management projects that result in reduced harvesting relative to the baseline are assumed to have a leakage rate of 20% of the difference in harvest volume. When land had been actively grazed, the leakage factor ranges from 10 to 50% as expected canopy cover under the project increases once canopy cover reaches 30% (canopy cover less than 30% is assumed to have 0% leakage).

The updated VCS approach (Version 3.4, VCS, 2013) states that the potential for leakage should be identified and that projects are encouraged to include leakage management zones as part of the project design. Leakage management zones should be used to minimize the displacement of land use activities outside the project area by maintaining the production of goods and services within areas under control of the project proponent or by addressing socio-economic factors that drive land use change. Activities to mitigate leakage and reduce deforestation and/or forest or wetland degradation are encouraged.

In calculating leakage, specific carbon pools and GHG sources do not have to be accounted for if the omitted decrease in carbon stocks or increase in GHG emissions amounts to less than 5% of the project GHG reduction. Peer-reviewed literature or the CDM afforestation/reforestation methodological tools may be used to determine whether changes in carbon stocks and emission meet this *de minimus* level. In addition, there are specific sources defined as *de minimus* (e.g., GHG emissions from removal or burning of vegetation and collection of non-renewable wood sources for fencing off the project area). The protocol also calls for methodologies used for project accounting to adjust emissions for all significant sources of leakage using verifiable assumptions. VCS requires accounting for market leakage (production shifting elsewhere to make up for reduced supply), activity-shifting leakage (agent of deforestation or degradation moves to an area outside the project boundary and continues the deforestation or degradation activities), and ecological leakage (project causes changes in GHG emissions or fluxes of GHG emissions from ecosystems that are hydrologically connected to the project area). International leakage does not need to be quantified under this protocol. In addition, projects cannot include positive leakage where net GHG emissions outside the project area are reduced.

# 6. Summary

The manner in which leakage is calculated or incorporated for a particular policy, program, or study will be highly dependent on the analytical requirements of the project, the assessment scope, the feedstock(s) under consideration, and the methodology developed to carry out such an analysis. A national or global analysis of changes in feedstock demand and related commodity market and land use activities could generate estimates of the potential directionality and magnitude of leakage effects due to changes in biogenic feedstock use. However, application of the framework in this way may not be required for certain U.S. domestic policy analyses. Therefore, because this framework is intended to be policy neutral, it does not prescribe a particular leakage estimation method. For any potential application of the framework that aims to incorporate impacts from leakage, many important factors must be considered, as discussed in this appendix and shown in the examples above.

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# Appendix F: General Algebraic Representation of the Biogenic Assessment Factor Equations

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# 1. Introduction

The equations and conceptual diagrams in this appendix are intended to illustrate the various carbon flows that contribute to the calculation of net atmospheric contribution of biogenic $CO_2$ emissions associated with the production, processing, and use of biogenic material at a stationary source *(NBE)* and biogenic assessment factor (*BAF*) values for stationary source biogenic feedstock consumption. This appendix builds on the primary *NBE* equation provided in the main report to develop a generic algebraic formulation describing how the net atmospheric contribution of biogenic $CO_2$ emissions for a stationary source could be calculated at different points of assessment, depending on the specific policy or programmatic context, and to provide simple concrete examples of how the generic equation could be applied.

# 2. Simple Algebraic Representation

As discussed throughout this report, the Net Biogenic Emissions *(NBE)* from stationary source biomass consumption equation can be presented as the following equation:

$$\boldsymbol{NBE} = (\boldsymbol{PGE})(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P}) = (\boldsymbol{PGE})(\boldsymbol{BAF}) \quad \textbf{(EQ. F.1)}$$

Where the *BAF* is given by Equation F.2:

$$\boldsymbol{BAF} = \frac{\boldsymbol{NBE}}{\boldsymbol{PGE}} = (\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P}) \qquad \textbf{(EQ. F.2)}$$

The equations above are designed to transform a measurable or estimated quantity (the carbon content of the biogenic feedstock used at the point of assessment [potential gross emissions or *PGE*]) into a quantity that cannot be directly measured (the net atmospheric biogenic $CO_2$ contributions associated with different stages of biogenic feedstock production, processing, and use at a stationary source [*NBE*]). The terms in the *NBE* equation each play a specific role in this transformation:

- *PGE* is the carbon content of the biogenic feedstock used by a specific entity or generally consumed). This is a quantity that could be measured or estimated at different points of assessment (e.g., at the boiler mouth, stationary source gate, feedstock production site, or at the stack: wherever the point of assessment needs to be. Thus, this term can have different values indicated by subscripts, representing different points along the supply chain).
- *L* is a unitless adjustment factor greater than or equal to one that represents biogenic feedstock carbon that leaves the supply chain (e.g., via transit or decomposition, deviated for use as a product) between the feedstock production site and input into the conversion process at a stationary source. *L* scales *PGE*, as it was measured at the point of assessment, up to account for any losses during transportation or storage between the feedstock production site and the point of assessment. *PGE* times *L* is thus the carbon content of the biomass that was grown at the feedstock production site in order to deliver the quantity of feedstock measured at the point of assessment.
- *P* is a unitless adjustment factor between zero and one, equal to the share of the carbon content of the feedstock at the point of assessment that is emitted to the atmosphere by a stationary source (versus that which is embedded in products). In effect, this term also reflects the share of carbon that remains in products, that is either not emitted to the atmosphere or is sold and eventually emitted to the atmosphere by a downstream user.
- *(GROW + AVOIDEMIT + SITETNC + LEAK)* represents the landscape emissions effect. This landscape emissions effect is the sum of four unitless factors that relate the total biogenic carbon content of the feedstock grown at the feedstock production site, i.e. *(PGE)*(L)*, to related landscape biogenic carbon pools. The details of these terms are discussed elsewhere in this framework. For the purposes of this appendix, we can think of the terms *(GROW + AVOIDEMIT + SITETNC + LEAK)*(PGE)*(L)* as the estimated net contribution to the atmosphere associated with growing, harvesting, producing, processing, and using the feedstock that was measured at the point of assessment. This amount is then multiplied by *P* to determine the share that is actually emitted and is the responsibility of a particular entity.

The framework itself is designed to be flexible enough such that it can be applied to a variety of programs with different requirements. For example, *PGE* in the *NBE* equation above is the potential gross emissions at the point of assessment for the purposes of applying the biogenic assessment factor. Depending on how the framework is applied, this point of assessment could be interpreted as biogenic emissions at the boiler mouth, the stack, or total potential biogenic emissions at the farm gate once biomass has been harvested. Additionally, losses can occur and products can be produced at different points along the supply chain.

To understand how the *NBE* equation adapts to different points of assessment and accounts for products and losses at different points along the supply chain, we need to understand how the *L* and *P* terms are calculated and what exactly they represent. In order to build this understanding, this appendix begins with a simple example and then shows how the generic version of the equation is able to capture more complexity in subsequent examples.

## 2.1. Example 1: Simple Carbon Flow with Point of Assessment at the Boiler/Fermenter Mouth

Consider the following conceptual example provided by Figure F-1. This flow diagram shows the evolution of *PGE* along the production supply chain. This representation uses atmospheric accounting methods, so terms that represent an emission (or potential emission) to the atmosphere (e.g., *LOSS, PGE*) are positive, and terms that represent sequestration (e.g., *PROD*) are negative.

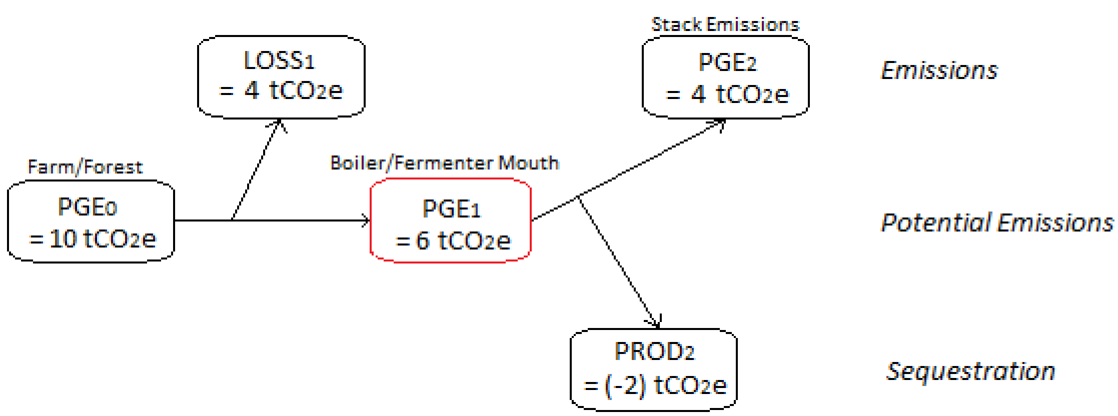


**Figure F-1: Conceptual Diagram Illustrating the Calculation of Potential Gross Emissions at Different Points of Assessment.**

The first thing to note in this diagram is that it contains different variables from the variables in Equations F.1 and F.2 such as *LOSS* and *PROD*. The main equations discussed above are designed to be applied with limited data requirements. The example here is designed as a thought experiment to "follow the tons of biogenic $CO_2$," and we show how these quantities can be used to calculate the variables that go into the *BAF* equation. Another difference is that the *PGE* variables gain subscripts to represent the different points of assessment at which *PGE* could be measured (e.g., at the forest/farm, at the boiler mouth, out the stack). This can be generalized to $PGE_i$ in order cover any number of measurement points, where *i* indexes over all the points of measurement. The red circle indicates where the point of assessment is in this example, so here $PGE_1$ would be equal to *PGE* from Equations F.1 and F.2 as applied at the boiler/fermenter mouth.

The terms *LOSS* and *PROD* represent the actual tons of carbon lost in transportation or storage along the supply chain (*LOSS*) or stored in final products or by-products (*PROD*) (rather than emitted during conversion). The subscripts indicate at which stage in the supply chain each occurs, e.g., $LOSS_1$ occurs before $PGE_1$, and $PROD_2$ occurs before $PGE_2$. These subscripts will become necessary when we move to more complex supply chain examples and generalize the equations. In the schematic of this example (Figure F.1), the top row represents actual gross emissions to the atmosphere. Here losses are assumed to generate actual emissions, and $PGE_2$ represents actual emissions out the stack (note that at this point the emissions are no longer "potential," but for notational reasons we still refer to these stack emissions as *PGE*). The middle row is for potential emissions at various assessment points. These values represent tons of biomass moving through the supply chain that have not yet been emitted or sequestered or contained in products. The bottom line of Figure F.1 is tons that go into products or by-products and are either not emitted to the atmosphere or not emitted by this entity. Note that because we are using atmospheric accounting, emissions and potential gross emissions have a positive value, and carbon contained in products or by-products is assigned a negative value.

- $PGE_0$: Represents *PGE* at the forest/farm gate. This is the total harvested biogenic $CO_2$ with the potential to be emitted at the forest/farm gate and includes all biomass that is harvested and transported from the forest or farm to the stationary source facility.
- $LOSS_1$: Represents the biogenic $CO_2$ lost in transportation or storage between the forest/farm gate ($PGE_0$) and the boiler/fermenter mouth ($PGE_1$).
- $PGE_1$: Represents *PGE* at the boiler/fermenter mouth.
- $PROD_2$: Represents biogenic $CO_2$ stored in long-term product pools (including lumber, ethanol, or other purely marketable products produced with a portion of the harvested biomass) or other nonmarketable industrial by-products (including ash).
- $PGE_2$: Represents emissions at the stack.

Thus, the equations relating these variables are:

$$\boldsymbol{PGE_1 = PGE_0 - LOSS_1 = 10 - 4 = 6\ tCO_2e.} \qquad \textbf{(EQ. F.3)}$$

$$\boldsymbol{PGE_2 = PGE_1 + PROD_2 = 6 + (-2) = 4\ tCO_2e.} \qquad \textbf{(EQ. F.4)}$$

Now let us define how the terms in the *NBE* equation *(GROW + AVOIDEMIT + SITETNC + LEAK), (L),* and *(P)* are calculated and apply our example values. Note that in the equations below we adopt the notation convention that *PGE* at the point of assessment ($PGE_1$ in this example, or simply *PGE* in Equations F.1 and F.2) is written as $PGE_j$, where the subscript "*j*" represents the point of assessment (in this example, $j$=1, the boiler/fermenter mouth).

The landscape emissions effects terms (*GROW, AVOIDEMIT, SITETNC, and LEAK*) are discussed extensively elsewhere in the framework, including detailed discussions of how they can be calculated in practice. It is important to note that *GROW, AVOIDEMIT, SITETNC*, and *LEAK* are all unitless.[1] For the purposes of this appendix, we can define our landscape emissions effects terms from the overall *BAF* equation as follows:

[1] However, the framework can be adapted to use units instead of unitless values as needed for a specific application.

$$(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK}) = \left(\frac{\boldsymbol{G+A+S+Lk}}{\boldsymbol{PGE_0}}\right). \qquad \textbf{(EQ. F.5)}$$

This is merely another way to specify the landscape effect. The original term is a unitless ratio of the contribution of landscape effects on the overall *BAF* value. However, we can also think of each of these elements as being relative to the total amount of biomass that is harvested at the forest or farm ($PGE_0$). *G, A, S,* and *Lk* are variables that represent the actual tons of landscape net emissions resulting from producing $PGE_0$ tons of biomass.[2] In that sense, *(G+A+S+Lk)* represents actual net emissions on the landscape caused by a harvest of $PGE_0$. The landscape-level emissions are normalized by $PGE_0$ to arrive at the original, unitless term.

Now let's consider the terms *L* and *P* from the *NBE* equations. *L* accounts for transportation or storage losses, relating the carbon content of biomass used by the facility at the boiler fermenter mouth ($PGE_1$) to the carbon content of that biomass when it was grown and harvested at the forest/farm ($PGE_0$).

*L* is defined as the ratio of harvested potential biogenic emissions ($PGE_0$) and *PGE* evaluated at assessment point *j*.

$$\boldsymbol{L} = \left(\frac{\boldsymbol{PGE_0}}{\boldsymbol{PGE_j}}\right) \qquad \textbf{(EQ. F.6)}$$

Note that this fraction will result in a positive number greater than or equal to one. Also note that although *LOSS* does not appear in this equation, we could substitute in from equation 0 to express *L* in terms of just one of the *PGE* variables and the *LOSS* term.

The purpose of *L* here is twofold. First, it represents the transformation of $PGE_j$ to $PGE_0$, implicitly capturing any losses and products[3] that occur between the point of assessment and the forest/farm. Furthermore, if considered in combination with the landscape effects term, it serves to bring the landscape effects in relation to $PGE_j$. That is, while landscape-level terms implicitly capture the emissions impact of total biogenic feedstock harvests ($PGE_0$), *L* provides the necessary adjustment for total potential biogenic emissions within the stationary source boundary.

*P* accounts for biogenic feedstock carbon embodied in process products that pass out of the supply chain as product prior to combustion or after combustion by exiting the stationary source through forms other than as stack emissions. *P* is a unitless term between zero and one that scales *PGE* down so that the portion of biogenic feedstock carbon embodied in products or byproducts is not included in the final results when calculating the assessment factor for biogenic $CO_2$ emissions. As shown in the subsequent sections of this appendix, the equation used to calculate *P* can become quite complex depending on the point of assessment and the production supply chain. This example

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[2] Where $GROW = G / PGE_0$; $AVOIDEMIT = A / PGE_0$; $SITETNC = S / PGE_0$; and $LEAK = Lk / PGE_0$.

[3] In the example presented here in section 1.1 only losses occur between the point of assessment and the forest/farm, so *L* only needs to account for these losses. However, if any products or by-products are produced between the forest/farm and the point of assessment (as in the example presented later in section 1.5), then in order to scale $PGE_i$ up to $PGE_0$, *L* needs to account for these products, as well as any losses. The responsibility for the carbon embodied in these products will separately be accounted for in the *P* term as discussed below.

is constructed to produce the most simplified version of the *P* equation. Equation F.7 defines *P* as calculated in this simplified example:

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j}}. \qquad \textbf{(EQ. F.7)}$$

Note that for many stationary sources, biomass will be used to produce a number of final products, so the summation in equation 0 simply sums over all *i* products that are produced, indexed from 1 to *S*, where *S* is the last point on the carbon trail and represents the stack. In this example, $PROD_2$ is the only product, so this summation would simply be equal to $PROD_2$. Because $PROD_i$ is always negative and the absolute value of the sum of all $PROD_i$ must be less than $PGE_j$, the sum of all $PROD_i$ divided by $PGE_j$ will be a fraction between negative one and zero, and *P* will be a positive number between zero and one.

*P* is one minus the share of $PGE_j$ that is sequestered in products or by-products and is either not emitted to the atmosphere or is sold and eventually emitted to the atmosphere by a different entity. (Note that because *PROD* is negative, technically this expression is one plus a negative number.) In effect, we are taking away the portion of carbon the products or by-products are responsible for and attributing the remainder to the facility. Although this is a relatively simple example, the mathematical representation of this term will change as we consider more complicated scenarios.

We can now rewrite the equation for *NBE* by substituting into Equation F.1. from Equations F.5, F.6, and F.7, resulting in the following:

$$\boldsymbol{NBE = PGE_j \left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left(1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j}\right)}. \qquad \textbf{(EQ. F.8)}$$

The $PGE_0$ in numerator of the *L* term cancels with the $PGE_0$ in the ratio of $(G+A+S+Lk)/\ PGE_0$, and the $PGE_j$ in the denominator of the *L* term cancels with the first $PGE_j$ in the *NBE* equation, so that the term *(GROW + AVOIDEMIT + SITETNC + LEAK)* (PGE)*(L) = G+A+S+Lk*. As discussed above, this represents the actual net emissions on the landscape caused by a harvest of $PGE_0$. All of this canceling is important, because *G, A, S,* and *Lk* are not observable, and even $PGE_0$ may not be readily observable, but we can estimate the unitless values of *L, GROW, AVOIDEMIT, SITETNC,* and *LEAK* through other means (such as retrospective reference point or future anticipated baseline modeling methods) and calculate net emissions while only needing to observe $PGE_j$.

Now, applying numerical values from Figure F-1 and assuming that (*G+A+S+Lk*) = 3 $tCO_2e$ (for the purpose of these examples), net biogenic emissions at point of assessment "*j*=1" are calculated as:

$$\boldsymbol{NBE = 6\left(\frac{3}{10}\right)\left(\frac{10}{6}\right)\left(1 + \frac{(-2)}{6}\right) = 3\left(\frac{4}{6}\right) = 2\ tCO_2e}. \qquad \textbf{(EQ. F.9)}$$

## 2.2. Example 2: Changing the Point of Assessment to Stack Emissions

Continuing through the following examples, necessary modifications are made to the BAF equation for alternative points of assessment. The biogenic $CO_2$ trail depicted by Figure F-2 is used in the following examples to demonstrate that the same NBE can be calculated regardless of the point of assessment. This provides a consistency check because this exercise merely alters the point of assessment but not the biogenic carbon trail.

The second example also relies on the simple hypothetical carbon trail from Figure F-1, with the only difference being that the new point of assessment is stack emissions (represented by $PGE_2$), i.e., "$j$=2."

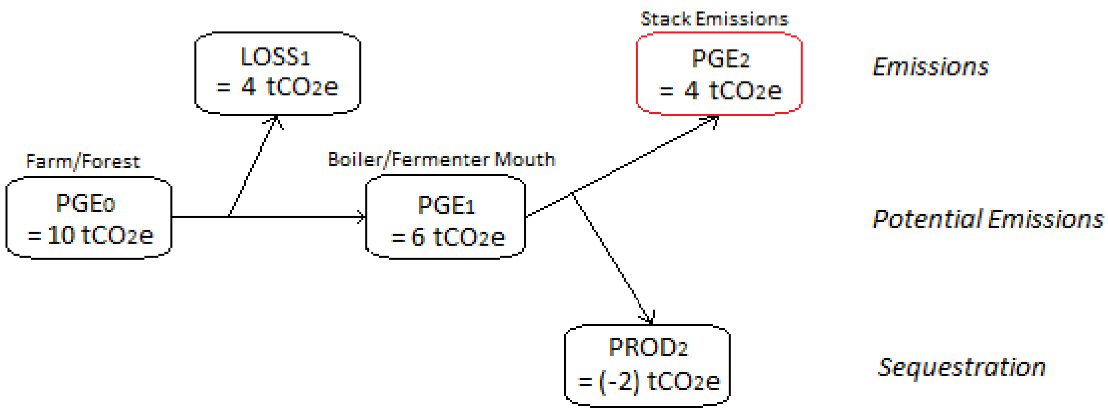


**Figure F-2: Stack Emissions as the Point of Assessment.**

In the first example, all losses occurred before the point of assessment, and all products were produced after the point of assessment. In this example, both losses and products occur before the point of assessment. This change will require us to refine our interpretation of *L* and rewrite our definition of *P* in more general form, as shown in Equation F.10. Equation F.11 is the new more generalized *NBE* equation that covers this example and incorporates this revised definition of *P*. Note that example one is a special case of this new equation.

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=1}^{j} PROD_i}} \qquad \textbf{(EQ. F.10)}$$

$$\boldsymbol{NBE = PGE_j \left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=1}^{j} PROD_i}\right]} \qquad \textbf{(EQ. F.11)}$$

The first point to notice is that *L* remains the ratio of $PGE_0$ to $PGE_j$. Although the expression for *L* has not changed, the numerical value and the interpretation will be different, because $PGE_j$ is now stack emissions. In our original interpretation, *L* was a unitless adjustment factor greater than or equal to one, that scales *PGE* as it was measured at the point of assessment up to account for any losses during transportation or storage between the forest/farm and the point of assessment. In a more general representation, when there can be both products and losses before the point of assessment, *L* scales *PGE* as it was measured at the point of assessment up to account for any losses during transportation or storage between the forest/farm and the point of assessment and to account for any products produced between the forest/farm and the point of assessment. In the general form, *L* does more than just account for losses, it serves as a general scaling factor to relate the carbon content of the biomass feedstock at the point of assessment ($PGE_j$) to the carbon content of biomass that was grown and harvested at the forest/farm and that was required to generate that feedstock ($PGE_0$), accounting for all differences between those two points of measurement.

As in the previous example, *(GROW + AVOIDEMIT + SITETNC + LEAK)* (PGE)*(L) = G+A+S+Lk* represents the primary portion of the equation, the net emissions on the landscape caused by a harvest of $PGE_0$. The remaining job for the *P* term is to determine what share of these net landscape emissions a particular facility is responsible for. The *P* definition from Equation F.7 must be altered to ensure that products that are processed *before* the point of assessment are properly accounted for, resulting in the new definition of *P* in Equation F.10. To do so, the denominator in the fraction used to calculate *P* must be adjusted. Instead of representing the ratio of all products produced to $PGE_j$, this new fraction deducts contributions from products that occur prior to the point of assessment from $PGE_j$ in the denominator ($PGE_j - \sum_{i=1}^{j} PROD_i$). Thinking back to Equation F.4, $PGE_j = PGE_{j-1} + PROD_j$, rearranging this expression and generalizing, the new denominator is simply the expression for $PGE_{j-1}$, so even though the point of assessment is now at the stack, after the products have been produced, our expression for *P* still needs to consider products as a share of *PGE* at a point before the products were produced.

The following numerical example illustrates this point. Because all that has changed in this example is the point of assessment, the value calculated for *NBE* should be the same as in the first example. Substituting numerical values from Figure F-1, and again assuming that (*G+A+S+Lk*) = 3 $tCO_2e$, *NBE* is calculated in equation 0.

$$\boldsymbol{NBE = 4\left(\frac{3}{10}\right)\left(\frac{10}{4}\right)\left[1+\frac{(-2)}{4-(-2)}\right] = 3\left[1-\frac{2}{6}\right] = 3\left(\frac{4}{6}\right) = 2\ mtCO_2e} \qquad \textbf{(EQ. F.12)}$$

Note that although the point of assessment has changed, net biogenic emissions remain consistent with the previous example that evaluated *PGE* at the boiler/fermenter mouth.

## 2.3. Example 3: Changing the Point of Assessment to Forest/Farm Gate

The third example (Figure F-3) also relies on the simple hypothetical carbon trail from Figures F-1 and F-2, with the only difference being that the new point of assessment is the forest/farm (represented by $PGE_0$), i.e., $j$=0.

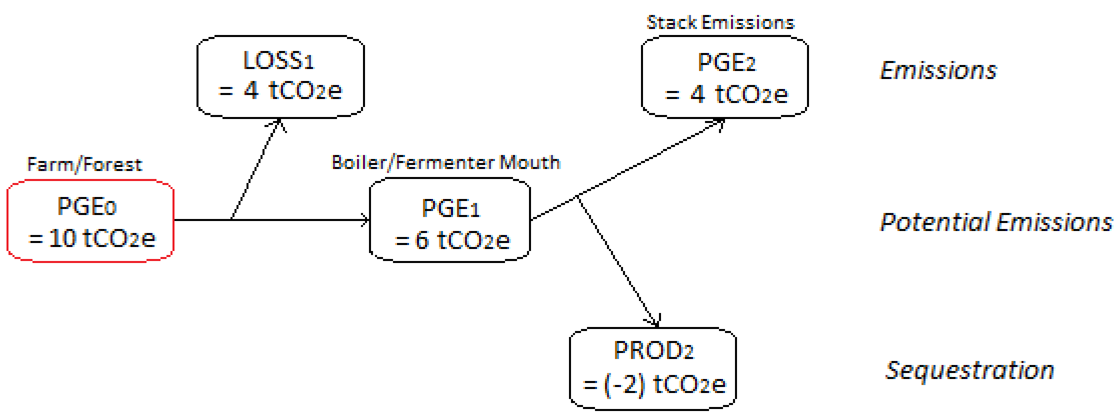


**Figure F-3: Forest/Farm as the Point of Assessment.**

In this example, because the point of assessment is at $PGE_0$, the *L* term is equal to one. The *L* term is still theoretically scaling between the point of assessment and the forest/farm level, but as they are the same, the $L$ term has a value of one. $PGE_0$ is directly measured as it is $PGE_j$ in this instance, so $PGE_j$ does not need to be scaled up, and $L$ is simply equal to one.

In this example, losses ($LOSS_1$) do not enter into the $L$ term, but because the mass balance has not changed, losses still are occurring, but they are occurring after the point of assessment, so the question is how are losses accounted for now? In all examples, ($PGE_j$)*($L$) = $PGE_0$. The equation then assumes that all of $PGE_0$ is emitted, unless the $P$ term specifies that a share of $PGE_0$ is sequestered or contained in products. This means that once $PGE_j$ is transformed to $PGE_0$, unless there are products, the calculation does not change regardless of whether the emissions come from a loss or from the stack. Consider for a moment an example without products (i.e., $P$=1); here $NBE = PGE_j\left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)$, and whether the point of assessment is before or after the losses occur (i.e., $j$ = 0 or 1; or, in this example, at the forest/farm or at the boiler/fermenter mouth), either way, all the $PGE$ terms cancel so that $NBE = G+A+S+Lk$.

Returning to the example shown in figure 3, although losses do not directly enter the $L$ term in this example, they do enter the $P$ term. In this instance, the equation must account for losses that occur after the point of assessment $j$, but before the first source of products, in the $P$ term (as $LOSS_1$ does in this example). Similar to the previous example, a more generic form of the $P$ term is applied. The calculation of $P$ is modified such that the denominator accounts for the losses that occur after the point of assessment, but before the first product is produced. Similar to the previous example, in the denominator of the fraction in the $P$ term, losses are deducted from $PGE_j$ in the denominator ($PGE_j - \sum_{i=j+1}^{PR} LOSS_i$), and as demonstrated in Equation 0F.13, this is essentially the expression for $PGE_{j+1}$. In this case, the term $PR$ stands for the process stage $i$ where the first product is made (that is, the first stage of the industrial process in which $PROD_i > 0$).

In effect, as the $P$ term ensures that the facility is not held responsible for the portion of biogenic carbon that is passed on embodied in products or by-products, the $P$ term also ensures that the facility is not held responsible for the portion of the losses that the products or by-products are responsible for. The responsibility for the losses is shared between the facility and the products or by-products in proportion to their respective shares or biomass that remains after the losses occurred. Equation F.14 presents the $NBE$ equation for this scenario.

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=j+1}^{PR} LOSS_i}} \qquad \textbf{(EQ. F.13)}$$

$$\boldsymbol{NBE = PGE_j\left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=j+1}^{PR} LOSS_i}\right]} \qquad \textbf{(EQ. F.14)}$$

Substituting numerical values from Figure F-1, and again assuming that *(G+A+S+Lk)* = 3 $tCO_2e$, yields the following $NBE$ calculation, which generates the same value for $NBE$ using the more general equation in this example:

$$\boldsymbol{NBE = 10\left(\frac{3}{10}\right)\left(\frac{10}{10}\right)\left[1 + \frac{(-2)}{10-4}\right] = 3\left[1 - \frac{2}{6}\right] = 3\left(\frac{4}{6}\right) = 2\ tCO_2e} \qquad \textbf{(EQ. F.15)}$$

### 2.4. Example 4: Combining Forms from Examples 2 and 3

In order to write a more general form of the *NBE* equation that can either account for products that occur *before* the point of assessment or any losses generated before any products are produced, both forms of the equation as established in examples 2 and 3 must be combined. Doing so results in Equation 0F.17. This generalized form can be applied to all the previous examples of biogenic $CO_2$ calculation at different points of assessment. Note that *S* in the equations below represents the last point on the carbon trail; i.e. the stack, so $PGE_S$ would be stack emissions.

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=1}^{j} PROD_i - \sum_{i=j}^{PR} LOSS_i}} \qquad \textbf{(EQ. F.16)}$$

$$\boldsymbol{NBE = PGE_j \left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j - \sum_{i=1}^{j} PROD_i - \sum_{i=j}^{PR} LOSS_i}\right]}. \qquad \textbf{(EQ. F.17)}$$

Equation F.17 is general enough to cover all points of assessment for any carbon chain so long as all losses ($LOSS_i$) occur before the first product ($PROD_i$) is generated.

As explained in the previous examples, the denominator in the fraction in the *P* term uses the relationships in Equations F.3 and F.4 to transform $PGE_j$ into *PGE* at another point in the carbon trail. Before providing a more complex carbon trail scenario in example 5, it will be useful to introduce some new subscript notation and be a bit more explicit about the subscript notation used so far.

- *0*: First point on the carbon trail; represents the forest/farm. *PGE* begins at *0*.
- *S*: Last point on the carbon trail; represents the stack. $PGE_S$ is stack emissions.
- *PR*: Index number for the first product produced on the carbon trail. $PROD_{PR}$ is the first product.

The indexing for points along the carbon trail begins at 0, the forest/farm, and ends at *S*, the stack. The convention for indexing *LOSS* and *PROD* is that they take an index number equal to the next *PGE*, and a *LOSS* and a *PROD* cannot both occur at the same stage.[4] Additionally, any loss that occurs after all products are produced is indistinguishable from stack emissions $PGE_s$ in these equations. To simplify the equation, it is assumed that *LOSS* is not specified after the last product is produced in the carbon trail, and instead any such losses are rolled into the calculation of $PGE_s$.

With this notation in mind, Equations F.16 and F.17 are rewritten as:

$$\boldsymbol{P = 1 + \frac{PROD_{Total}}{PGE_{PR-1}}} \qquad \textbf{(EQ. F.18)}$$

[4] If $LOSS_i$ and $PROD_i$ occurred at the same stage *i*, it would be ambiguous which came first and whether that $PROD_i$ should share responsibility for $LOSS_i$ or not. For purposes of specifying the full theoretical carbon trail, a PGE would need to be inserted between $LOSS_i$ and a $PROD_i$.

$$NBE = PGE_j \left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1 + \frac{PROD_{Total}}{PGE_{PR-1}}\right], \qquad \textbf{(EQ. F.19)}$$

Where we also define a new variable, $PROD_{Total} = \sum_{i=1}^{S} PROD_i$, or the sum of all products produced along the carbon supply chain. $PGE_{PR-1}$ is potential gross emissions at the point in the carbon supply chain before the last product is produced. In examples 1 through 3 above, $PGE_{PR-1}$ = $PGE_1$. With this new notation, the expression for $P$ in Equation F.18 does not vary with different points of assessment.

The examples so far all assume that losses occur before any products are produced in the carbon supply chain. If a product is produced at a point in the carbon chain before a loss occurs, then that product should not be held responsible for the subsequent loss. Example 5 below shows how the *NBE* equation can be fully generalized to cover this situation.

## 2.5. Example 5: Extending to a More Complex Biogenic $CO_2$ Trail—Toward Fully Generalizing the *NBE* Equation

To illustrate how one could apply the *NBE* equation to a more complex carbon trail, consider the more complex hypothetical carbon trail in Figure F-4. The primary differences between this conceptual diagram and the previous examples are that this example includes four potential points of assessment for $PGE_j$, and multiple points where products are produced and losses occur.

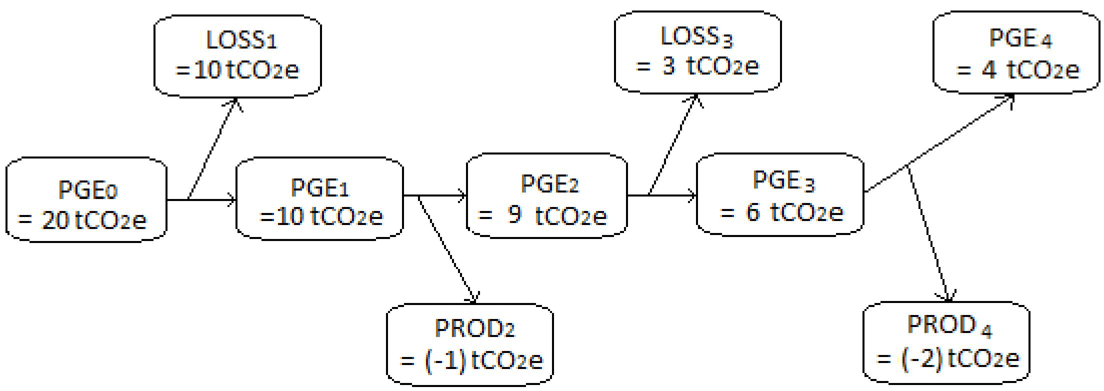


**Figure F-4: Conceptual Diagram Illustrating a More Complex Carbon Trail.**

If one attempts to calculate the $P$ term in the *NBE* equation for this example using Equation F.180, the resulting value would be $P = 1 + PROD_{Total} / PGE_1 = 1+(-3/10) = 7/10$. This would be incorrect though, because it assigns the facility full responsibility for the $LOSS_3$ term and thus does not account for the portion of $LOSS_3$ that $PROD_4$ is accountable for. To properly account for how responsibility for $LOSS_3$ is shared, one should apply the following equations for $P$ and *NBE*:

$$P = 1 + \frac{PROD_{Total} + \sum_{i=PR}^{S}\left[LOSS_i\left(\frac{\sum_{k=i}^{S} PROD_k}{PGE_i}\right)\right]}{PGE_{PR-1}} \qquad \textbf{(EQ. F.20)}$$

$$NBE = PGE_j \left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1 + \frac{PROD_{Total} + \sum_{i=PR}^{S}\left[LOSS_i\left(\frac{\sum_{k=i}^{S} PROD_k}{PGE_i}\right)\right]}{PGE_{PR-1}}\right] \qquad \textbf{(EQ. F.21)}$$

Assuming *j*=1, adding numerical values, and again assuming that *(G+A+S+Lk)* = 3 $tCO_2e$, gives the following calculation of NBE:

$$\boldsymbol{NBE = 10\left(\frac{3}{20}\right)\left(\frac{20}{10}\right)\left[1+\frac{(-3)+(3)\left(\frac{-2}{6}\right)}{10}\right] = 3\left[1-\frac{3+3\left(\frac{1}{3}\right)}{10}\right] = 3\left[1-\frac{4}{10}\right] = 3\left[\frac{6}{10}\right] = 1.8} \quad \textbf{(EQ. F.22)}$$

This example is more complicated than the earlier examples but can still be thought of intuitively. At the end of the carbon trail, $PGE_4$ and $PROD_4$ are responsible for 4 $tCO_2e$ and 2 $tCO_2e$ of $PGE_3$, respectively. They share responsibility for $LOSS_3$ proportionally, 4/6 for $PGE_4$ and 2/6 for $PROD_4$. So $PGE_4$ is responsible for 2 $tCO_2e$ of $LOSS_3$ and $PROD_4$ is responsible for 1 $tCO_2e$ of $LOSS_3$. Put these together and $PGE_4$ is responsible for 6 $tCO_2e$ of $PGE_2$ and $PROD_4$ is responsible for 3 $tCO_2e$ of $PGE_2$. These are also the quantities they are responsible for of $PGE_1$, because the only difference between $PGE_1$ and $PGE_2$ is the 1 $tCO_2e$ that $PROD_2$ is responsible for. So $PGE_4$ is responsible for 6/10 of $PGE_1$. As shown in the calculation above in Equation F.22, this confirms that $P$ = 6/10. Note that because $LOSS_1$ occurs before all products, responsibility for it is shared in the same proportion as responsibility for $PGE_{PR-1}$, so it does not need to be accounted for in $P$.[5]

Although Equation F.21 is a more general representation of NBE than the previous examples, it does not cover all possible scenarios. If the carbon chain is more complex, and there are multiple losses after the first product, then the $\sum_{k=i}^{S} PROD_k$ term in the numerator of the fraction that shares out responsibility for $LOSS_i$ will not account for $PROD_k$'s share of any subsequent losses, instead assigning full responsibility for those subsequent losses to $PGE_s$. Instead of demonstrating this with another example, the same effect can be illustrated by calculating $P$ with respect to $PGE_0$ instead of $PGE_{PR-1}$. This is not a more general way of calculating $P$, but simply an equivalent expression. With this change, one needs to account for $LOSS_1$ in $P$, resulting in the following equations:

$$\boldsymbol{P = 1+\frac{PROD_{Total}+\sum_{i=1}^{S}\left[LOSS_i\left(\frac{\sum_{k=i}^{S}\left(PROD_k+\sum_{l=i+1}^{k}\left(LOSS_l\frac{PROD_k}{PGE_l}\right)\right)}{PGE_i}\right)\right]}{PGE_0}} \quad \textbf{(EQ. F.23)}$$

$$\boldsymbol{NBE = PGE_j\left(\frac{G+A+S+Lk}{PGE_0}\right)\left(\frac{PGE_0}{PGE_j}\right)\left[1+\frac{PROD_{Total}+\sum_{i=1}^{S}\left[LOSS_i\left(\frac{\sum_{k=i}^{S}\left(PROD_k+\sum_{l=i+1}^{k}\left(LOSS_l\frac{PROD_k}{PGE_l}\right)\right)}{PGE_i}\right)\right]}{PGE_0}\right]}. \quad \textbf{(EQ. F.24)}$$

Again assuming *j*=1 and adding numerical values gives the following calculation of NBE:

$$\boldsymbol{NBE = 10\left(\frac{3}{20}\right)\left(\frac{20}{10}\right)\left[1+\frac{(-3)+(10)\left(\frac{(-1)+\left(-2+3\frac{(-2)}{6}\right)}{10}\right)+(3)\left(\frac{-2}{6}\right)}{20}\right]} \quad \textbf{(EQ. F.25)}$$

---

[5] Note that while $LOSS_1$ does not show up in the $P$ term in this example, it does not drop out of the *NBE* calculation entirely as it is accounted for in the *L* term.

$$\boldsymbol{NBE = 3\left[1 - \frac{3+10\left(\frac{1+3}{10}\right)+3\left(\frac{1}{3}\right)}{20}\right] = 3\left[1 - \frac{3+4+1}{20}\right] = 3\left[1 - \frac{8}{20}\right] = 3\left[\frac{12}{20}\right] = 1.8}. \qquad \textbf{(EQ. F.26)}$$

As shown in this numerical example, this expanded equation still generates the same answer for *NBE* in this example.

More complex supply chain carbon trails will require continually more complex expressions for *P.* Comparing the two equivalent expressions above for *P* in Equations F.20 and F.23, defining *P* in relation to $PGE_{PR\text{-}1}$ required an additional nested summation. Adding losses and products to the example will similarly require additional nested summations to fully assign responsibility for subsequent losses to the products and by-products in the supply chain. An important note about the additional complexity required to fully account for a more complex supply chain, is that using the more complex version of the equation (e.g., Equation F.21 instead of Equation F.19) will always assign more responsibility for the losses to products and by-products, thus lowering the calculated value of *P*, lowering the calculated *NBE*, and lowering the ultimate *BAF* value, so it is in the interest of the facility to use the more complex expression. For implementation purposes though, it may be that in some cases supply chains are uniform enough that default values of *P* could be calculated, so that facilities would not need to perform these complex calculations.

The examples contained in this appendix show how to calculate *NBE* in a way that meticulously "follows the tons of biogenic $CO_2$" through a series of hypothetical stationary source production processes. In cases where the supply chain is long and complex, this quickly becomes a very complicated exercise. In application, it is unlikely that all of the relevant carbon masses will be measurable and known. As was described above for *L*, it may be necessary to estimate *P* without actually performing the full calculations described in these examples. Nonetheless, it is useful to think through this idealized application to better understand the *NBE* equation and how it is adaptable across different production processes and points of assessment.

# Appendix G: Illustrative Biogenic Process Attributes

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# 1. Introduction

The purpose of this appendix is to describe possible methods for developing values for biogenic process attributes, which are attributes that reflect feedstock carbon procured by a stationary source that exits assessment through pathways other than emissions from the stack. These biogenic process attributes can include feedstock carbon that is deviated from the process prior to conversion[1] to bioenergy (such as products produced from feedstock material or feedstock losses during transport, storing, and other processing) or feedstock carbon in products that exit the stationary source bioenergy conversion process (such as ethanol or ash).

For demonstrative purposes in this appendix, illustrative values are presented by general stationary source technology and/or process type for specific feedstocks. Values for products can depend on the specific process used at a stationary source, as well as the type of feedstock used.

[1] Conversion refers to technologies or processes that convert biomass into energy directly, in the form of heat or electricity, or may convert it to another form, such as liquid biofuel or combustible biogas. Examples of biomass conversion processes include biomass-fired or co-fired boilers, biomass gasification or pyrolysis systems, and anaerobic digesters.

Values for losses can depend on the type and duration of transport, storage, and processing. In cases where specific information is not available, or if the framework is being applied at a scale larger than stationary-source specific, estimated values for these terms may need to be generalized representative factors. The appendix also provides a detailed technical discussion of the data sources and analytical methods used to develop illustrative values.

# 2. Framework Representation of Feedstock Carbon Losses and Products for Forest- and Agriculture-derived Feedstocks

Feedstock carbon from forest- and agriculture-derived feedstocks can be transformed into products or exit the supply chain (pre- and post-conversion) through means other than the stack. The biogenic assessment factor (*BAF*) equation is designed to transform a measurable or estimated quantity (the carbon content of biomass feedstock used at the point of assessment, represented by potential gross emissions, *PGE*) into a quantity that cannot be directly measured (the net atmospheric contribution of biogenic carbon resulting from use of the quantity of biogenic feedstock that the entity is responsible for, represented by net biogenic emissions, *NBE*). The framework equation, as discussed in the main report as well as Appendix F, is:

$$\boldsymbol{BAF} = \frac{\boldsymbol{NBE}}{\boldsymbol{PGE}} = (\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P}) \qquad \textbf{(EQ. G.1)}$$

The terms below each play a specific role in this equation:

- *PGE* is carbon content of the biogenic feedstock used by a specific entity (or generally consumed). This is a quantity that could be measured or estimated at different points of assessment (e.g., at the boiler mouth, stationary source gate, feedstock production site, or at the stack: wherever the point of assessment needs to be. Thus, this term can have different values indicated by subscripts, representing different points along the supply chain).
- *L* is a unitless adjustment factor greater than or equal to 1 that represents biogenic feedstock carbon that leaves the supply chain (e.g., via transit or decomposition, deviated for use as a product) between the feedstock production site and input into the conversion process at a stationary source. *L* scales *PGE*, as it was measured at the point of assessment, up to account for any losses during transportation or storage between the feedstock production site and the point of assessment. *PGE* times *L* is thus the carbon content of the biomass that was grown at the feedstock production site in order to deliver the quantity of feedstock measured at the point of assessment.
- *P* is a unitless adjustment factor between zero and one, equal to the share of the carbon content of the feedstock at the point of assessment that is emitted to the atmosphere by a stationary source (versus that which is embedded in products). In effect, this term also reflects the share of carbon that remains in products, that is either not emitted to the atmosphere or is sold and eventually emitted to the atmosphere by a downstream user.

## 2.1. Equations Underlying *P* and *L*

### 2.1.1. *P*

*P* accounts for the carbon content of the feedstock material that is emitted to the atmosphere and therefore also accounts for biogenic carbon in products that is not emitted from the stack and leaves the producing entity (pre- and post-conversion) for downstream use or disposal. Examples of products that may exit the supply chain prior to conversion include:

- Wood material in products (lumber, wood pulp, panel products);
- Mill residues sold/transferred to a separate stationary source for use as raw material or fuel;
- Bark sold/transferred to a separate stationary source for fuel;
- Bark sold for mulch;
- Agricultural by-products (e.g., stover, stalks, straws, husks, hulls, etc.) sold/transferred to a separate stationary source for use as fuel; or
- Pulping by-products (tall oil, turpentine).

Examples of products that may exit the supply chain after conversion include:

- Dried distillers grains (from ethanol production);
- Ethanol; or
- Bottom ash, flyash, or biochar (e.g., materials containing unburned carbon).

Equation F.7 from Appendix F defines *P* as calculated in this example.

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j}} \qquad \textbf{(EQ. G.2)}$$

$PROD_i$ represents the sum of all pre- and post-conversion products and $PGE_j$ is *PGE* evaluated at point of assessment (*j*). Note that for many stationary sources, biogenic feedstocks will be used to produce a number of final products, so the summation in Equation G.2 simply sums over all *i* products that are produced prior to the bioenergy conversion process with biogenic $CO_2$ stack emissions, indexed from 1 to *S*, where *S* is the last point on the carbon trail and represents the stack. For example, if $PROD_2$ is the only product, this summation would simply be equal to $PROD_2$. Since $PROD_i$ is always negative and the absolute value of the sum of all $PROD_i$ must be less than $PGE_j$, the sum of all $PROD_i$ divided by $PGE_j$ will be a fraction that has a negative value, and *P* as calculated above will be a positive number between 0 and 1 (technically this expression is 1 plus a negative number). In effect, this calculation takes away the portion of carbon the products are responsible for, and attributing the remainder to the stationary source. Although this is a relatively simple example, the mathematical representation of this term and more complicated scenarios are illustrated in Appendix F.

The framework recognizes that some post-conversion materials—such as fly ash—can potentially be sold and used for a number of commercial purposes (i.e., cement manufacturing) or disposed of. Either way, such post-conversion materials are considered under the *P* term because they pass through the stationary source, exiting through means other than emissions out the stack. There is limited data on some post-conversion materials: limited and/or emerging scientific data in some cases (such as biochar) and publically available production and market data (e.g., end uses, sales of such materials). Also, whether or not these materials are sold, the carbon contained in this material can in some circumstances remain sequestered for some duration of time.

### 2.1.2. *L*

The *L* term reflects biogenic feedstock losses in transportation to, storage, and processing at a stationary source and any products that exit the supply chain prior to the conversion process. This value facilitates the link between the quantity of feedstock received at the stationary source, the quantity of feedstock that enters the stationary source process, and the quantity of feedstock grown on or removed from (e.g., in the case of crop residues) the land. Some stationary sources may measure both feedstock delivered at the gate and at the point of entry into the conversion process (e.g., boiler mouth) and thus have the data to calculate its onsite *L* term. Depending on the data that stationary sources have, *PGE* could be estimated, measured (i.e., the mass of the biogenic feedstock as it enters a stationary source process and converting that mass into its $CO_2$ equivalents), or perhaps back calculated from the direct process emissions at the stationary source. An estimate of losses associated with storage at the stationary source and any pre-conversion products plays a role in the estimation of the amount of feedstock actually delivered to the site versus that entering the supply chain at the production landscape.

The purpose of *L* here is twofold. First, *L* represents the transformation of $PGE_0$ (*PGE* evaluated at the feedstock production/harvest site) to $PGE_j$ (*PGE* evaluated at point of assessment *j*), implicitly capturing any losses ($LOSS_i$) and any pre-conversion products ($PROD_i$) that occur between these two PGEs. Note that this fraction will result in a positive number. Second, if considered in combination with the landscape effects term, it serves to bring the landscape effects in relation to $PGE_j$.

$$\boldsymbol{L = \left(\frac{PGE_0}{PGE_j}\right)} \qquad \textbf{(EQ. G.3)}$$

Calculations presented in Section 4 of this document generate illustrative loss terms for when such site or supply chain-specific calculations are not possible or when representative factors are necessary within a specific program or policy application (e.g., regional application of the framework).

# 3. Evaluation of Post-Conversion Products

In addition to information on the carbon content of the input feedstock, the methodology applied in this appendix also requires knowledge of the relationship between input and output. This information is often necessary to properly express *P* as demonstrated in the equations below:

**P Equation**

$$\boldsymbol{P = 1 + \frac{\sum_i (Mass\ of product_i \times Carbon\ content\ of\ product_i)}{(Mass\ feedstock\ input \times Carbon\ content\ of\ feedstock\ input)}}$$

Thus:

$$P = 1 + \frac{\sum_i PROD_i}{PGE_j} \qquad \textbf{(EQ. G.4)}$$

Where:

$$PROD_i = (Mass\ of\ product_i \times Carbon\ content\ of\ product_i) \qquad \textbf{(EQ. G.5)}$$

$$PGE_j = (Mass\ feedstock\ input \times Carbon\ content\ of\ feedstock\ input)$$
$$at\ the\ point\ of\ assessment\ j$$

As explained in Appendix F, *P* is 1 minus the share of $PGE_j$ that remains in products or by-products and is either not emitted to the atmosphere or is sold and eventually emitted to the atmosphere by a different entity. (Note that because *PROD* is negative, technically this expression is 1 plus a negative number.) Because of the variability of different conversion technologies (e.g., boiler types), this appendix relies on mass balance data to establish the relationship between input and output and then uses the equations above to calculate example values for *P*.

The following section explores methods that could be used to estimate $PROD_i$ parameters contributing to *P*. The example process and stationary source combinations below are addressed in this appendix for use in case study applications per different baselines:

- Co-firing in electric generating units (EGUs) (using forest-derived feedstocks); and
- Pyrolysis chambers in pyrolysis facilities (using corn stover).

Carbon content values for agricultural feedstocks are taken from Spokas (2010), while values for forest-derived feedstocks are from Skog (2008). Table G-1 shows values for each, expressed as percent carbon on a dry weight basis.

**Table G-1. Carbon Content of Different Biogenic Feedstocks.**

| Input Feedstocks | Percent C by Dry Weight |
|---|---|
| **Roundwood** | 50 |
| **Logging Residue** | 50 |
| **Corn Stover**[2] | 44 |

## 3.1. Co-firing in Electric Generating Units (EGUs)

Some EGUs are designed for 100% biomass combustion (Johansson et al., 2003). However, most EGUs are optimized for coal, natural gas, or oil. Yet, some EGUs can co-fire a certain percentage of biogenic fuel to be used in addition to the fossil-based feedstock (Demirbas, 2005), depending on the biogenic fuel and boiler design (e.g., solid-fuel boilers may be able to co-fire solid biomass). Generally, any type of biogenic feedstock can be used, but most often in the U.S., feedstocks are

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[2] Although presented more specifically in examples as 44% in this appendix, in Appendix L, the carbon content of corn stover is assumed to be 50% for consistency with other FABA feedstock case studies.

forest-derived (e.g., roundwood, logging, or mill residues), often converted to pellets or chips (Demirbas, 2005). Regardless of feedstock blend (i.e., percentage of biomass in fuel input), an EGU combusts the feedstock in a boiler to heat water into steam, which is used to drive turbines and generate electricity.

### 3.1.1. Estimated *P* Term Values

Application of the framework could first reflect the relevant stationary source technologies and processes used at EGUs, then the applicable biogenic feedstock inputs and residual outputs. The process of co-firing biomass at an EGU does not produce non-energy/heat commercial products other than post-combustion materials like ash. As noted above, an EGU combusts the feedstock in a boiler to heat water and produce steam, which is used to drive turbines and generate electricity.[3]

To calculate *P* in this illustrative scenario, the amount of carbon in the post-process materials produced can be expressed as a proportion of carbon in the input feedstock. This value differs depending on the boiler type and efficiency. Boilers vary from facility to facility, because there are a number of different boiler types to choose from, including fluidized bed boilers, stoker boilers, or cyclone boilers. Each functions at a different efficiency, and some can accept higher percentages of biogenic feedstock in the fuel blend. Differences among boilers in efficiency or temperature of combustion can result in different amounts of post-process materials, or different carbon content of the materials (Tarelho et al., 2011). Literature suggests boiler type is an important predictor of unburnt carbon remaining, but the feedstock blend could impact efficiency of combustion and thus result in more or less post-process materials (e.g., ash) or a higher or lower carbon content of the materials (Demirbas, 2005). Generally, agricultural biomass features higher ash content than woody biomass (Cassidy and Ashton, 2007).

Regardless of process, the amount of unburnt carbon remaining in the ash is typically expected to be low compared to the amount of feedstock carbon entering the conversion process (Kaufmann et al., 2000), but, as it depends on the conversation process efficiency, rates can vary substantially. Compared with modern, high-efficiency boilers, traditional grate boilers often result in less efficient combustion—and therefore feature higher levels of unburnt carbon. Tollin (2000) suggests carbon content of ash can approach 50% in low efficiency grate boilers. Biomass stoker boilers may have 30–40% carbon in the ash (Gustafson and Raffaeli, 2009). Pitman (2006) reports average carbon content in wood ash to range from 7 to 50% (average 26%) in commercial boilers in the eastern U.S.

Those values pertain to carbon content of the ash itself, not the proportion of incoming feedstock carbon remaining. The amount of ash generated in the combustion process is typically ≈10% (by weight) of incoming feedstocks. Table G-2 shows an example calculation based on a feedstock with 50% carbon content, as well as other hypothetical assumption values based on Table G-1.

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[3]Theoretically, if combustion in an EGU was 100% efficient in converting biogenic carbon to $CO_2$, then $P$ could be set to $P = 1$ to reflect that there is no carbon remaining in post-combustion material (ash) and that there are no products produced from the feedstock used in the EGU (i.e., $PROD = 0$).

**Table G-2. Sample *P* for Ash Information for Low Efficiency Boiler.**

| Variable | Value |
|---|---|
| **Incoming feedstock** | 1 ton |
| **Carbon content of feedstock** | 50% |
| **$PGE_j$: Total incoming carbon** | 0.5 tons |
| **Ash content of feedstock** | 10% |
| **Ash remaining post-combustion** | 0.1 tons |
| **Carbon content of ash** | 26% |
| **Total remaining carbon post-combustion** | 0.026 tons |
| **PROD: Proportion of initial carbon remaining post-combustion to PGEj** | −0.052[1] |
| **P** | 0.948[2] |

Notes:

1. Carbon in products is subtracted (i.e., has a negative value) because product carbon does not enter the atmosphere with the biogenic $CO_2$ from the conversion process. *PROD* = -(0.026/0.5)
2. *P* = 1 + (-0.052) = 0.948

*PROD* (ash) values would be lower for modern, high-efficiency boilers, leading to higher *P* terms. Nussbaumer and Hasler (1999) and Johansson et al. (2003) report fly ash carbon concentrations of 1–10% in modern boilers. Demirbas (2000) uses an average ash content of 0.5%. Kaufmann et al. (2000) analyzed ash samples and determined carbon content to be no more than 0.05%. Fluidized-bed boilers typically achieve nearly 100% combustion, meaning practically 0% carbon remains in ash (EPA, 2007; Gustafson and Raffaeli, 2009). As above, the mass of the ash itself is a small fraction of total incoming feedstock, meaning the remaining carbon is only a small percentage of the total incoming carbon. Tarelho et al. (2011) present the best data on this, showing total unburnt carbon left in post-combustion residuals to be only 0.7–2.8% of the original feedstock carbon (i.e., 0.007–0.028 tons per ton of feedstock carbon input).

The value varies depending on the efficiency of the boiler, but generally the majority of the incoming carbon will be released via combustion as $CO_2$. Table G-3 shows example illustrative values of the *P* term based on boiler efficiency. Actual values for *P* are expected to vary depending on boiler technology being used at a facility and feedstock type.

**Table G-3. Example Estimated Range of *P* Values for Co-Firing Biomass in Electrical Generating Units.**

| Co-firing Boiler Type | *P* |
|---|---|
| High Efficiency | 0.972[1] to 0.993[2] |
| Low Efficiency (as shown in Table G-2 above) | 0.948 |

[1] *P* = 1 + (-*PROD*/*PGE*) = 1 + (-0.028 tons C in ash/ton of C input in feedstock) = 0.972

[2] *P* = 1 + (-*PROD*/*PGE*) = 1 + (-0.007 tons C in ash/ton of C input in feedstock) = 0.993

### 3.1.2. Discussion

Serious data gaps exist in developing representative *P* values for co-firing boilers at EGUs. Overall, little information exists about carbon content of post-combustion materials. Because of this, determining representative *P* values is difficult, and considerable uncertainty remains.

More detailed data from proximate and ultimate analysis of different biogenic feedstocks are required, as are engineering analyses related to combustion using different technologies. For example, ash content will impact boiler heat surface fouling, which will impact the completeness of combustion—and by extension the *P* value for different combustion technology—for different feedstock combinations. As noted above, the process of co-firing biomass at an EGU can be done with a number of different biogenic feedstocks and a number of different boilers. Furthermore, the fossil fuel/biomass blend can vary greatly depending on cost, supply, boiler specifics, and a number of other factors.

However, despite the lack of data, it is clear that in most scenarios much of the carbon is emitted as $CO_2$ during combustion. Therefore, although the carbon content of post-combustion materials may vary, the average is expected to be low.

## 3.2. Pyrolysis

The pyrolysis process involves the thermal destruction of organic materials in an environment void of oxygen (Demirbas, 2005). When the biomass is heated in a pyrolysis chamber, it produces a hydrocarbon rich gas mixture (syngas), an oil-like liquid (bio-oil), and a carbon rich solid material (biochar). The relative amounts of each product depend on whether the facility employs high heat fast pyrolysis or low heat slow pyrolysis. Typical mass yield ranges for fast pyrolysis are bio-oil 50–70%, syngas 10–30%, and biochar 10–25%. Typical mass yield ranges for slow pyrolysis are: bio-oil 20–50%, syngas 20–50%, biochar 25–35% (UK Biochar Research Center (UK BRC), 2009). In most pyrolysis processes, the syngas is used to fuel the system (Mullen et al., 2010). Figure G-1 shows the pyrolysis process in the context of the carbon cycle.

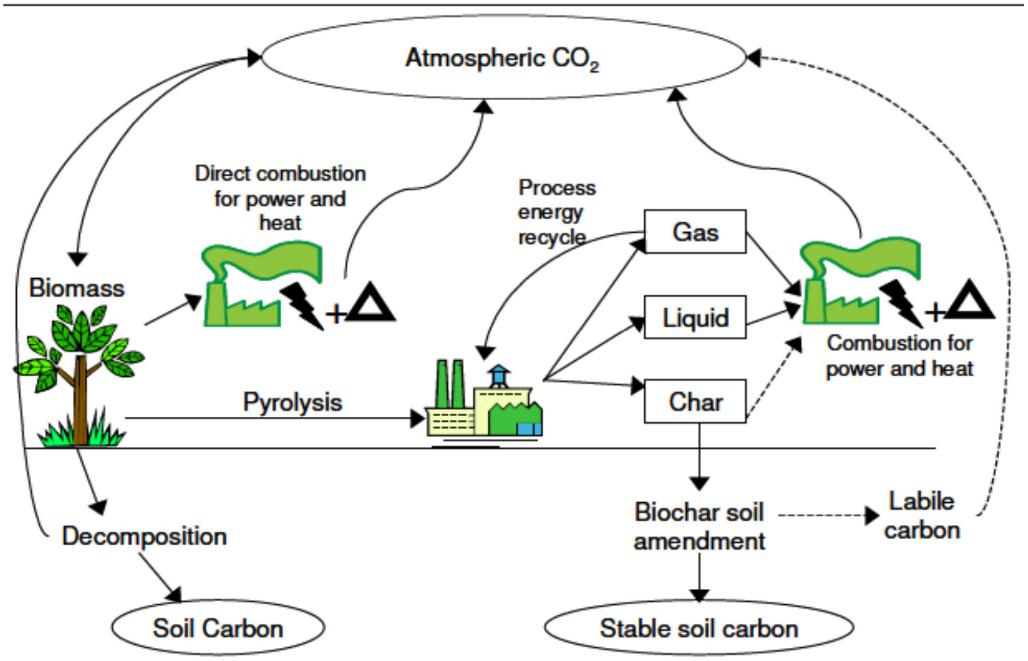


**Figure G-1. Diagram of Pyrolysis Process.**

### 3.2.1. Estimated *P* Term Values

For purposes of this framework, biochar, bio-oil, and syngas can be considered under the *P* term. Values are estimated for both fast and slow processes, based on the sample mass balance information in Table G-4. Table G-4 uses Mullen et al. (2010) numbers for fast pyrolysis and Ringer et al. (2006) numbers for slow pyrolysis. Carbon content of biochar is derived from Roberts et al. (2010) and Brown et al. (2010). Corn stover is used as an example feedstock, assuming a carbon content of 44%.

Note that carbon content of bio-oil and syngas is the same for both processes, the differences lie merely in the amount of each material produced (Mullen et al., 2010).

**Table G-4. Sample *P* Information for Pyrolysis of Corn Stover.**

| Sample P Information | Fast Pyrolysis (high heat, 600°C) | Slow Pyrolysis (low heat, 300°C) |
|---|---|---|
| | % | |
| **Input feedstock (Corn stover)** | 100 | 100 |
| **Bio-oil** | 61.6 (includes water content) | 30[a] |
| **Syngas** | 21.9 | 35[a] |
| **Biochar** | 17.0 | 35[1] |
| **Carbon Content of Bio-oil** | 53.97 | 53.97[2] |
| **Carbon Content of Syngas** | 16.7 | 16.7[3] |
| **Carbon Content of Biochar** | 57.29 | 67.68[c] |

[1]Ringer et al. (2006).
[2] Mullen et al. (2010).
[3] Roberts et al. (2010). p. 9.

This illustrative stover pyrolysis example assumes that the assessment boundary encapsulates the actual pyrolysis process chamber including the syngas boiler and represents a pyrolysis facility where all of the syngas is used onsite for process energy (instead of some of it being shipped offsite). In the syngas boiler stage, the syngas is fully combusted and therefore *P* (syngas) = 0. Thus, the *P* value in this example is only calculated for the bio-oil and biochar products. As explained in Appendix F, *P* is 1 minus the share of $PGE_j$ that remains in products or by-products and is either not emitted to the atmosphere or is sold and eventually emitted to the atmosphere by a different entity. (Note that because *PROD* is negative, technically this expression is 1 plus a negative number.)

For fast pyrolysis, *P* values can be calculated as follows per ton of feedstock input:

*P* Equation for Fast Pyrolysis

$$\boldsymbol{P = 1 + \frac{\sum_i (Mass\ of product_i \times Carbon\ content\ of\ product_i)}{(Mass\ feedstock\ input \times Carbon\ content\ of\ feedstock\ input)} =}$$

$$\boldsymbol{1 + \left(-\frac{0 + (Mass\ bio\text{-}oil \times C\ in\ bio\text{-}oil) + (Mass\ biochar \times C\ in\ biochar)}{(Mass\ of\ corn\ stover \times Carbon\ content\ of\ corn\ stover)}\right)} \qquad \textbf{(EQ. G.6)}$$

$$\boldsymbol{P = 1 + \{-[0 + (0.616 \times 0.5397) + (0.17 \times 0.5729)] \div (1 \times 0.44)\}}$$

$$\boldsymbol{P} = \mathbf{1} + (-\mathbf{0.9769}) = \mathbf{0.0231}$$

For slow pyrolysis *P* values can be calculated as follows per ton of feedstock input:

*P* Equation for Slow Pyrolysis

$$\boldsymbol{P} = \mathbf{1} + \frac{\sum_i (\boldsymbol{Mass\ of product_i} \times \boldsymbol{Carbon\ content\ of\ product_i})}{(\boldsymbol{Mass\ feedstock\ input} \times \boldsymbol{Carbon\ content\ of\ feedstock\ input})} =$$

$$\mathbf{1} + \left(-\frac{\mathbf{0} + (\boldsymbol{Mass\ bio\text{-}oil} \times \boldsymbol{C\ in\ bio\text{-}oil}) + (\boldsymbol{Mass\ biochar} \times \boldsymbol{C\ in\ biochar})}{(\boldsymbol{Mass\ of\ corn\ stover} \times \boldsymbol{Carbon\ content\ of\ corn\ stover})}\right) \qquad \textbf{(EQ. G.7)}$$

$$\boldsymbol{P} = \mathbf{1} + \{-[\mathbf{0} + (\mathbf{0.30} \times \mathbf{0.5397}) + (\mathbf{0.35} \times \mathbf{0.6768})] \div (\mathbf{1} \times \mathbf{0.44})\}$$

$$\boldsymbol{P} = \mathbf{1} + (-\mathbf{0.9063}) = \mathbf{0.0937}$$

#### 3.2.2. Discussion

The values estimated in this appendix are meant to be illustrative examples only. Pyrolysis is not yet being undertaken on a commercial scale; as a result, it is difficult to derive robust values. Hammond (2011) notes that "pyrolysis has not yet been demonstrated at … large scales," and that "due to the immature state of [pyrolysis] technology … there is a lack of good quality datasets and therefore greater uncertainty in the data than is desirable." This is similarly true for stationary sources using dedicated energy crops (e.g., switchgrass) and short-rotation woody crops (e.g., poplar) as a feedstock for pyrolysis.

For example, Ringer et al. (2006), Enders et al. (2012), and Wright et al. (2008) show the production percentage of biochar can range from 0–77%. Much of this range is the result of differences in process, namely the decision to employ fast pyrolysis or slow pyrolysis. Slow pyrolysis produces more biochar (35%); fast pyrolysis produces less (17%). Table G-4 combines values from Enders et al. (2012), Wright et al. (2008), and Ringer et al. (2006).

Spokas (2010) notes that biochar "variability is based on the conditions of pyrolysis and the biomass parent material, with biochar spanning the range of various forms of black carbon. Thereby, this variability induces a broad spectrum in the observed rates of reactivity and, correspondingly, the overall chemical and microbial stability." Furthermore, the literature is complicated by the lack of uniformity when it comes to the nomenclature for the products of biomass conversion. Char, charcoal, soot, graphitic carbon, ash, coal, and black carbon have all been used to describe the solid residual products." As a result, the potential values of *P* for pyrolysis techniques could vary from facility to facility. Representative values could be calculated using ranges, averages or median values, as done in the example above.

### 3.3 Conversion of Forest-Derived Industrial Byproducts

Producers of industrial forest-derived products and by-products may use roundwood or wood chips as raw material to manufacture paper or wood products. Various wood product manufacturing processes can result in the production of mill residues including bark, saw dust, sander dust, and panel trim, which may be sold downstream to other users as products or used onsite as boiler fuel. The mass of wood converted to products is dependent on the specific products being produced and production processes used. The amount of mill residues used as onsite boiler fuel instead of being sold for use as a raw material or fuel at a separate entity is highly site-specific.

This section provides an example of a hypothetical wood products facility that converts roundwood into veneer for onsite plywood manufacture, uses the cores remaining after peeling of veneer to produce lumber, and produces additional lumber from logs. This illustrative facility burns 60% of the mill residues it generates (bark, plywood trim, hog fuel, and sanderdust) in an onsite boiler and sells the remaining 40% of mill residues (sawdust and planar shavings) to other downstream users for fuel or use in other products (e.g., pellets, particleboard).

### 3.3.1 Estimated *P* Term Values

To calculate *P* for this example wood products facility, the amount of carbon in the products produced can be expressed as a proportion of carbon in the input feedstock. For each dry ton of carbon in the input feedstock, in this hypothetical example, a site-specific estimate of 0.2 dry tons of carbon in mill residues are generated, of which 0.6 x 0.2 = 0.12 dry tons of carbon are used onsite as boiler fuel and 0.4 x 0.2 = 0.08 dry tons of carbon are sold as products. Mill residues are collected from various points in the production process and are either routed to onsite boiler fuel storage (0.12 dry tons of carbon per dry ton of carbon in the input feedstock) or to loadout bins for transfer to other entities (0.08 dry tons of carbon per dry ton of carbon in the input feedstock). Mill residues collected for boiler fuel at this example mill are used within a few days so boiler fuel storage losses are negligible. In addition, this hypothetical example includes a site-specific estimate of 0.00012 tons of carbon remaining in the boiler ash following combustion of mill residues.

Because the boiler is supporting various different equipment for wood products manufacturing, it is separate from the plywood and lumber production lines. The mass of mill residues combusted in the boiler is not measured at every mill residue generation point in the facility, but instead is determined at the boiler. Similarly, the mass of mill residues (sawdust and planar shavings) sold as a product is determined at the loadout bins rather than being measured at every saw and planar within the facility. It may not be necessary to distinguish between pre- and post-conversion products because the facility can calculate the following dry mass amounts on a per-ton-of-carbon-input basis:

- Mass of carbon in the incoming logs ($PGE_j$ = 1 ton of C input);
- Mass of carbon retained in the plywood and lumber products ($PROD_1$ = 0.8 tons of C in plywood and lumber);
- Mass of carbon in mill residues sold ($PROD_2$ = 0.08 tons of C in residuals sold);
- Mass of carbon in mill residues burned in the boiler (0.12 tons of C burned); and
- Mass of carbon remaining in the boiler ash ($PROD_3$ = 0.00012 tons of C remaining in ash).

All products may be considered within the *PROD* and *P* terms. *P* can be calculated as follows:

$$\boldsymbol{P = 1 + \frac{\sum_{i=1}^{S} PROD_i}{PGE_j} = 1 + \left[\frac{-(PROD_1 + PROD_2 + PROD_3)}{PGE_j}\right]} \qquad \textbf{(EQ. G.8)}$$

$$\boldsymbol{P = 1 + \left[\frac{-(0.8 + 0.08 + 0.00012)\ tons\ C\ in\ products}{1\ ton\ C\ in\ the\ feedstock\ input}\right] = 0.12}$$

Carbon in products is subtracted (i.e., has a negative value) because product carbon does not enter the atmosphere with the biogenic $CO_2$ from the conversion process at this wood products facility.

### 3.3.2 Discussion

The hypothetical example above shows that, in the case of a wood products facility, the majority of the carbon entering the facility can remain in the wood products produced by the facility, including mill residues sold to other entities. The example also illustrates inclusion of all products (*PROD*) in the *P* term when it is not clear whether the products are pre- or post-conversion products because the material is collected from various points within the production process that are not necessarily upstream or downstream from the bioenergy conversion process (i.e., the boiler). In this case, use of the point of assessment ($PGE_j$) as the feedstock carbon input to the facility (roundwood) allowed for the *P* calculation to be done without distinguishing between pre- and post-conversion products in the facility.

As shown in Equation G.3 above, *L* in this example would be calculated as the ratio of $PGE_o$ (at the feedstock production site) and $PGE_j$ (roundwood entering the facility). Thus, for the calculation of *P* in this example, it is assumed that there are no physical feedstock losses (*LOSS*) prior to the residues entering the conversion process and therefore no need to distinguish products generated within the facility as pre- or post-conversion for purposes of calculating *L*.[4]

# 4. Evaluation of Pre-conversion Products and Losses During Transportation, Storage, and Processing

There are few established methods for tracking and/or calculating feedstock losses along the supply chain, especially for raw herbaceous feedstock (Miranowski et al., 2010). However, when conducting a mass balance-based calculation, having some means to estimate the flows of biogenic feedstock carbon from feedstock production to either emissions out of the stationary source stack or post-conversion products, some assessment of these supply chain flows can relate biogenic $CO_2$ emissions to feedstock production. This framework equation includes a term (*L*), which scales *PGE* as it is estimated or measured at the point of assessment up to account for any pre-conversion products exiting the supply chain or feedstock losses during transportation, storage, or processing between the feedstock production site (e.g., forest, farm) and the point of assessment.

As the feedstock moves through the supply chain, deviated or lost feedstock carbon may occur due to pre-conversion products or losses during transport, storage, and processing/handling between or within different stages of the supply chain. The *L* term actually refers to two separate loss components: one that reflects feedstock carbon taken from the supply chain for other purposes such as pre-conversion products (e.g., lumber, bark for mulch) and another that reflects physical losses between the feedstock production site and the point of assessment. The volume of physical losses can depend on feedstock type, technologies applied, storage length and type, and several other factors. Therefore, measuring direct stack $CO_2$ emissions at the stationary source would likely

[4] Losses and only the pre-conversion products occurring between $PGE_o$ and $PGE_j$ are included in *L*.

be an underestimate of the total net biogenic $CO_2$ emissions associated with procurement and use of the biogenic feedstock. The adjustment of *PGE*, the carbon content of the biomass feedstock used by a specific entity at different points of the assessment, by the term *L* can vary for each supply chain, unless generalizations are made to create representative values.

Representative values for losses could depend on identification of different storage scenarios and transportation scenarios by feedstock type and could also take the region into consideration if appropriate (e.g., effects of regional climate on stored feedstock). Methods for calculating representative feedstock loss values related to transport, storage, and processing that could be developed for stationary sources to use, in lieu of case-specific, customized calculations, are shown below. This appendix does not develop values for possible products coming out of the supply chain between the feedstock production site and point of assessment because of the many different product possibilities and permutations that exist. More detailed discussion of the underlying mathematical computations related to how products can be accounted for in Appendix F. The reminder of this appendix focuses on the physical losses component of *L*.

## 4.1. Feedstock Losses during Transportation, Storage and Processing

Weighing the feedstock at the sale transaction point is considered common practice. In commercial operations, this is usually done at a stationary source's delivery point scale. The delivered moisture content can be used to calculate the dry weight in the load, and the payment is normally on a dry weight basis. The material is usually placed in storage prior to being introduced into the process. The conditions for storage, including feedstock type, moisture content, time of storage, climate conditions, and the amount of protection against added moisture offered by the stationary source, can affect feedstock storage losses. Feedstocks taken immediately from the scale to the process or into storage that is dry and cool are likely to have fewer losses than feedstocks stored in uncovered piles on the ground in a warm, moist climate for several months (see below for research findings relating to such losses).

As described above, a portion of the *L* term represents the $CO_2$ equivalent of feedstock losses (e.g., decomposition, too dirty or destroyed to use in boiler), or $LOSS_i$[5], that occur between the weigh-in point of the feedstock at the stationary source and the entry point of the production process (boiler or other processor). If the feedstock is processed immediately by the stationary source, there may be minimal, or zero, losses in storage. In many situations, however, the feedstock must be stored prior to utilization, because harvest and delivery schedules are cyclical, whereas biogenic feedstock demand and utilization needs would be continuous. Those losses are due to the decomposition in a storage system, such as a chip pile and onsite storage, which is likely to last one year or less. Specific applications of the framework could include calculations of the emissions related to decay: however, the illustrative calculations in this appendix count those losses as immediate and do not include any long-term decay rates (see section below on onsite storage losses).

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[5] There can be various points of losses along the supply chain. These losses can be given individual loss terms (e.g., $LOSS_1$, $LOSS_2$, $LOSS_3$).

In contrast to stationary source onsite storage losses that can happen subsequent to the feedstock weighing, potential harvest and transport losses could also occur and thus could be considered for treatment in the framework, depending on available data. For some feedstocks, losses may occur during transportation between the feedstock production site and the stationary source. For example, some of the material harvested from a farm (e.g., switchgrass) may be lost from the truck while in transit, or decay might occur while the feedstock is in storage at the stationary source (Qin et al., 2006). Such losses are not typically calculated or included in the delivered weight; thus, few data are available on this aspect of losses. These losses are expected to be rather minor and likely depend on the biogenic feedstock material that is harvested or collected.

## 4.2. Possible Methods for Developing Illustrative *LOSS* Values for Onsite Storage and Processing

This section describes a methodology for calculating representative values for the relevant loss factors ($LOSS_i$) included in the $L$ term. Biomass types considered below include herbaceous (switchgrass, hay, etc.), woody biomass (logging residues, etc.), and agricultural residues (corn stover, etc.).

In general, of all the biomass losses occurring across a given supply chain, the *storage* losses (e.g., decomposition, too dirty or destroyed to use in boiler) are potentially the most significant. Even for long-distance supply chains including international shipping, Hamelinck et al. (2005) estimated a total loss throughout the international supply chain of 15% of dry matter; 13% of these losses were associated with storage, 2% with processing, and <1% with handling.

One way to calculate onsite storage losses is to determine the feedstock delivered to the stationary source versus the amount that enters into the conversion process. The difference between the delivered weight and the pre-conversion input weight represents the onsite loss. This pre-conversion input weight can also be compared with the direct emissions (stack emissions) from the process to help determine accuracy of the calculation. However, such multi-point measurement onsite is currently not common practice. If the point of assessment, where *PGE* is calculated, is when the feedstock is entering the conversion process (e.g., after any storage losses have occurred), the $L$ term (which includes losses represented by $LOSS_i$) is needed in the equation that converts *PGE* to *NBE*, to represent any losses (and pre-conversion products) that could have occurred prior to initial calculation of *PGE*.

Biogenic feedstocks contain a significant amount of water which can change as the material dries in storage and/or absorbs water through precipitation. Dry matter refers to the weight of material without water (i.e., the weight of material when it is completely dried). Dry matter loss in storage (e.g., due to decay, physical losses) is a function of several factors that will differ between stationary sources. The factors include (1) the type and packaging (loose, bundles, etc.) of feedstock; (2) the moisture content of the stored feedstock; (3) the type of storage facility at the stationary source; (4) the length of time the material is stored; and (5) the climatic conditions (e.g., temperature, humidity) during the storage period.

Several studies give estimates of dry matter loss for different types of feedstocks. For woody biomass, losses of 12% were found in green chips stored in a large pile for 7 months (Thornqvist and Jirjis, 1990). Although little to no dry matter losses can be expected when storing logs, bundles of logging residues exhibit higher dry matter losses (Jirjis, 1995). Afzal et al. (2010), for instance, observed higher dry matter losses in woodchip piles (8–27%) than in bundles (≈3%) over one year. Jirjis and Nordén (in Eriksson and Gustavsson, 2010; 2002) observed only 5.8% dry matter loss of green logging residue bundles after 5½ months in Sweden.

Moisture is the most important factor in dry matter loss, because it directly affects the microbial activity that produces the loss (Ashton, 2008). Studies of switchgrass losses show results ranging from zero for a year's inside protected storage to 13% for unprotected bales stored outside for 6 months (Sanderson et al., 1997). Buckmaster (1992) showed dry matter losses in baled hay under indoor storage ranging from near zero dry matter loss at 12% moisture to around 7% dry matter loss at 25% moisture at baling. Studies of indoor hay storage indicate that, to reduce the risk of severe heat production and fire in the storage stationary source, moisture content of incoming bales should be 25% or less (Buckmaster, 1992).

The type of storage facility (indoor, outdoor unprotected, outdoor protected, etc.) and the underlying foundation (sod, concrete, gravel, etc.) affect storage losses, primarily through protection from increased moisture and differences in aeration (Sanderson et al., 1997). For instance, in a Swedish study on forest residue bales, the 10 month dry matter losses due to biological activity were highest in the outdoor-uncovered stack, which had an average total dry matter loss of 18.5%, while the indoor stacks had a dry matter loss of 14% (Eriksson and Gustavsson, 2010; Jirjis, 2003). For Atlantic Canada conditions, Afzal et al. (2010) estimate 6.6 to 15.6% dry matter losses during 6 months for uncovered birch chip piles of (naturally) dried and fresh forest residue, respectively.

Besides initial moisture content of feedstock, type of storage facility (indoor, outdoor unprotected, outdoor protected, etc.) and use of an appropriate foundation, storage losses also strongly depend on the "packaging" of the feedstock. For instance, losses are higher in chipped material than in loose residue piles, because chipping increases the amount of exposed surfaces on which microbial action can occur and releases soluble contents of plant cells that provide nutrition to microbes (Richardson et al., 2002). Small chips also reduce air flow in piles and prevent heat dissipation and moisture release.

The length of storage time affects losses in combination with moisture and facility type. The decay process starts as soon as 1 week, with the highest dry matter losses in the first weeks of storage (Wihersaari, 2005; for Finland). Indoor storage may result in moisture content going from 25% to around 12% after 2 months or more of storage (Buckmaster, 1992), at which point microbial activity will be minimized.

These studies indicate that, if necessary, it may be possible to develop a representative estimate that could be applied to a given stationary source on the basis of the five variables (type of biomass and particle size, initial moisture content, type of storage, average length of storage period, and regional climate conditions). Emissions will be composed almost entirely of $CO_2$ except in storage facilities where aeration is limited. For example, one study of logging residues stored in laboratory vessels found maximum concentrations of $CO_2$, CO, and $CH_4$ to be 13.8%, 0.16%, and 0.15%, respectively, over a 35-day period (He et al., 2011).

A literature search identified studies that investigate dry matter loss over time for various types of biomass, including those with original research on dry matter loss. Based on 22 studies, a dataset containing 112 cases of reported dry matter loss over time was produced (see Addendum I in Section 6, with case categories including source, origin, region, biomass type, storage type, original moisture content, storage length, and dry matter loss). If data were presented from abroad, the study location was matched by climatic zones with one of the U.S. regions described in Smith and Heath (2002).

Addendum II (see Section 7) expands upon the literature review presented in this Section and in Addendum I, focusing on dry matter losses from storage of woody biogenic materials.

## 4.3. Representative Values for Storage Losses

Analysis of the dataset (see Addendum I in Section 6 below) revealed no clear trends in storage loss by biomass type. Even herbaceous crops that were assumed to produce comparable results across studies (usually baled and pre-dried) did not produce notable and distinguishable trend lines by region or storage type (see Figure G-2). Additional data and statistical analysis could provide more insights. Based on the available data, trend lines describing the storage losses for each of the three storage types (outdoor, uncovered; outdoor, covered; indoor) were produced for one feedstock (woodchips) (see Figure G-3). Because of the limited research on indoor storage losses for woodchips, all indoor storage data were aggregated irrespective of region and biomass type and a "worst-case" scenario was used, assuming rapid storage losses.

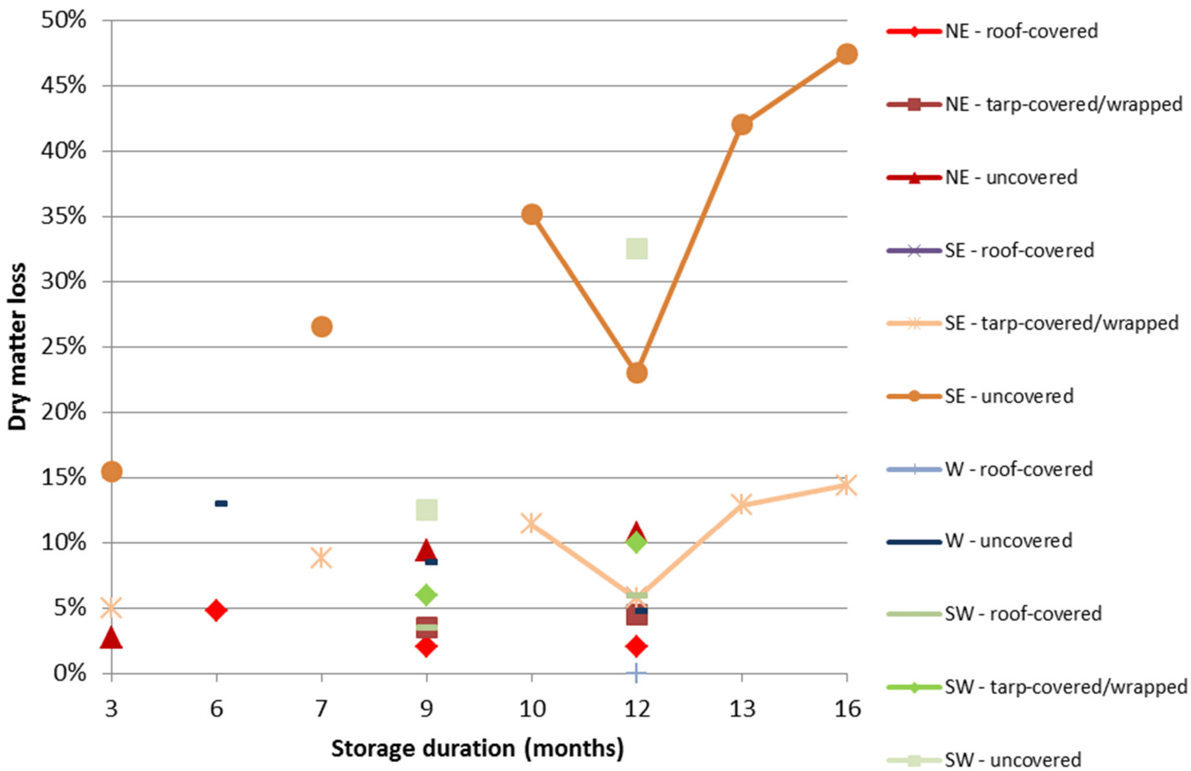


**Figure G-2. Data Points for Herbaceous Crops Storage for Western (W), Southeastern (SE), Northeastern (NE) and Southwestern (SW) Regions. Storage losses are not clearly distinguishable by region or storage type (N = 69).**

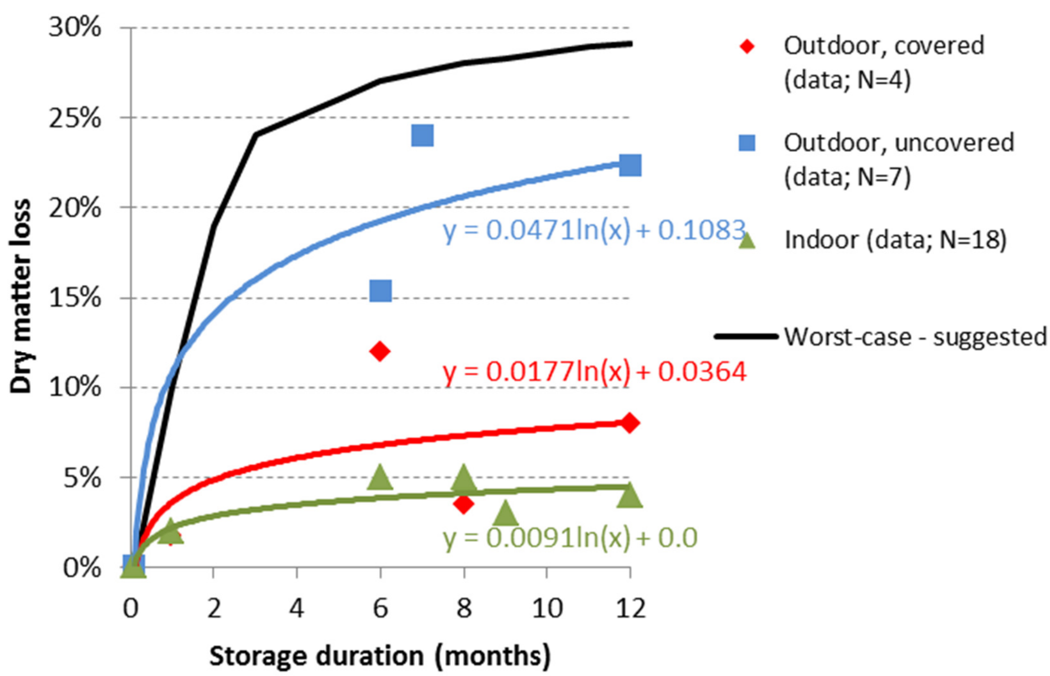


**Figure G-3. Data Points and Trendlines for Storage Losses for Woodchips in the Northeast (N = 29).**

Table G-5 describes the trend lines shown in Figure G-3 and is a first approximation for a storage loss term lookup table for woodchips in the Northeast provided for illustrative purposes only.

**Table G-5. Example Representative Values for Storage Losses for Woodchips in the Northeast (based on Figure G-3).**

| Storage Duration (months) | Outdoor, Uncovered | Outdoor, Covered | Indoor |
|---|---|---|---|
| **1** | 11% | 4% | 2% |
| **2** | 14% | 5% | 3% |
| **3** | 16% | 6% | 3% |
| **4** | 17% | 6% | 4% |
| **5** | 18% | 6% | 4% |
| **6** | 19% | 7% | 4% |
| **7** | 20% | 7% | 4% |
| **8** | 21% | 7% | 4% |
| **9** | 21% | 8% | 4% |
| **10** | 22% | 8% | 4% |
| **11** | 22% | 8% | 4% |
| **12** | 23% | 8% | 5% |

Based on the data identified in the figures and tables above, four options for developing representative *LOSS* values are explored (though other options could be used in applications of the framework):

1. Develop a "maximum" *L* term for each feedstock type by region. This could reflect a "highest possible case" system (see black line in Figure G-3) i.e., outside, uncovered, on-ground, long-term (>6 months) storage (assuming 50% initial moisture, which results in>20% loss ($LOSS_1$ => 0.2).
2. Provide "loss deductions" that could be used to reduce the maximum losses in situations in which stationary sources use improved management (e.g., a system that stores material at a moisture content of 20% indoors on concrete for <1 month could lead to an estimate of a zero loss).
3. Provide for a stationary source-specific loss calculation method that could be used to estimate the *LOSS* term for a particular stationary source (Sale point dry weight (DW) – Process point DW) / Process point DW = *L*).
4. Provide one single representative *LOSS* value (at a broader scale than an individual stationary source, e.g., region) that presents average losses for herbaceous and woody biomass for an average storage time.

The following example discusses development of a regional representative *LOSS* value for illustrative purposes: 10% loss (*LOSS* = 0.1) for woody and herbaceous feedstocks, including stover, is discussed and calculated below using the Northeastern data.

It is assumed that stationary sources consuming woody biomass usually store a 3-month supply uncovered on their premises. Averaging covered and uncovered outdoor storage data for woodchips in the Northeast (Figure G-3), this would equal a loss of 11%.

There are limited data on storage-related data for stover and agricultural residues, but trends for these feedstocks were similar to herbaceous crops (Figure G-2), so both feedstock categories were combined. In terms of storage time for herbaceous feedstocks, this example assumes a 4-month harvest window. It is also assumed that during this harvest window, feedstocks are immediately converted to usable heat without storage losses. An average storage time of 2.7 months for feedstock was calculated based on an assumed 8 months of off-harvest season. This average storage time takes into account that some of the feedstock is burnt immediately after harvest (during the harvest months), while other sections of the feedstock are stored in decreasing quantities for up to 8 months until the next harvest window. The data points for a 3-month outdoor storage (Figure G-2) suggest a loss of 3 to 15% with an average of around 9 to 10% across all regions because no separate regional trends are discernible with current data.

The dataset contained no reports on storage losses for roundwood, and only one case was recorded for wood pellets. Because of this lack of data, this appendix does not provide illustrative examples of storage losses for these feedstock categories.

Potential feedstocks could also be oil seed crops and sugar-based feedstocks such as corn kernels or sweet sorghum. Sugar-based feedstocks can experience rapid storage losses in a matter of days (e.g., Bennett and Anex, 2009; Jasberg et al., 1984), which is usually not measured in dry matter losses and is therefore not compatible with the approach followed in the dataset described above.

## 4.4. Discussion of Other Losses: Transportation and Harvest

### 4.4.1. Transportation

Dry matter lost in transportation is almost wholly due to mechanical losses. Chemical losses, such as the degradation of dry-matter, generally occur during storage and at a much larger timescale than that which occurs during transport (weeks to months of sitting in storage vs. hours to days in transit). Although mechanical losses occur in the loading and unloading of dry matter, these losses are small and are considered more or less as inevitable with few mitigation options available.

Transport losses are often less that 1% (Sanderson et al., 1997) even in the case of long-distance supply chains for woody biomass (Hamelinck et al., 2005). One study found that bale weight changes and biomass loss of switchgrass during handling and transportation from the field resulted in a total dry matter loss of 0.4% (Sanderson et al., 1997). In the case of roundwood, transport losses are generally not expected to occur. As an example, developing a lookup table for transport losses could be done by using research specifying representative values by transport type (truck, vessel, train) and distance (e.g. <50 mi, 50–100 mi, >100 mi). If they are included, transport losses for one crop should be added together to provide an exhaustive view to systematically quantify total losses (Hamelinck, 2005). For instance, if biomass is transported by truck (e.g., 1% loss) followed by train (1% loss), total transport losses could equal 2%. However, due to lack of data, no such illustrative example is included in this appendix.

#### 4.4.2. Harvest

Harvest losses are not included in the example calculations in this appendix, because such losses are not often measured or included in the delivered product weight and thus not typically calculated in yield computations. However, harvest losses could be significant in the case of specific feedstocks or for vertically integrated stationary source operations that oversee farm/forest operations. Harvesting efficiencies for crop residues using conventional multipass harvesting systems are relatively low, with only one-third to two-thirds of the available crop residues actually collected (Hess et al., 2009). For instance, Monti et al. (2009) note that "Unlike storage, major biomass losses[6] occurred during harvest [of hay], either due to the biomass not picked-up by the baler machine (up to 17%), and uncut biomass by the mower machine (up to 29%). Overall, of the potentially harvestable biomass, only 64% was actually baled." However, Hess et al. (2009) noted that for most farmers, it is general agricultural practice to intentionally leave a certain amount of crop residue on the landscape to ensure soil health. The amount of crop residue left behind is somewhere between 40% and 70%, with an average amount of biomass left onsite of 60% (Hess et al., 2009). Nevertheless, these harvesting values are for specific crop yields and are highly variant in nature; more data would be required to estimate such losses.

### 4.5. Representative Factor Discussion

There may be cases where information on supply chain losses is unavailable between the feedstock production site (e.g., farm, forest) and the feedstock conversion facility. For example, if an intermediate supplier or primary processing facility supplies feedstock (e.g., woody mill residues) to the stationary source, there may be little information about the transportation and storage losses incurred between the landscape where the feedstock is grown and the stationary source. Representative values for losses may be determined based on a review of literature as described in Section 4.3 above. Alternatively, a representative factor derived from the literature may be assigned for a particular feedstock type in the absence of refined information.

## 5. References

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# 6. Addendum I. Literature Reviewed for Storage and Processing at Stationary Sources Facilities

**Table G-6. Dataset Sources and Cases by Source for Storage and Processing at Stationary Sources.**

| Source | No of cases |
|---|---|
| Afzal, M., A. Bedane, S. Sokhansanj, and W. Mahmood. 2010. Storage of comminuted and uncomminuted forest biomass and its effect on fuel quality. *BioResources*, 5(1):55-69. | 6 |
| Buckmaster, D.R. 1992. *Indoor hay storage: Dry matter loss and quality changes*, Fact Sheet PSU/92. State College, PA: Agricultural and Biological Engineering Department, Pennsylvania State University. | 2 |
| Collins, M., D. Dirtsch, J.C. Henning, L.W. Turner, S. Isaacs, and G.D. Lacefield 1997. Round Bale Hay Storage in Kentucky. AGR-171, Kentucky Cooperative Extension Service. | 8 |
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| Hamelinck C.N., Suurs, R.A.A., and Faaij, A.P.C. 2005. International bioenergy transport costs and energy balance. Biomass and Bioenergy 29(2): 114–134. | 2 |
| He, X., Lau, A.K., Sokhansanj, S., Xiaotao, C.J., Bi, X.T., and Melin, S. 2011. Dry matter losses in combination with gaseous emissions during the storage of forest residues. Fuel 95:662–664. | 1 |
| Hess, J.R., Kenney, K.L., Park Ovard, L., Searcy, E.M., Wright, C.T. 2009 Uniform-format solid feedstock supply system: a commodity-scale design to produce an infrastructure compatible bulk solid from lignocellulosic biomass. Idaho National Laboratory. | 22 |
| Hess, J.R., Kenney, K.L., Wright, C.T., Perlack, R., Turhollow, A. 2009. Corn stover availability for biomass conversion: Situation analysis. Cellulose 16 (4): 599-619 | 6 |
| Hudson, J. B., Mitchell, C. P., Gardner, D. and Storry, P. 1988. A comparative study on storage and drying of chips and chunkwood in the UK. In Proceedings of the lEA/BE Conference Task Ill/Activity 6&7: Production, Storage and Utilization of Wood Fuels, Vol. H, ed Danielsson, B.O., Dept of Operational Efficiency, Swed. University of Agricultural Science, Garpenberg, Sweden. | 4 |
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| Jirjis, R. 2003. Storage of forest residues in bales. Department of Bioenergy, Swedish University of Agricultural Sciences, Uppsala. p. 23. | 2 |
| Jirjis, R., and Nordén, B. 2002. Lagring av buntat skogsbränsle – små substansförluster, inget arbetsmiljöproblem [Stockpiling of composite residue logs (CRLs) – small biomass losses and no health problems]. Skogforsk Resultat 12:1–2. | 1 |
| Jirjis, R., Pari, L., Sissot, F. 2008. Storage of poplar wood chips in Northern Italy. Proceedings of the 8th World Bioenergy Congress, 27-29 Maggio Jonkoping, Sweden: 85-89. | 1 |
| Monti, A., Fazio, S., Venturi, G. 2009. The discrepancy between plot and field yields: Harvest and storage losses of switchgrass. *Biomass and Bioenergy* 33(5): 841-847. | 1 |
| Mooney, D.F., Larson, J.A., English, B.C., Tyler, D.D. 2012. Effect of dry matter loss on profitability of outdoor storage of switchgrass. *Biomass and Bioenergy* 44: 33-41 | 20 |
| Nilsson, T. 1987. Comparison of storages of chunkwood and fuel chips. Report no. 192, Dept of For. Prod, Swed. Univ. Agric. Sci., Uppsala, Sweeden. (In Swedish, English summary.) | 3 |
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| Sanderson, M.A., Egg, R.P., Wiselogel, A.E. 1997. Biomass losses during harvest and storage of switchgrass. *Biomass and Bioenergy* 12(2): 107-114 | 4 |

| Source | No of cases |
|---|---|
| Shinners, K.J., B.N. Binversie, R.E. Muck, and P.J. Weimer. 2007. Comparison of Wet and Dry Corn Stover Harvest and Storage. Biomass & Bioenergy, 31:211-221. | 7 |
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# 7. Addendum II. Literature Review of Dry Matter Losses from Woody Biogenic Material Storage

## 7.1. Introduction

This addendum expands upon the literature review presented above in Section 4 on storage dry matter losses for woody biogenic material. Woody biomass contains roughly 50% (by weight) water when harvested. The water content of biomass can fluctuate between harvest and processing or use for bioenergy as a result of drying or reabsorption of water from precipitation. Dry matter loss (DML) is of interest because dry matter excludes the weight of water in the biomass and water does not contain biogenic carbon. The biogenic carbon is part of the dry weight fraction of the biomass (e.g., woody biomass dry matter contains approximately 50% carbon). For mechanical losses due to feedstock handling, there is no chemical change in the feedstock so it is reasonable to conclude that the proportion of carbon in material lost due to mechanical processes is the same as the proportion of carbon in the bulk feedstock. For chemical losses due to degradation of dry matter in storage, the DML can be assumed to be reasonably proportional to the weight percentage of carbon loss from the feedstock (e.g., a 1% loss in dry matter can be assumed to represent a 1% loss in feedstock carbon).[7,8]

## 7.2. Storage Losses

Several factors, including storage method and time and related parameters such as material moisture content, temperature and particle size influence DML from woody biomass (He et al., 2012). According to Forest Products Laboratory (FPL) (FPL, 2010), most decay can progress rapidly at temperatures that favor growth of plant life in general. For the most part, decay is relatively slow at temperatures below 10°C (50°F) and above 35°C (95°F). Decay essentially ceases when the temperature drops as low as 2°C (35°F) or rises as high as 38°C (100°F).

Significant decay can occur only when the moisture content of wood is above the fiber saturation point (average 30% moisture). Only when previously dried wood is contacted by water in the form of rain or condensation or is in contact with wet ground will the fiber saturation point be reached. By itself, the water vapor in humid air will not wet wood sufficiently to support significant decay, but it will permit development of some mold fungi (FPL, 2010). Springer (1979) explained that

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[7] One Italian study (Barontini et al., 2014) of poplar chips from stem wood and crowns representative of logging residue found that the proportion of carbon in the feedstock before and after 6 months of storage was similar although the overall dry matter of wood decreased. For crowns, the fraction of carbon in the dry matter varied from 47.7 to 46.9% before and after storage, respectively. For stem wood, the fraction of carbon in the dry matter varied from 47.2 to 47.8% before and after storage, respectively.

[8] In addition to observing a loss of dry matter over 1 year of storage, Nurmi (1999) found only a slight variation in the carbon fraction in the dry matter of comminuted Norway spruce logging residue feedstock (i.e., an increase of the carbon fraction in the dry matter from 50% to 51% before and after storage, respectively).

wood dried below its fiber saturation point (20% to 24% moisture) is not subject to bacterial and fungal attack.

Anheller (2009) noted that DML becomes higher when woody biomass is stored for longer periods and in larger piles. The shape of the piles influences the heat development during storage of biomass. To minimize biomass losses, Anheller (2009) recommended keeping the biomass as dry as possible before storage, keeping as large a particle size as possible, preferably under roof, uncommuted and uncompacted, as long as possible before combustion. Anheller (2009) suggested that the most advantageous way to store wood fuels is as uncomminuted fuels, such as whole tree logs, reducing the size of fuel wood to chips as close as possible to the time of use, to shorten the time of storage as chips and decrease degradation losses associated with microbial activity. Similarly, Wilkerson et al. (2008) cited studies recommending that comminuted material be used as quickly as possible after grinding to minimize DML.

### 7.2.1. Forest Biomass Storage Methods

There are a variety of forest biomass storage methods depending on the source and intended use for the material. Forest biomass may be stored as whole logs, in wood chip piles, as logging residue prior to chipping (piled or in bales), as bark, or as wood pellets. Whole logs may be processed into forest products such as lumber or veneer or chipped for use in pulp, wood product, or bioenergy production. Bark is often stored separately from wood chips, especially at mills with different uses for bark (e.g., as fuel) and chips (e.g., forest product raw material).

#### Whole Logs

Wood can be too wet for decay as well as too dry. If the wood is water-soaked, the supply of air to the interior of the wood may not be adequate to support development of typical decay fungi. For this reason, logs stored in a pond or under a suitable system of water sprays are not subject to decay by typical wood-decay fungi. For logs, rapid conversion into lumber or storage in water or under a water spray is used to avoid fungal damage. (FPL, 2010)

FPL (2004) explained that, today, softwood logs decked in a log yard are typically protected by water sprinkling during warm weather. Sprinkling provides an effective method of reducing checking, sapwood stain, and decay. Sprinkling will not protect against insect attack, although it tends to be more effective than dry land storage. To be effective, the ends of logs and exposed wood must be kept wet continuously during the entire storage period. Sprinkling reduces available oxygen, thereby deterring sapwood staining and decay. Pond storage (e.g., in mill ponds, lakes, rivers, and saltwater estuaries), although once common practice, is seldom used in the continental U.S. today (FPL, 2004).

Sprinkling of logs has been practiced for decades. In 1959, Wright et al. of the USDA Forest Service Station (Portland, Oregon) discussed sprinkling to prevent decay in decked Western hemlock logs. Wright et al. (1959) noted that wood containing less than 20% moisture (dry-weight basis) does not decay, and for decay to progress fairly rapidly, a moisture content of 30% or more is required. However, oxygen is also necessary for the growth of wood-rotting fungi, and decay will not develop when complete saturation with water eliminates air from the wood. Based on experimental research, Wright et al. (1959) concluded that:

- Both green and old western hemlock logs can be stored in un-sprinkled decks for 1 year without appreciable additional decay;
- Green logs can be decked for as long as 2 years without serious additional loss, but old logs containing decay when decked suffer considerable loss; and
- Constant sprinkling with water during the warmer months of the year, to keep ends and faces of logs wet at all times, effectively inhibits the development of additional decay in old logs.

### Chip or Bark Piles

Outside chip storage at pulp mills has been practiced since the 1950's in order to maintain large inventories without the use of bins or silos. In Smook (2002), it is recognized that losses of 1% wood substance per month are typical of outside storage due to a combination of respiration, chemical reactions, and micro-organism activity. Smook (2002) reports that wood losses and degradation during outside chip storage can be minimized with effective chip pile management such as ground barriers (e.g., concrete, asphalt), fines screening, avoiding contamination of sound wood with decayed wood, and minimizing storage time. While bark may be a contaminant in the pulping process, bark is less problematic for chips destined for use as boiler fuel (Smook, 2002).

Mill residues may be stored in piles (indoors or outdoors) or silos. Mills and other wood-using facilities keep chip or sawdust piles on-site or at nearby facilities when supply is low. Woody biomass is reduced in size in the forest and then transported for storage, or it is transported, reduced in size at the mill, and then stored. The resulting material, usually chips, is stored outside in large piles and under cover in large silos or bins. Chips stored in bins are typically used within several hours or days while silos are used for longer-term storage needs. Silos and bins protect against contamination while at the same time allowing for uniform feeding and metering of the material. While storing, comminuted biomass makes handling and transport relatively easy. If not managed carefully, the biomass will succumb to DML and in some cases self-ignition. High temperatures and acetic acid odor are signs that a chip pile is in danger of DML and self-igniting. Additionally, chip piles with excessive mold and fungi growth can lead to health risks for humans (Hubbard et al., 2007).

Dry matter loss of forest biomass, which includes the degradation of lignin, cellulose, and hemicellulose, occurs when woody biomass, in any form, is not used immediately after harvesting and has to be stored within a facility. The degree to which DML occurs depends largely on the material's moisture content. Woody biomass having high moisture content is more susceptible to colonization by fungi and mold and at a faster rate. These microorganisms, via metabolic activity, generate heat, which in turn accelerates oxidation, moisture adsorption, hydrolysis, pyrolysis, and other chemical processes resulting in DML (Hubbard et al., 2007). Fungal activity is also responsible for the fermentation of the fractionated holocellulose. However, when this happens in the presence of aerobic bacteria and oxygen, they can change the ethanol produced by natural degradation processes into acetic acid or vinegar. The degradation of biomass through hydrolysis, fermentation, and oxidation that results in the production of acetic acid lowers the average pH of the moisture in the wood chips. This decrease in pH can lead to rapid, premature decay of the piled material. For example, in a chip and bark pile during six months of storage, the initial pH dropped from near-neutral to neutral (5 to 7) to an average of 4 (Slaven et al., 2011).

As noted above, biomass moisture can lead to an increase in temperature in piled biomass, which in turn leads to DML. As shown in Figure G-4, Wihersaari (2005) used temperature to illustrate the pattern of DML for dried and fresh (green) feedstock. Wihersaari (2005) and others explain how pile temperature relates to DML as follows:

- Dry matter that is intentionally dried through a mechanical process and then stored undergoes an initial increase in temperature, which in turn increases microbial activity within the stored pile and acts as a catalyst to material degradation. However, this increase in temperature is short-lived since after approximately one week, microbial colonies begin to die out due to their residence in an inhospitable, low-moisture environment. The death of these microbes in turn reduces temperature within the pile and provides a decline in the rate of decay of the stored material (Wihersaari 2005).
- Stored piles of fresh forest matter generally do not reach internal temperatures as high as piled dry forest materials, but they maintain a higher temperature (above 40$^{o}$C [104$^{o}$F]) much longer than dried forest materials. As Figure G-4 shows, fresh forest matter tends to maintain a temperature above 40$^{o}$C (104$^{o}$F) for longer than 27 weeks. While fresh forest materials stored outdoors will lose less material due to mechanical losses such as wind, due to their weight, storing forest materials without drying them first creates problems. Storing fresh forest materials in large piles increases microbial activity by providing a hospitable environment for growth, which leads to rapid material decay. These fresh forest material piles, with extended periods of high internal temperatures, are also more susceptible to spontaneous combustion (Wihersaari, 2005; Slaven et al., 2011; Hubbard et al., 2007).

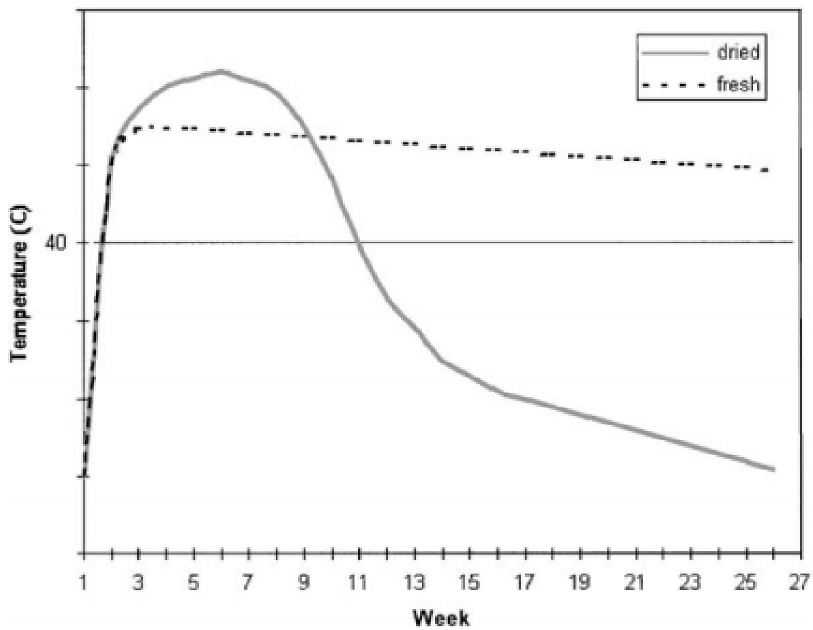


**Figure G-4. Simplified Principle of Degradation Behavior (Wihersaari, 2005)**

In his study, Wihersaari (2005) found that when the average chip size in a pile of dried forest matter was 30 millimeters (mm), the temperature rose to 40 to 50°C (104 to 122°F); but when the average chip size was 70 mm, the temperature did not rise above 30°C (86°F). Wihersaari (2005) explained that the biological activity producing heat takes place on the surface of the chips; the smaller the chip size, the larger the surface area per volume and consequentially the higher the biological activity leading to higher temperatures. Hubbard et al. (2007) explained that the small particle size gained by chipping restricts air flow and prevents heat dissipation, while chipping releases the soluble contents of plant cells providing microbes with nutrients.

Anheller (2009) also explained that heat development can occur in in chip piles if the moisture content is high enough for microbial growth, usually >20% for wood fuel. In fresh wood chips, respiration heat from the living cells contributes to initial heat development and the chipping process releases soluble sugar from the wood, which together with heat, moisture and oxygen can create a favorable environment for microbes. By chipping the material, the area where the microbes can attack increases. In a pile of chipped material, the air movement is also more limited because of the smaller material, which prevents heat dissipation and causes heat accumulation and thereby increases degradation losses. In large piles, the material can additionally become more compact because of weight, which further amplifies the abovementioned factors (Anheller 2009).

Similarly, Janze (2011) concluded that particle size within a biomass pile effects moisture absorption, heat build-up, heat dissipation and DML. Piles containing a large amount of fines absorb greater amounts of water, generally heat up faster due to greater microbial action, and restrict air movement through the pile, thereby limiting heat dissipation; all of this leading to increased DML and possibly spontaneous combustion. Conversely, piles consisting of large wood chunks heat up more slowly, permit better air flow due to the large voids, dissipate the heat faster due to better air circulation, have lower rates of microbial action and lower DML.

Janze (2011) recommended that for long storage periods that fuel be stored in fairly large particles and only re-processed to the final size shortly before usage in the boiler. Janze (2011) further recommended that radically different types of material such as clean wood chips, short rotation coppice and forest debris should not be stored in the same pile, as they decompose at different rates, which can lead to spontaneous combustion. Such materials should be stored separately and blended just prior to transport to the boiler (Janze, 2011).

### Logging Residue Prior to Chipping

Logging residues may be left in small piles at the feedstock production site, chipped after some time, or compacted into bundles. Chipping is the method most commonly used in North America (Afzal et al., 2010). Hubbard et al. (2007) explained that there are several advantages to storing unconsolidated woody biomass immediately after harvest. When stored unconsolidated in mounded piles of moderate size, leaves and needles can fall, reducing the material's ash content. Moreover, when woody biomass is stored in smaller piles, drying occurs (i.e., moisture escapes through leaves and other open wood surfaces). This process lowers the moisture content and increases the heating value. Drying occurs when biomass is stored in windrows as well, but it is not as efficient as small piles because foliage is not allowed to drop. Additionally, when stored on the harvesting site, vital nutrients are released back into the soil. The major disadvantages to storing woody biomass immediately after harvest on-site are (1) the need for detailed inventory tracking, (2) the cost of forgoing reforestation until the piles of biomass have been removed, and (3) the cost and time-sensitivity of having a contractor return to the site to collect, pre-process, and transport the material to the wood-using facility (Hubbard et al., 2007).

Woody biomass can be bundled and stored under cover to gain advantages (ease of handling and transport) that come along with storing chipped material. At the same time this approach protects the material from the disadvantages that come along with chipped material: DML, moisture retention, heat generation, and health hazards (Hubbard et al., 2007).

### Logging Residue Chips

Wilkerson et al. (2008) noted that whole tree chips containing a large proportion of leaves and bark have a propensity to self-heat and can possibly cause chip pile fires. Wet chip pile heating can be mitigated by keeping pile heights under 9 meters (m) (30 ft) and/or limiting the amount of time the chips are piled to less than 10 days. Wilkerson et al. (2008) suggested that whole tree chips for energy that must be stored for more than 10 days may need some form of treatment to avoid degradation or spontaneous combustion. Drying to less than 20% moisture is the only viable option for safe long term storage of whole wood chips, but drying results in added expense, and dried fuel would need to be protected from rain with shelter or covering.

Earlier, Springer (1979) also explained that whole-tree chips deteriorate more rapidly than clean, debarked chips and present a greater hazard for spontaneous ignition when stored in outdoor piles. Springer (1979) noted that whole-tree chips can be stored for only a short period of time to prevent ignition, that frequently rotating storage piles increases handling costs, and drying prevents deterioration and heating if the dried chips are stored under cover. According to Springer (1979), the costs of drying can be recovered if the chips are burned for fuel. Springer also referenced studies conducted in Norway in the last 1970's that concluded that bark and foliage increase the rate of chip deterioration, with chip weight loss during storage following the order: clean debarked chips < whole-tree chips < bark < foliage (Springer, 1979).

### Wood Pellets

Wood used to form woody biomass pellets is dried prior to the pelletizing process. Drying temperatures from 100 to 400°C (212 to 750°F) are used to reduce moisture content to less than 10% (Yazdanpanah et al., 2014). The average moisture content of commercially produced pellets (in Sweden) is about 10% to 12% (Lehtikangas, 2001). The Pellet Fuels Institute (PFI) standards for pellets produced in the U.S. range from less than 8% to 10% moisture (PFI, 2011).

Wood pellets have flow characteristics and are well suited for storage in silos (Janze, 2011). Pellets must be kept dry during storage (Janze, 2011; NEBTWG, 2012).

#### 7.2.1. Storage Time and Challenges in Tracking Biomass Inventory

Janze (2011) explained that the amount of woody biomass storage required depends upon:

- Minimum fuel storage; how much storage the plant's financiers, regulators, clients and/or insurers require as a minimum to ensure continuous operation;
- Known fuel delivery interruptions; the length of periods when fuel delivery can be anticipated to be interrupted, say for long weekends, or when supplier's mills are shutdown;
- Reliability of fuel delivery; there must be enough fuel on hand to cover likely transportation delays;
- Contingency supplies, to cover periods when forests are inaccessible for fire season, the wet season or during spring break-up;
- Fiber supply contractual requirements; the ability to continue to stockpile fuel when the power plant is shut down for annual maintenance; and
- Often the space available dictates the size and shape of the storage pile and the maximum amount of fuel that can be stored.

According to Janze (2011), typically, biomass-fired power plants [in the European Union] will stockpile a minimum of 20 to 30 days of fuel, but many will store for 60 days or more (i.e., 1 to 2 months storage). Wilkerson et al. (2008) noted that some areas of the U.S. can harvest woody biomass year around, and long term storage is not necessary. However, in other areas a 1 to 6 month supply of biomass may be required.

Hubbard et al. (2007) reported that chip-pile storage is most common type of storage in the southeastern U.S. and recommended that owners shorten the storage time of chipped material to minimize the risk of microbial decomposition which will in turn decrease DML. Hubbard et al. (2007) indicated that the ideal storage period typically varies from 2 to 6 weeks (0.5 to 1.5 months) as determined by each facilities wood supply situation.

Smook (2002) indicated that, because chip deterioration is largely a function of storage time, the most effective way to minimize losses is to minimize storage time (e.g., use first-in, first-out practices). Referring to pulp wood chips, Smook (2002) noted that optimum chip handling depends on use for the chip. For example, 2-month storage of chips used for sulfite pulping reduces extractives problematic in the sulfite pulping process. However, if maximum recovery of extractives-based pulping byproducts (e.g., tall oil, turpentine) is desired then fresh chips should bypass storage in order to maximize byproduct yield.

Janze (2011) explained some of the challenges with tracking biomass pile inventory, noting that problems stem from the variability in biomass physical properties, including different species, moisture contents, and bulk densities, and varying amounts of compaction and dry fiber loss. Additionally, using multiple inventory tracking measures, including "green" or "bone dry" mass, solid wood or bulk densities, and solid wood volumes or bulk volumes, lead to inventory errors when converting from one measure to another and back again (Janze 2011).

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# Appendix H: Illustrative Biogenic Landscape Attributes Using a Retrospective Reference Point Baseline

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# 1. Introduction

This appendix describes potential methods for calculating regional default values for the landscape biogenic attributes (*GROW*, *AVOIDEMIT*, and *SITETNC*) associated with sample feedstocks and regions using a retrospective reference point baseline. The calculations presented in this appendix are meant for illustration only, as proof-of-concept exercises, to show how these values might be calculated. They are not meant as final values for use in a particular stationary source program for any particular feedstock.

This appendix uses the retrospective reference point baseline approach to produce illustrative equation term values for three of the landscape biogenic attribute terms (*GROW*, *AVOIDEMIT*, and *SITETNC*) from the biogenic assessment factor equation in the main report (Part 2):

$$\boldsymbol{NBE} = (\boldsymbol{PGE})(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P}) \qquad \textbf{(EQ. H.1)}$$

Although leakage (represented by the *LEAK* term) is also a landscape biogenic attribute, this term is not calculated in the proof-of-concept term calculations presented here. Appendix E provides further discussion on leakage.

The framework defines the biogenic landscape attribute equation terms addressed in this appendix as:

- *GROW*: *GROW* represents the ratio of net feedstock growth on the biogenic feedstock production landscape relative to landscape biogenic carbon removals. This term only includes biogenic carbon within the feedstock carbon pool.

- *AVOIDEMIT*: *AVOIDEMIT* represents the ratio of avoided biogenic emissions that would have occurred on the feedstock production landscape without biogenic feedstock removal (such as removal of corn stover and logging residues) to landscape biogenic carbon removals.
- *SITETNC*: *SITETNC* represents the ratio of the estimated total net change in non-feedstock carbon pools on the feedstock production site due to land use management or land use management changes associated with feedstock production to landscape biogenic carbon removals.

Illustrative equation term values are generated using the retrospective reference point baseline approach in the context of three specific feedstock/region combinations:

- Roundwood in the Southeast (SE);
- Logging Residues in the Pacific Northwest (PNW); and
- Corn Stover in the Corn Belt (CB).

# 2. Illustrative Method for Developing Regional Default Values for Biogenic Attributes: Southeast Roundwood

This section explains the method by which regional default values for *GROW*, *AVOIDEMIT*, and *SITETNC* were developed for the illustrative application to Southeast Roundwood. In this context, roundwood is defined as the portion of tree biomass that would be defined as "merchantable" according to existing forest inventory definitions. This includes trees of commercial species, with good form (e.g., not hollow or "cull"), large enough to be harvested, and includes the main bole or stem but not branches or tops.

## 2.1. *GROW*

In general, the *GROW* term represents net feedstock growth on the biogenic feedstock production landscape. Estimating a value for *GROW* at a regional level using the retrospective reference point baseline could use an assessment of recent forest growth and harvest in the feedstock's source region. Therefore, in this specific baseline context, *GROW* can be represented as the ratio of removals less growth over removals of roundwood in the source region over the most recent forest inventory cycle:

$$\boldsymbol{GROW_{Roundwood}} = \frac{\boldsymbol{REMOVALS - GROWTH}}{\boldsymbol{REMOVALS}} \qquad \textbf{(EQ. H.2)}$$

For this illustrative retrospective reference point baseline approach application, computation of the *GROW* term ($(R{-}G)/R$) is based on forest inventory data collected by USDA Forest Service in the Forest Inventory and Analysis (FIA) program. The FIA data representing the most recently completed inventory for a region are used. The FIA program defines several types of growth and removals values. Growth is measured in FIA by comparing tree measurements from specific plots in the current inventory cycle with measurements from previous cycles, enabling a tree-by-tree estimate within remeasured plots. In addition, as plots that were previously not forested become forested, any tree volume on these newly forested plots represents growth, or additions to the forest stocks. FIA defines "gross growth" as the sum of growth across all trees on all plots. FIA defines "net growth" as the gross growth minus the volume of trees that die during a remeasurement period, termed mortality:

$$\textit{Net Growth} = \textit{Gross Growth} - \textit{Mortality} \qquad \textbf{(EQ. H.3)}$$

Standard FIA reports include net growth and mortality, but not gross growth. As this framework application is meant to assess the feedstock carbon (C) stock on the landscape in the *GROW* term, and thus includes standing dead trees in the estimate of *GROW*. Thus the mortality is added back to the reported net growth to obtain an estimate of gross growth for use in the calculation of *GROW*.

Removals are estimated by FIA when remeasured plots reveal the absence of trees that were measured in a prior inventory cycle. Removals may include wood biomass removed from forests during normal harvest cycles and during conversion of forest to some other land use. Both components are important in evaluating the balance of growth and removals in a region to assess the drivers of biomass removal and fluxes between terrestrial and atmospheric carbon pools.

### 2.1.1. Data

The FIA program data is utilized as it represents forest inventory data across all ownerships and regions across most of the United States. The FIA program measures forest plots in different states annually, such that a portion of field plots (termed a "panel") are measured in each state each year. For example, in the east (easternmost five regions), there are five panels so that it takes 5 years for all plots in a state to be remeasured. The length of time it takes to remeasure all panels in a state is called the inventory cycle. Inventory cycles in the west are longer than in the east: up to 10 or 15 years may be required to measure all panels in a western state's inventory.[1] All FIA data (except precise locations of field plots and specifics about ownership of each plot) are available to the public via the FIA web site: http://fia.fs.fed.us. For this example, data were downloaded in Access database format for each state in the Southeast. This analysis used data from the most recently completed inventory for each state in the region.

### 2.1.2. Forested Land Designation Used

In some policy-specific framework applications, it may be appropriate to use a specific forested land designation, such as all forestland, all timberland, or working forest. In a situation where a "working forest" designation is appropriate, FIA data can be screened to develop estimates only for the forest resource in each region that is defined as the "working forest." The approach and definitions of working forest used in this application, as well the implications of choosing alternate proof-of-concept, land-based designations—e.g., all forest lands, all timberlands, private forest

[1] FIA data are now collected in a nationally consistent manner, although the program is still in transition to this format and many western states do not yet have sufficient data collected under this program to enable computation of growth and removals. For example, remeasured data are currently not available for many states in the west (the Intermountain, Pacific Northwest, and Pacific Southwest regions). Specifically, California, Oregon, and Washington started collecting the nationally consistent data later than other states, and USDA Forest Service will only release data once a sufficient number of panels have been measured to allow for reliable estimates.

lands, public forest lands, working timberlands, private working timberlands—are described below.

#### 2.1.2.1. Defining the Working Forest

The concept of a working forest recognizes that portions of the forest resource within a region are unlikely to be used for feedstock production (Buchholz et al., 2011). Examples of such forest resource areas include protected forest areas, areas not conducive to harvest due to physical conditions (e.g., inoperable soils or steep slopes), areas subject to regulatory restrictions on harvest (e.g., elevation limits in the Northeast), and areas where harvest is not economically feasible (e.g., large distance to transportation networks).

Because harvest of forest resources for biogenic feedstocks is unlikely for certain forest conditions, those areas are excluded from consideration in the *GROW* term in this specific illustrative application. If the growth/removals balance used to calculate values for the *GROW* term were applied to the entire forest landscape within a region, and not to the working forest only, then growth (and related $CO_2$ capture) in areas protected from harvest or not viable for harvest would be included along with removals (and related $CO_2$ emissions) from the working forest. This could mask the actual growth/removals dynamics related to the use of forest biogenic feedstocks, and the related carbon cycle effects on the landscape of such use. For application of this framework at the regional scale, the goal is to identify the biogenic carbon cycle impacts related to stationary sources using biogenic feedstocks. Clearly defining the working forest, i.e., that portion of the landscape from which biogenic feedstocks are most likely produced, is an important first step.

There is an active debate about exactly what constitutes the working forest land base (i.e., Alig et al., 2002). Some fraction of the land base is "reserved" by legal limits on logging, and there is clearly a significant fraction of the remaining forest land that is not available for harvest because of a wide range of biological, physical, legal, economic, and social concerns (Buchholz et al., 2010; Butler, 2008). These limits on the availability of working forest land are difficult to quantify and may vary over time. For example, the increasing "parcelization" of forest land (i.e., subdivision into smaller ownerships) is generally assumed to reduce the land available for harvest because harvest operations are impractical on very small landholdings.

To stratify by working forest for this analysis, the first step is the selection of criteria (parameters) that can be used to define the working and non-working forest, and to set thresholds for those criteria. For example, if it is unlikely that harvest occurs on slopes exceeding 50%, then a slope threshold of less than 50% can be applied to a "slope" parameter to form a part of the working forest definition.

For the illustrative calculations for the *GROW* term below, working forest was defined using a set of physiographic, location, and other factors determined through the plot characteristics in the FIA dataset from the USDA Forest Service. For proof-of-concept, these national criteria nationwide are applied, as working forest can be defined using criteria that may differ by state and/or region,[2] using limits set by local expertise, or using analysis of FIA plot characteristics that have had harvests recorded.

To obtain estimates of forest biomass in the working forest, the first step is to "screen" or filter FIA plots to determine which plots fall within definitions of the working forest. Then, FIA data are summarized for the screened subset of all forest plots. In this example, working forest is defined by five criteria, following Butler et al. (2010):

(1) Access: Areas greater than a mile from improved roads are considered too costly to harvest;
(2) Physiographic condition: Hydric soils are indicative of wetland conditions and are often not suitable for operation of harvesting equipment, so sites classified as hydric physiographic conditions are excluded from the working forest;
(3) Productivity: Sites with very low forest productivity are usually not suitable for biomass production and are thus not typically used for feedstock production; these are excluded from the working forest;
(4) Harvest restrictions: Sites where timber harvest is legally restricted (for example, some national parks and wildlife refuges) are excluded from the working forest; and
(5) Steep slopes: Timber harvest is typically constrained on steep slopes. Thus, sites with slope greater than 50% are excluded from the working forest.

The working forest was defined here by screening out FIA *conditions* (portions of plots) not meeting these specified criteria for working forest. Plots that do not match the definition of working forest were removed from the analysis dataset for each year. The remainder of the plots represents the working forest, and these were used to develop the carbon stock estimates for calculating values for the *GROW* term).

Table H-1 shows the proportions of overall forest area and aboveground biomass that would be included as working forest for each region, based on the five screening criteria described above.

**Table H-1. Proportions of Overall Forest Area and Biomass that Would Be Included as "Working Forest" for the Southeast and South Central RPA Regions (2010 FIA Data Using FIA Database (FIADB) (version 5; Woudenberg et al., 2010) Query Tools to Screen for Working Forest).**

| Region | Percent of Forest Area | Percent of Biomass |
|---|---|---|
| **Southeast** | 82.1% | 80.4% |
| **South Central** | 62.4% | 77.7% |

FIA reports focus on specific variables with important meanings. First, they report on "forestland" and "timberland." Timberland is a subset of forestland that is not specifically reserved from timber production and meets a minimum productivity threshold. For maximum flexibility in application, the analysis herein is based on reports using "forestland." Second, volumes reported may be for "all

[2] For example, some states have harvesting restrictions that apply to certain elevations, slopes, or proximity to water that may not apply in other states.

live" trees or "growing stock" trees. Growing stock trees are limited to commercial species meeting specified standards of size, quality, and vigor. Because these default criteria are oriented towards traditional forest products and not biomass harvest, the analysis herein expanded the analysis to use "all live" tree reports. To include standing dead trees in the estimates, mortality is added to net growth to obtain gross growth; if standard net growth estimates had been used, standing dead trees would have been excluded from the analysis.

Standard FIA reports for growth, mortality, and removals use units of cubic feet. Conversion to metric tons of $CO_2e$ involves multiplying by a constant conversion factor if metric tons $CO_2e$ are the units being used. However, if landscape attribute terms are presented as unitless ratios in a specific application of the framework, it is not necessary to convert to metric tons $CO_2e$ (and results can be left in the original units of cubic feet).

### 2.1.3. Results

The methodology used for the analyses here began with identification of the time period for analysis, which was the inventory cycle ending in 2010. Next, the extent of the working forest was identified. Then, using the FIA Database (FIADB) (version 5; Woudenberg et al., 2010) query tools, the following queries were run for working forests in each state for each year:

- "Net growth of all live on forestland: cu. ft/year;"
- "Mortality of all live on forestland: cu. ft/year;" and
- "Removals of all live on forestland: cu. ft/year."

The annual net growth and mortality estimates were added to compute gross growth.[3] Estimates for each state in a region are summed. Then, *GROW* is computed as the ratio of removals less growth over removals.

**Table H-2. *GROW* Term for the Southeast Roundwood Example (2010 FIA Data, Based on Working Forests).**

| Region | Gross Growth (million cu. ft/yr) | Removals (million cu. ft/yr) | *GROW* |
|---|---|---|---|
| **Southeast** | 7,603.5 | 4,379.7 | −0.74 |

[3] Note that removals from plots that were converted from working forest at their previous measurement to non-forest as of the most recent measurement were not included in this analysis. This occurred because the screening for working forest is based on the area of the working forest at the most recent remeasurement. For example, consider a plot that is part of the working forest at the prior inventory measurement. The plot is then harvested as part of a conversion to a non-forest type (e.g., development). Because the plot is not part of the working forest at the most recent remeasurement, these removals are not included in the standard reports that are based on working forest plots. Thus, a more complex approach involving selection of plots that have been converted from working forest to something else, and then removals estimation for these plots, is needed. The removals from these converted plots should be added to the removals in the denominator of the *GROW* term. Because the number of plots in this category is so small, and the value of G/R is so comparatively large, this omission is unlikely to change the *GROW* term substantially. It is possible that in cases where gross growth barely exceeds removals, the inclusion of additional removals from land clearing could tip the balance such that removals exceed growth. However, at present, *GROW* in the sample region shows that current growth (including accumulation of biomass in standing dead trees) is substantially higher than removals.

| Region | Gross Growth (million cu. ft/yr) | Removals (million cu. ft/yr) | *GROW* |
|---|---|---|---|
| **South Central** | 9,557.1 | 5,379.2 | −0.78 |

## 2.2. *AVOIDEMIT*

*AVOIDEMIT* is not applicable in this case study as there are no avoided emissions from removal of this roundwood feedstock, or emissions would have happened regardless of the forest roundwood harvest that occurred (i.e., the trees would have been kept in place); thus, *AVOIDEMIT* drops out of the equation for roundwood.

## 2.3. *SITETNC*

This section develops an illustrative regional approach to estimating default values for *SITETNC* for roundwood feedstock using a retrospective reference point approach. If changes in feedstock demand did not induce land use changes shifting non-forested lands into forests or vice versa, and if non-feedstock C pools were constant at the site of feedstock production during two consecutive measurements, then *SITETNC* would be 0. However, if emissions (or increased sequestration) would occur from the non-feedstock carbon pools at the feedstock production site due to feedstock production and/or removal, *SITETNC* will be positive (or negative). In situations of roundwood removal for wood products or combustion at a stationary source, a certain percentage of logging residue is normally left on site. Thus the onset of increased roundwood removals may alter production site emissions because the corresponding increased input of those residues also occurs, causing higher C stock levels in the detrital pools such as the forest floor. Quantifying these changes is difficult, however, because site-level variability is substantial and because any changes are typically small in comparison to the large C pools involved. Letting the variable *"DETRITAL"* represent the change in detrital pools, where a negative value represents sequestration (i.e., increases to the detrital pool) and a positive value represents emissions (i.e., decreases in the detrital pool), the calculation of *SITETNC* is given by Equation H.4:

$$\boldsymbol{SITETNCroundwood} = \frac{\boldsymbol{DETRITAL}}{\boldsymbol{REMOVALS}} \qquad \textbf{(EQ. H.4)}$$

In order to estimate the change in detrital and soil pools on a regional basis associated with increased roundwood removals, this calculation uses empirical measurements to compare C stocks in these non-feedstock C pools at two points in time. While this baseline approach does allow for the detection of change in these pools, it cannot attribute the specific drivers of change. Therefore, where change in these pools occurred, the calculation used data on estimated roundwood harvest to attribute the measured change to production site detrital pools. This approach, in which trends in C pools are monitored over a predetermined spatial scale, follows the logic of the *GROW* term under a reference point baseline.

When roundwood is harvested, there are changes to C stocks at the site of harvest in detrital forest C pools including coarse woody debris, forest floor C, and mineral soil C. While the forest floor C pool and the relationships between forest harvest practices and soil C responses are increasingly well understood (e.g., Lal, 2005), the impact of harvesting and utilization on mineral soil C is unknown (Buchholz et al., 2013).

Thus, while it is possible that increased roundwood removal and related detrital pool changes may affect the deep mineral soil C pool, quantification of these effects as they relate to roundwood removal are not be discussed here due to a lack of deep soil C data as well as scientific consensus on the issue. Forest floor and coarse woody debris C pools are better understood, and data are available through the USDA Forest Service's FIA program (discussed further in the logging residues case study below).

### 2.3.1. Data

Removal of roundwood from a forest site has the potential to affect all detrital carbon pools covered in the FIADB, namely above and below ground carbon in seedlings, shrubs and bushes, as well as carbon in stumps, coarse woody debris and litter. As the standing dead C pool is already covered in the *GROW* term, it is not further considered for *SITETNC* to avoid double counting. A major concern with the detrital C pools is that in the FIA dataset, all of them are modeled and based on other measurements and are therefore associated with uncertainties (see Section 2.3.5).

### 2.3.2. Results

As examined in this case study, the above and below-ground carbon in seedlings, shrubs and bushes, as well as carbon in stumps, coarse woody debris and litter are based on plot-level C models (Smith and Heath, 2008) in all of these estimates. While model outputs for coarse woody debris are close to field-based estimates (Domke et al., 2013), these models were developed for national greenhouse gas inventories and not designed to detect changes on a plot level itself based on a shift in management regimes. Moreover, uncertainties accrue as some detrital C pool estimates rely on model outputs for other detrital pools. For instance, C in stumps and dead roots is currently calculated based on a ratio of down dead to live biomass. If down dead biomass is removed (e.g., for bioenergy applications), this method would underrepresent C in stumps and roots (EPA, 2011).[4] To avoid attributing emissions or sequestration to these C pools as land moves into and out of forest and timberland we base our estimate of change on the change in per acre values of *SITETNC* between FIA inventories rather than on the aggregate amounts.

Results for the Southern regions are given in Table H-3. In the Southeast, 77.9 million tons of roundwood were removed annually and C stocks in detrital pools rose 1.9 million tons per year. These values leads to a *SITETNC* value of −0.024 tons of detrital carbon per ton of roundwood removals. Note that these calculations assume that a contribution to the atmospheric $CO_2$ stock would be a "positive" emissions flux, while an increase in terrestrial carbon uptake would have a negative sign. Thus, *SITETNC* impacts are negative as sequestration in Southeastern detrital pools increases between the historical reference points utilized.

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[4] Dead organic matter in FIA is initially calculated as three separate pools: (1) standing dead trees; (2) down dead wood; and (3) forest floor carbon. Down dead wood includes stumps and roots of harvested trees. Ratios of down dead wood to live tree are used to estimate this quantity.

**Table H-3. *SITETNC* Term for the Southeast Roundwood Example (2010 FIA Data).**

| Region | Detrital Pool Change (million tons/yr) | Removals (million tons/yr) | *SITETNC* |
|---|---|---|---|
| **Southeast** | 1.9 | 77.9 | −0.024 |
| **South Central** | 2.0 | 99.1 | −0.020 |

### 2.3.3. Detecting Changes in Management Regimes in Detrital Pools through FIADB Sampling Designs

As described above, it is difficult to attribute change in non-feedstock measured C pools directly to changes in management regime such as increased roundwood production and resulting increased residue contributions to the detrital pool and impacts of residue removal (the latter is discussed in the next section on logging residues). For example, a subset of the FIA plots (e.g., 1/6$^{th}$ to 1/16$^{th}$) are monitored for these residue pools across a measurement cycle, and only a fraction of these plots might be affected by harvest activities, let alone residue removal. As such, the signal from changed forest management might be lost in the overall noise (meaning that the change in residue C pool stats from the subsample of FIA plots may not be significantly different from 0), particularly if large-scale stochastic disruptions occur such as fires or insect outbreaks. While the latter issue is not of concern for the *GROW* term as long as the regions are held large enough, such stochastic events may affect the larger noise to signal ratio associated with a smaller plot size and smaller relative changes in the affected non-feedstock (detrital) C pools compared with the live biomass pool (e.g., Westfall et al., 2013). For instance, if a region exhibits a 50-year harvest reentry interval for harvest activities and FIA measurements occur in 5-year cycles, around 10% of the C influx to the detrital pools would be removed if: (1) all harvest sites experience residue removal; and (2) all of the residues would be removed. As the reentry interval is 50 years, this signal would occur only in 2% of the measured plots.

In addition to uncertainties derived from imprecision in residue C pool measurements or sampling sizes, another source of uncertainty lies in the conversion factors applied to produce C pool estimates from direct measurements such as computing the bulk density of rotting material (e.g., Russell et al., 2013).

Despite these uncertainties, the measurements of detrital pools developed as part of the *SITETNC* term do provide an estimated baseline value that can be used to monitor the extent to which these pools are changing due to increased roundwood production, and thus the extent to which emissions are occurring from at the feedstock production site as a result of these changes.

### 2.3.4. Other Detrital Carbon Pools than Woody Debris in FIA

As described above, to allow for assessment of the impact of roundwood removal on all detrital forest C pools, it would be advisable to also include the dynamics of the mineral soil pool. Other pools are expected to be covered by measurements in upcoming FIA P3 sampling designs, and pool dynamics are reasonably well understood with the exception of the mineral soil C pool (Harrison et al., 2010).

### 2.3.5. Uncertainties and Areas of Future Research

There are several issues when monitoring detrital forest in FIA's assessments, which will require additional efforts to make the approach outlined above a better representation of *SITETNC* for roundwood. First, merchantability of the roundwood resource is determined in large part by market conditions. In strong forest products markets pulp prices may be lower due to high mill residue availability, and more roundwood material is left in the forest. Likewise, in weak forest products markets pulp prices might rise due to a constrained mill residue supply leading to greater recovery from forest harvesting operations. Thus, roundwood utilization is a key component of quantifying the carbon impact of feedstock removal on detrital carbon pools. Second, the uncertainty derived from converting detrital field measurements (e.g., coarse woody debris diameter) to C pool estimates (e.g., coarse woody debris C) adds uncertainty to C pool estimates (Russell et al., 2013). Third, change in management regimes, i.e., the level of roundwood utilization and extraction, is difficult to detect using current FIA sampling designs (Westfall et al., 2013). Fourth, the specific impacts of forest management on the large deep mineral soil C pool are not well understood, a fact which is unlikely to change anytime soon (Buchholz et al., 2013) adding additional uncertainty to any soil C estimates.

In summary, uncertainties in determining roundwood harvest impacts on detrital C pools are associated with the data for all detrital C pools considered here, based in large part on conversion factors and extrapolation methods. Whether monitoring detrital resources strictly through model-based or field-based approaches, there is cumulative error (i.e., sampling error, measurement error, and model error) that should be acknowledged and incorporated into assessments.

# 3. Illustrative Method for Developing Regional Default Values for Biogenic Attributes: Pacific Northwest Logging Residues

This section develops illustrative regional default values for the use of logging residues, i.e., material that would otherwise have been left on the forest floor as harvest residue, as a feedstock for a stationary source that emits biogenic $CO_2$, in the Pacific Northwest. It is important to note that this analysis includes data and results for avoided emissions from logging residue feedstock use that is not retrospective in nature: it is based on a literature-based alternative fate counterfactual assessment value which includes decay rates into the future. In an application of the framework that necessitated only retrospective analysis, inclusion of decay rates and other future alternative fate counterfactual assessments could not be included. This alternative fate analysis is included as an alternative method to the modeled detrital pool analysis presented in the Southeast Roundwood section above.

The discussion in this section is based on the assumption that the only action involved is the extraction and utilization of logging residues from already occurring forest harvesting operations[5],

[5] Forest products, in general under current market conditions, are characterized by a joint production function, as many products/materials can be produced from the harvest of a single tree. Firms strive to optimize production to maximize the amount of high-value products (e.g., saw lumber, paper) and minimize the amounts of lower value

and thus the *GROW* term is set to 0. Furthermore, this case study is not completely retrospective as it assumes some additional amount of logging residue harvest that would not have occurred under business-as-usual conditions. In the previous roundwood example, logging residues would ultimately contribute to the detrital carbon pools in *SITETNC*. As such, the methodology employed for logging residue use in this case study assumes that the biogenic feedstock is extracted from *SITETNC* pools, resulting in a positive emission that reflects reduced sequestration in these detrital pools. This is a valid assumption as residues from logging operations would contribute to these pools in the absence of collection and utilization. Since logging residues are assumed to be additional, keeping this emissions impact of biomass removal within the *SITETNC* pool (rather than *GROW*) is justified. Finally, the resulting reduction in woody debris decay emissions in the forest is credited a negative *AVOIDEMIT* value.

## 3.1. *GROW*

For logging residues in this illustrative application of the reference point baseline, the *GROW* term does not apply because the logging residue feedstock is assumed to taken place due to already occurring harvesting operations therefore not impacting the *GROW* term's growth or removals.

## 3.2. *AVOIDEMIT*

The *AVOIDEMIT* term does apply for this feedstock because *AVOIDEMIT* represents the emissions that would have occurred at the field site had the feedstock (i.e., logging residues) not been removed for bioenergy. The carbon stocks considered in the context of *AVOIDEMIT* can be termed "detrital" stocks to denote that they are dead and "non-growing." In forests, these detrital feedstocks could include tree tops, branches, and stumps left after a roundwood harvest.

Deciding on an appropriate value of *AVOIDEMIT* requires an assessment of the ratio between the amount of C that is stored long-term on site via leaching into the soil C pool, and the amount of C that would have been emitted to the atmosphere via feedstock decay if the residue were left onsite or emitted to the atmosphere from open-burning onsite (Miner et al., 2014). As some of the C in the residue feedstock would have leached into the forest floor if the residue were not removed, the value of *AVOIDEMIT* should include at least some level of long-term C storage in most cases (discussed below).

### 3.2.1. *AVOIDEMIT* for Logging Residues in Non-fire-prone Regions

The literature was reviewed in order to assess the degree of long-term C storage on site associated with leaching of C into the soil C pool, in order to estimate appropriate regional values for *AVOIDEMIT* for logging residues. While there is a large scientific literature on wood decomposition rates in U.S. forests (Jandl et al., 2007; Johnson and Curtis, 2001; Jones et al., 2011; Laiho et al.,

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products (e.g., mill or logging residues). While there is some responsiveness to relative price movements (e.g., higher demand and prices for wood pellets may lead to an increased proportion of mill or logging residues going to this use and a decreased proportion going to particleboard or other uses), the elasticity of transformation between outputs may be very inelastic, and even with a negative price some low-value products would still necessarily be produced as a byproduct of the production of higher value products (e.g., sawdust, black liquor).

2003; Smith and Heath, 2002), the fraction of detrital C that will be mineralized and stored long-term in the forest soil remains largely unknown (Buchholz et al., 2013; Nave et al., 2010), and is probably highly site- and management-specific. Deep mineral soil C measurements are not available for FIA plots (Harrison et al., 2010) and measuring deep soil C is difficult (e.g., Johnston et al., 2004): thus, an empirical dataset applicable for smaller spatial scales that can link forest management activities with changes in deep soil C is not currently available (Smith et al., 2012).

The values for *AVOIDEMIT* presented here utilize a non-combustion (not open burn) related *AVOIDEMIT* term of −0.98, which means that 2% of the logging residue would not have been released into the atmosphere from on-site residue decomposition and would instead have entered long-term sequestration in deep soil pools (see Table H-4) (Zanchi et al., 2012). Soil type, microbiological activities, solar radiation reaching the forest floor, land use, or climate can influence the rate of long-term sequestration in soils. While a value of -0.98 is used here, note that changes in mineralization rates and long-term storage have also been observed, but as of yet the drivers of these changes are not understood (e.g., Nave et al., 2010; Pregitzer and Euskirchen, 2004; Zummo and Friedland, 2011).

### 3.2.2. *AVOIDEMIT* for Logging Residues in Fire-prone Regions

With the exception of the Northeast and North Central regions, all other regions in the United States practice slash burning to varying extents.[6] If current practice entails the burning of all logging residues onsite in slash piles, such that combustion of slash is the baseline, it can be argued that the fraction of carbon remaining from the logging residues following the burning equals 0.06–0.08 for the affected regions. Finkral et al. (2012) found that "on average, burning released between 92% and 94% of the carbon in each slash pile to the atmosphere," while DeLuca and Aplet (2008) discuss the longevity of charcoal C pools under typical U.S. forest floor conditions. Additional research will be required to: (1) produce evidence-based and proven numbers on the long-term fraction of carbon remaining from the logging residues for both combustion and non-combustion related soil C dynamics; (2) identify current slash handling practices across all regions to establish a defensible fraction of carbon remaining from the logging residues including 'slash burning baselines;' and (3) categorize intra-regional ecosystem variations with potentially diverging logging residue baselines such as in the Pacific Northwest that combines forest types with both very high and very low fire frequencies.

Burning of logging residues is the common current practice, and assuming near-complete combustion, the content of C in wood ash is diminutive (e.g., Demeyer et al., 2001) and *AVOIDEMIT* approaches a value of −1 (i.e., −0.98 as indicated in column one of Table H-4). In contrast, less complete combustion of residues leaves more ash behind, thereby changing the balance between emissions to the atmosphere and C stored long-term via leaching (i.e., the 7% difference from a value of −1 (i.e., −0.93) indicated in column two of Table H-4).

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[6] Slash burning is the deliberate and controlled incineration of logging residues onsite to reduce the risk of uncontrolled ignition (Smith et al., 1997).

**Table H-4. Estimated *AVOIDEMIT* Values for Logging Residues. Values Were Estimated for Two Management Practices: Without and With Onsite Combustion of Logging Residues. Numbers are uncertain at this point and are presented as placeholders until further research confirms long-term storage of combustion and non-combustion related soil C.**

| Management Practice | *AVOIDEMIT* |
|---|---|
| **Current management does not combust logging residue on site** | −0.98 |
| **Current management combusts all logging residue on site** | −0.93 |

### 3.3. *SITETNC*

In logging residue collection systems, periodic entries to remove harvesting residuals alongside roundwood harvest operations leave the rest of the standing live C pool largely intact in comparison to a roundwood-only harvest. As long as the roundwood harvest is part of the normal operation for the working forests in the source region (and does not represent a new management practice or a change in management intensity), then other impacts on *SITETNC* can be expected to be 0 (Masek et al., 2011). In traditional harvests tree boles are removed while tops and limbs are left on site (Stenzel et al., 1985). The addition of these tops and limbs to the *SITETNC* C pool would constitute sequestration to the pool (negative emissions flux to the atmosphere). However, with these tops and limbs being collected and utilized as a biogenic feedstock that negative addition to *SITETNC* does not occur and so when compared to traditional practices declines in litter, dead wood, and soil C stocks will result. In this case, where a transition from a traditional practice of leaving tops and limbs on site to a logging residue collection system, the *SITETNC* impact is equal to 100% of the residue portion of the harvest removed. When *SITETNC* is considered together with *AVOIDEMIT*, the combined assessment factor in the case where the logging residues would not have been burned is 0.02 (−0.98+1.00) representing the 2% of logging residue carbon that would have remained long term in the soil.

**Table H-5. Estimated *SITETNC* Values for Logging Residues.**

| Management Practice | *SITETNC* |
|---|---|
| **Proportion of logging residues removed from the forest** | +1.00 |

## 4. Illustrative Method for Developing Regional Default Values for Biogenic Attributes: Corn Belt Corn Stover

Developing regional landscape attribute values for many non-traditional agriculture-derived biogenic feedstocks including corn stover as well as switchgrass, hybrid poplar, and other dedicated energy crops using a retrospective reference point baseline is currently challenging. The main reason for this is that these feedstocks (unlike traditional crops used for liquid biofuel production) have either not been used traditionally for large scale stationary source energy production (e.g., stover) or they have not been commercially cultivated to the extent that there is observed historical data at a regional or national scale (e.g., dedicated energy crops). Without such datasets, it is difficult to estimate the alterations in management practices or in land usage and the associated biogenic C impact profiles when the feedstock is collected or produced for energy. In the future, it may be possible to calculate more concise *SITETNC* values with a retrospective reference point baseline using historical data providing that there is widespread use of and data collection pertaining to the feedstock of interest.

The discussion in this section focuses on possible ways to estimate *AVOIDEMIT* and *SITETNC* values for corn stover residues if needed before such datasets are available (the *GROW* term is not relevant to this feedstock category). The following section delves further into the challenges of producing similar estimates for other agricultural and dedicated energy crops using this baseline approach.

### 4.1. *GROW*

For crop residues created from annual crops such as corn stover, the carbon sequestered by the annual growth of the feedstock is counterbalanced by the carbon in the feedstock at harvest, hence *GROW* is set to 0.

### 4.2. *AVOIDEMIT*

The *AVOIDEMIT* term reflects the avoided emissions, those that would have occurred anyway from the field site without the removal of the biogenic feedstock for bioenergy use (i.e., decay). The fate of agricultural residues is different from forest residues for a number of reasons including differing decay rates, no commonly measured long-term detrital carbon pools, and site-specific management. That said, there are similarities between agricultural and forest residues. For example, when agricultural residues are burned, an unburned fraction remains. The unburned fraction of agricultural feedstocks varies by feedstock (e.g., see EPA, 1994, 2013). However, agricultural residue burning did not contribute to long-term carbon storage in a 31-year study by Rumpel (2008).

For agricultural crop residues such as corn stover in this illustrative calculation, if the residues had not been removed from the field, they would have decomposed, with all the carbon in the residues oxidizing before the start of the next production year. For example, leaving crop residues in the field was found not to contribute significantly to soil carbon in a study by Gale and Cambardella (2000). Tillage practices and fertilizer application had a larger effect on soil carbon change when compared to residue removal in a 30-year study by Reicosky et al. (2002) and in a study by Clapp et al. (2000). Dick et al. (1998) also found that tillage and rotation played a larger role in soil carbon change than did residues. Residues can, however, contribute to soil organic matter, provide a physical buffer, improve the chemical, physical and biological properties of the soil, reduce raindrop impact and wind shear, reduce erosion, and increase yield (under certain conditions) (Andrews, 2006; FAO, 2004). The magnitude of these effects is highly variable at a national scale. Variables such as crop type, growing conditions, and agricultural practices all affect the potential quantities of residue converted to soil carbon (Andrews, 2006). USDA recommendations for appropriate residue removal for biofuel production suggest that removal rates be based on regional yield, climatic conditions, and cultural practices with no specific national rates provided (Andrews, 2006).

Under a truly retrospective baseline approach, *AVOIDEMIT* could be assigned a value of −1 (i.e., all emissions would have occurred anyway). However, given data limitations on historic corn stover removals, this case study evaluates the net landscape emissions effect of a management switch from conventional corn production to corn production with residue harvesting using biophysical processing modeling coupled with relevant information on regional crop-mix and management data. The approach compares the soil carbon profiles of two different management regimes, and implicitly includes the portion of emissions from feedstock decomposition for each scenario. Thus, *AVOIDEMIT* is given a value of 0 for this hypothetical case study and all landscape emissions impacts are included in the *SITETNC* term (discussed in detail below). This approach also acknowledges the variability and inherent uncertainty in the contribution of crop residues to soil carbon at the national scale. It is noted that specific residue removal rates, if such data is gathered and compiled nationally, might influence the estimated contribution of crop residues to soil carbon (and other factors) under certain conditions.

## 4.3. *SITETNC*

This section discusses the data needs for deriving net landscape emission estimates for corn stover used in stationary sources for energy and then presents the methods used in this appendix to develop illustrative values for use in the illustrative case study presented in Appendix I.

### 4.3.1. Data and Methods

In an evaluation of corn stover, information on management activities, whether land use and/or management activities have changed, and the associated biogenic $CO_2$ flux profile would be needed. For example, one would need measures of what corn stover yield was and whether there were alterations in management that caused changes in soil carbon following corn stover harvest. Such information could be based on data collected by localized or national farm surveys such as the USDA ERS ARMS survey (if available), state-level Agricultural Extension Service reports (if available), the use of estimates from models with soil and GHG modeling capabilities, some mixture of agronomic experimental data and field measurements, or a combination of these methods.

Management activities such as increased stover removal and related soil carbon impacts are currently not captured in national datasets and can vary significantly by soil type, environmental condition, and management profile. Therefore for this illustrative framework application using the retrospective reference point baseline to evaluate corn stover impacts in the Corn Belt region, the *SITETNC* equation term uses a proxy that was generated by using results from the DAYCENT model in conjunction with data from USDA on the prevalence of particular tillage types by region and fertilizer use by crop and region (USDA ERS, 2013). These merged datasets were then applied to statistical meta-models (akin to response surface regressions) developed by Dr. Stephen Ogle of the Natural Resource and Ecology Laboratory to estimate soil carbon and nitrous oxide emission changes between different scenarios (with and without corn stover removals). This *SITETNC* calculation method differs from other methods used for agriculture- and forestry-derived feedstocks for which there is historical data that can be used to assess changes between two reference points in the past. For illustrative purposes this method is included, despite not being truly retrospective. The rest of this section briefly describes how the proxy values are generated and a detailed description of how the meta-models were derived and applied is provided in an Addendum to this section.

The *SITETNC* estimate reflects the change in carbon when corn stover collection occurs on land assumed to have previously been used to grow corn without the corn stover being removed. This *SITETNC* term calculation accounts for the difference in the amount of sequestered carbon on a $CO_2$-equivalent basis per ton feedstock between these two scenarios. Also included in the discussion below is calculation of related nitrous oxide ($N_2O$) emissions for use as a sensitivity in the case study appendix.

The first step includes estimation of sequestration and emissions from the land under corn production in both the without and with corn stover removal scenarios. These calculations required development of estimates of initial corn stover yields (estimated quantity of residues produced per acre) and initial soil carbon profile and $N_2O$ emissions under corn production without stover removal. Then, the soil carbon and $N_2O$ emissions profile or corn production was updated assuming management changes associated with corn stover removal. Appendix D of the Beach and McCarl (2010) RFS2 report describes calculation of the quantity of residues produced per acre by crop and provides citations for what was used for the percent of total residues that could be sustainably removed (Graham et al., 2007; Perlack et al., 2005). As a general rule, USDA National Resources Conservation Service (NRCS) recommends that about 30% residue cover is adequate to control soil erosion (Maung, 2007). Removable residue values used in the Forest and Agriculture Sector Optimization Model with Greenhouse Gases (FASOM-GHG) are calculated by adjusting the residue production per acre based on the harvestable percentages provided in Graham et al. (2007) and Perlack et al. (2005) which consider the effects of erosion and runoff. This approach uses a maximum percentage removal of residues,[7] which vary by crop and tillage.

The corn stover yield and related DAYCENT emissions estimates were developed in conjunction with regional dryland and irrigated crop budgets in the FASOM-GHG model. These budgets were developed over the years in FASOM-GHG based on farm budget data from regional extension services and publicly available USDA ERS datasets. In turn those data were disaggregated so that they contained different tillage and fertilization levels using runs from the DAYCENT model and budget data on costs from USDA NRCS. The budget data were updated to 2010 in terms of yields and nitrogen utilization based on USDA ERS ARMS data and USDA annual agricultural statistics at the state level.

With this information the two scenarios are developed:

- **Corn without Stover Harvested ($CropEmissioncrop_{r,corn}$).** Emissions and sequestration estimates in the without stover removal scenario includes the carbon that resides in the soil per acre when stover is left on site. The land use associated with stover removal is assumed to not change. This calculation involved usage of DAYCENT results on carbon sequestration rates and $N_2O$ emissions in metric tons $CO_2$ equivalent per acre.

[7] Many site specific factors associated with the sustainable removal of residue (e.g., crop type, soil type, soil fertility, slope, and climate) affect which geographic regions are suitable for crop residue removal. Detailed modeling of these factors was beyond the scope of this analysis.

- **Corn with Stover Harvested (*$CropEmissioncrop_{r,stover}$*)**. Consideration of emissions and sequestration estimates when stover is removed necessitated estimates of soil carbon stock changes and $N_2O$ emissions. The soil carbon sequestration and $N_2O$ estimates were derived from DAYCENT. The yield of harvestable corn stover was based on USDA NRCS estimates were deduced from the estimated amount of stover that needed to remain on site to limit erosion. The change in $N_2O$ emissions also considered the need for additional nitrogen to replace the nutrients removed when the stover was removed.

The second step involves solving for the *SITETNC* estimate per ton of feedstock by subtracting the carbon sequestration and $N_2O$ emissions for the average acre with corn stover removed from the average acre without stover removed. With this calculation, a positive result indicates an increase in emissions or a decrease in sequestration. These terms and results are in terms of net emissions/sequestration per acre of feedstock grown in metric tons of $CO_2$ equivalent:

$$\mathbf{SITETNCperacre_{r,stover} = CropEmissioncrop_{r,corn} - CropEmissioncrop_{r,stover}} \qquad \textbf{(EQ. H.5)}$$

The data were transformed to a per ton feedstock amount by dividing by regional per acre yields of corn stover. These are in units metric tons of $CO_2$ equivalent emissions/sequestration per short ton (2000 lbs) feedstock. Note that this approach does not capture additional land use change or management emissions that are attributable to the increased residue demand. That is, crop production area is held constant between the scenarios, so no land use change or emissions associated with crop switching are captured.

### 4.3.2. Results

The illustrative results presented in Table H-6 below reflect the annual average *SITETNC* values with and without $N_2O$ emissions expressed in metric tons of $CO_2e$ per ton of feedstock for corn stover in the Corn Belt.

**Table H-6. *SITETNC* per Ton of Feedstock for Corn Stover in the Corn Belt.**

| Feedstock | Region | SITE_TNC (carbon only) (metric tons $CO_2e$ per ton feedstock) | SITE_TNC (carbon + nitrous oxide) (metric tons $CO_{2e}$ per ton feedstock) |
|---|---|---|---|
| **Corn Stover** | Corn Belt | +.0026 | +0.0123 |

# 5. Supplemental Information

This section provides supplement information on the methods used to develop the illustrative *SITETNC* values for corn stover in the Corn Belt.

## 5.1. Details on the Meta-models Used to Derive Soil $N_2O$ and Soil Organic Carbon Stock Changes for *SITETNC*

Meta-models were developed to estimate soil organic carbon stock changes and soil nitrous oxide emissions associated with various management alternatives. The data were generated using the

DAYCENT ecosystem model, and were used to derive linear-mixed effect models that were incorporated into the FASOM-GHG model. This report provides information on how the meta-model were derived and applied.

#### 5.1.1. DAYCENT Model

The DAYCENT biogeochemical model is used to estimate crop grain and straw yields (including corn stover), soil organic carbon stock changes, and soil nitrous oxide emissions for different crops and management scenarios. DAYCENT (Del Grosso et al., 2006; Parton et al., 1998) is a process based model of intermediate complexity, and simulates the influence of management practices and other events, such as fire, grazing, cultivation, and fertilizer additions, on carbon and nutrient dynamics in plant-soil systems. The model requires several inputs, including soil texture; current and historical land use; and daily maximum/minimum temperature and precipitation data. Plant growth is a function of soil nutrient and water availability, temperature, and plant specific parameters, such as maximum growth rate, minimum and maximum biomass carbon to nutrient ratios, and above ground versus below ground carbon allocation. Soil organic carbon is represented as three pools that are kinetically-defined with slow, intermediate and long turnover times. Carbon is transferred from dead biomass into the soil organic carbon pools, and over time will decompose and return $CO_2$ to the atmosphere. Nitrogen gas emissions (nitrous oxide, nitrogen oxides, dinitrogen gas) from nitrification and denitrification are controlled by soil mineral N levels (nitrate and ammonium), water content, temperature, pH, plant N demand, and labile carbon availability. Nitrate leaching losses are controlled by soil nitrate availability, saturated hydraulic conductivity, and water inputs from rainfall, snowmelt, and irrigation.

The ability of DAYCENT to simulate yields, soil organic matter changes, nitrous oxide emissions, and nitrate leaching for conventional crops (e.g., corn, wheat, barley) has been validated by comparing model outputs with measurements from major commodity crops and grassland systems in North America (David et al., 2009; Del Grosso et al., 2005; Del Grosso et al., 2008). The model is also shown to simulate biomass yields reasonably well for switchgrass grown at different sites in Illinois (Davis et al., 2010) and the impact of nitrification inhibitors on nitrous oxide emissions (Del Grosso et al., 2008). DAYCENT has been applied for simulation of soil greenhouse gas fluxes at scales ranging from plots to regions and the globe (Del Grosso et al., 2010; Del Grosso et al., 2005). The model has been used since 2005 to estimate nitrous oxide emissions from agricultural soils in the U.S. National Greenhouse Gas Inventory compiled by the EPA, and reported annually to the United Nations Framework Convention on Climate Change (Del Grosso et al., 2010; Del Grosso et al., 2006; EPA, 2012).

#### 5.1.2. DAYCENT Simulations

DAYCENT is used to simulate cropping systems at the county scale across the contiguous United States. To compile model inputs, the centroid for the largest cluster of cropland in each county is identified based on National Land-Cover Dataset (Homer et al., 2007). For the county-scale simulations, model inputs for daily weather are based on the DAYMET dataset for the county centroid. DAYMET (Thornton and Running, 1999; Thornton et al., 1997) generates daily surface precipitation, temperature, and other meteorological data at 1 $km^2$ resolution using weather station observations and an elevation model. Soil properties were acquired from the dominant STATSGO (State Soil Geographic Database) (USDA NRCS, 1997) map unit at the centroid cropland in a county. Hydraulic properties are calculated from STATSGO surface texture class and Saxton et al. (1986) hydraulic properties calculator.

Land management data for annual crops are compiled at the agricultural region level (McCarl et al., 1993). Most states correspond to one of the 63 regions, except a few states that were further divided into two or more regions. Data for average fertilization rates, timing of planting/harvest, and crop rotation schedules are obtained from various sources based on farmer surveys and fertilizer sales data (EPA, 2012).

Management alternatives included adoption of conservation tillage practices; land use change to cropland management from grassland and forest land; reducing nitrogen fertilizer rates; applying organic fertilizers; varying timing of fertilization events between fall and spring; and applying fertilizer with nitrification inhibitors. The simulations are conducted by randomly combining the management options for these practices in a Monte Carlo analysis with 1000 simulation in each region. DAYCENT simulated direct $N_2O$ emissions, volatilization (nitrogen oxides, ammonia) and nitrate leaching. Indirect soil nitrous emissions are estimated based on converting 1% of volatilized nitrogen and 0.75% of leached/runoff nitrogen into $N_2O$ (IPCC, 2006).

### 5.1.3. Meta-models

Meta-models for soil organic carbon stock changes, direct and indirect nitrous oxide emissions are derived for crops in each region based on the simulation results. Meta-models are also derived for crop grain and straw yields. The meta-models are developed using a linear mixed-effect modeling approach (Pinheiro and Bates, 2000). The potential set of model predictors include nitrogen fertilization rates, timing of fertilizer application, use of nitrification inhibitors, mean temperature, mean precipitation: potential evapotranspiration ratio, soil texture, residue removal rate, tillage practice, and land use change. Only variables meeting an alpha level of 0.05 are included the model; additional variables is required to reduce the Akaike Information Criteria by at least a value of 2 digits (Akaike, 1973; Burnham and Anderson, 2002). The variables included in each model vary by agricultural management region. The models are incorporated directly into the FASOMGHG economic modeling framework.

Meta-models are developed to estimate the average change in soil organic carbon stocks over increments of five year time periods in the surface soil (20 cm). The resulting estimates are in metric tons C/ha, and can be annualized by dividing by 5. The annualized data can be converted into $CO_2$ equivalents using the conversion factor, 44/12. Following conversions, the value can be multiplied by the area of the crop or grass to obtain the total change in SOC stocks for the feedstock.

$N_2O$ emissions directly emitted in the field are estimated using a meta-model, and also N that is lost from a managed field through volatilization or leaching/runoff, and later emitted as $N_2O$ in waterways or following atmospheric deposition in soils. IPCC (2006) recommends that N leaching is not included in the estimate of indirect $N_2O$ emissions if annual precipitation minus potential evapotranspiration does not exceed field water holding capacity, with the possible exception of irrigated lands. The first step is to determine which areas will have nitrate leaching according to the IPCC guidelines. A logit model is developed to determine if leaching will occur with irrigation in the regions that are too dry for leaching without irrigation.

The meta-model results for the direct $N_2O$ models, nitrate leaching/runoff and volatilization are in natural log transformed space and require a backtransform. Units are $gNO_3$-N/$m^2$/yr for the nitrate leaching/runoff, and gN-$NH_3$+$NO_x$/$m^2$/yr for volatilization. To obtain the indirect $N_2O$ emissions, the leaching and volatilization estimates are multiplied by the indirect emission factors from IPCC, which are 0.0075 $kgN_2O$-N/$kgNO_3$-N/yr for nitrate leaching/runoff, and 0.010 $kgN_2O$-N/$kgNH_2$-N+$NO_x$-N/yr for volatilization (IPCC, 2006).

The direct and indirect emission results are in $kgN_2O$-N/$m^2$/yr, and are converted into $kgN_2O$/$m^2$/yr using the conversion factor, 44/28. In turn, the estimate can be converted into $CO_2$ equivalent using 310 or other alternative GWP conversion factors. Following conversions, the resulting value is multiplied by the area of crop or grass to obtain the total direct $N_2O$ emissions on an annual basis in $CO_2$ equivalent units.

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# Appendix I: Illustrative Forestry and Agriculture Case Studies Using a Retrospective Reference Point Baseline

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## List of Tables

# 1. Introduction

This appendix presents three illustrative case studies to demonstrate how values for biogenic landscape and process attributes could be combined to calculate the net biogenic emissions (*NBE*) and biogenic assessment factor (*BAF*) using a retrospective reference point baseline. Nested within each case study section are various sensitivity analyses. These analyses highlight the influence of alternative scenarios on the base case. The three illustrative case studies and associated sensitivity analyses focus on:

- Roundwood in the Southeast;
- Logging residues in the Pacific Northwest; and
- Corn stover in the Corn Belt.

These case studies use the biogenic assessment equation from the main report:

$$\boldsymbol{NBE} = (\boldsymbol{PGE})(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC})(\boldsymbol{L})(\boldsymbol{P}) \qquad \textbf{(EQ. I.1)}$$

This appendix uses the illustrative biogenic landscape attributes (*GROW*, *AVOIDEMIT*, and *SITETNC*) as calculated using the retrospective reference point baseline approach in Appendix H. For simplicity, feedstock carbon losses during storage, transport, and processing (*L*) are held constant at 1.1, feedstock carbon embodied in products (*P*) is also constant at 1 (both of these biogenic process attributes are discussed in Appendix G). Assessment of potential leakage effects associated with feedstock production (*LEAK*) is not included in this case study application.

# 2. Roundwood in the Southeast

This case study calculates the net biogenic $CO_2$ emissions from a hypothetical electricity facility with an electricity generating unit (EGU) that uses roundwood from the Southeast region as a biogenic feedstock. This case study also examines alternative scenarios as sensitivities:

- A regional aggregation of roundwood in both the Southeast (SE) and South Central (SC) regions;
- Increased roundwood removals, as reflected in increased removals in multiples of one billion cubic feet of removals (by 1, 2, 5, and 10);
- Equation term analysis (i.e., investigation of the impact of removing terms on the assessment factor calculation);
- Varied land bases in the Southeast (i.e., all forestland, all timberlands, private timberlands, all working timberlands, private working timberlands); and
- A temporal scale analysis for all timberland.

## 2.1. Base Case

For all of the case study scenarios, it was assumed that the electricity facility has an output of 30 MW, a capacity factor of 95%, and efficiency of 26% (consumes 1 bone dry ton [BDT] of roundwood per MWh of electricity produced), thus requiring an input of 250,000 BDT of roundwood per year. If we assume that 50% of the 250,000 BDT of feedstock is carbon and convert the short tons to tonnes and carbon to $CO_2$, we end up with a *PGE* of 0.42 mmt$CO_2$ for the hypothetical 30 MW plant.

**Table I-1. Biogenic Landscape Attributes: Roundwood in the Southeast.**

| Feedstock/Region | Growth (G) (cu ft/yr) | Removals (R) (cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* |
|---|---|---|---|---|---|
| **Roundwood/Southeast** | 7.6 | 4.4 | −0.74 | 0 | −0.02 |

**Table I-2. Process Attributes: Roundwood in the Southeast.**

| Feedstock/Process | *P* | *L* |
|---|---|---|
| **Roundwood/EGU** | 1 | 1.1 |

This case study then uses the main biogenic assessment factor equation from the main report:

$$\boldsymbol{NBE = (PGE)(GROW + AVOIDEMIT + SITETNC)(L)(P)} \quad \textbf{(EQ. I.2)}$$

Inserting the illustrative values for the relevant equation terms, this equation is now:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.74 + 0 + -0.02)(1.1)(1)}$$

And the result is:

$$\boldsymbol{NBE = -0.35\ mmtCO_2}$$

For this case study application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$\boldsymbol{BAF = NBE/PGE}$$

$$\boldsymbol{BAF = -0.35\ mmtCO_2e/0.42\ mmtCO_2e}$$

$$\boldsymbol{BAF = -0.84}$$

*GROW* is less than 0 because the Southeast is currently experiencing greater forest growth compared with removals. Because *AVOIDEMIT* is 0 in this application, and *SITETNC* has a small value, *GROW* is the driving factor and causes the *BAF* to be negative.

## 2.2. Regional Aggregation

To evaluate the sensitivity of estimates to the geographic domain, the *GROW* terms for the SE region and the SC region were computed separately and as an aggregated southern region to demonstrate the impact of using larger spatial scales to develop the *BAF* estimates. Table I-3 contains the growth and removals values (from the 2010 FIA survey period, which includes data collected between 2006 and 2010) for private timberlands for the SE and SC regions and then for the South as a whole.

The process attributes—*P* and *L*—remain the same as for the base case.

**Table I-3. Biogenic Attributes for the Southeast and South Central Regions.**

| Feedstock/Region | Growth ($G$) (billion cu ft/yr) | Removals ($R$) (billion cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* |
|---|---|---|---|---|---|
| **Roundwood/Southeast** | 7.60 | 4.38 | −0.74 | 0 | −0.024 |
| **Roundwood/South Central** | 9.58 | 5.38 | −0.78 | 0 | −0.020 |
| **Roundwood/Combined Southeast and South Central** | 17.16 | 9.76 | −0.76 | 0 | −0.022 |

Inserting the illustrative values for the relevant equation terms into the main biogenic assessment factor equation from the main report, this equation is now:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.76 + 0 + -0.02)(1.1)(1)} \quad \textbf{(EQ. I.3)}$$

And the result is:

$$\boldsymbol{NBE = -0.36\ mmtCO_2e}$$

For this case study application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$\boldsymbol{BAF = NBE/PGE}$$

$$\boldsymbol{BAF = -0.36\ mmtCO_2e/0.42\ mmtCO_2e}$$

$$\boldsymbol{BAF = -0.86}$$

Combining the Southeast and South Central regions results in a very slight change to the *GROW* term for the Southeast region, though overall growth still exceeds removals in the combined region.

## 2.3. Increased Removals

Increased removal scenarios were analyzed for roundwood in the SE region to demonstrate the potential impact of changing future roundwood harvests as a biogenic feedstock on the assessment factor under the reference point baseline. This analysis changes the removals term in isolation and thus does not mimic growth responses associated with land use change to meet increased demand or enhanced growth due to changes in management to accommodate increased removals. This variation represents the base case removals increased by different multiples of 1 billion cubic feet (Table I-4). In each case, *GROW* and *SITETNC* are calculated using the methods described in Appendix H, so the "*REMOVALS*" volume in the denominator of Equations H.2 and H.4 increases with each increased removal case. For *SITETNC*, the numerator of Equation H.4 stays constant, so the estimated ratio decreases with the level of removals. For GROW, the *"REMOVALS – GROWTH"* difference in Equation H.2 changes with greater removals, ultimately causing the sign of the *GROW* term to switch from negative to positive.

**Table I-4. Increased Removals by Multiples of 1 Billion Cubic Feet.**

| Region | Growth (*G*) (billion cu ft/yr) | Incremental Removal Increases (cu ft) | Removals (*R*) (billion cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* | *BAF* |
|---|---|---|---|---|---|---|---|
| **Southeast** | 7.60 | Base | 4.38 | −0.74 | 0 | −0.024 | −0.84 |
| **Southeast** | 7.60 | 1 billion | 5.38 | −0.41 | 0 | −0.020 | −0.48 |
| **Southeast** | 7.60 | 2 billion | 6.38 | −0.19 | 0 | −0.017 | −0.23 |
| **Southeast** | 7.60 | 5 billion | 9.38 | 0.19 | 0 | −0.010 | 0.20 |
| **Southeast** | 7.60 | 10 billion | 14.38 | 0.47 | 0 | −0.007 | 0.51 |

The calculations below step through the equation to generate the *BAF* values in Table I-4. In the increased removal scenarios presented below, as the *GROW* term increases and everything else stays the same, the *BAF* increases.

### Scenario 1: Incremental Removals by 1 Billion

One billion cubic feet increase: Inserting the values for *L*, *P*, and *SITETNC* into the equation results in:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.41 + 0 - 0.02)(1.1)(1)} \qquad \textbf{(EQ. I.4)}$$

And the result is:

$$\boldsymbol{NBE = -0.20\ mmtCO_2e}$$

For this case study application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$\boldsymbol{BAF = NBE/PGE}$$

$$\boldsymbol{BAF = -0.20\ mmtCO_2e/0.42\ mmtCO_2e}$$

$$\boldsymbol{BAF = -0.48}$$

### Scenario 2: Incremental Removals by 2 Billion

Two billion cubic feet increase: Inserting the values for *L*, *P*, and *SITETNC* into the equation results in:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.19 + 0 - 0.02)(1.1)(1)} \qquad \textbf{(EQ. I.5)}$$

And the result is:

$$\boldsymbol{NBE = -0.10\ mmtCO_2e}$$

For this case study application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = -0.10\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = -0.23$$

### Scenario 4: Incremental Removals by 5 Billion

Five billion cubic feet increase: Inserting the values for *L*, *P*, and *SITETNC* into the equation results in:

$$NBE = (0.42\ mmtCO_2e)(0.19 + 0 - 0.01)(1.1)(1) \qquad \textbf{(EQ. I.6)}$$

And the result is:

$$NBE = 0.08\ mmtCO_2e$$

For this case study application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = 0.08\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = 0.20$$

### Scenario 5: Incremental Removals by 10 Billion

Ten billion cubic feet increase: Inserting the values for *L*, *P*, and *SITETNC* into the equation results in:

$$NBE = (0.42\ mmtCO_2e)(0.47 + 0 - 0.01)(1.1)(1) \qquad \textbf{(EQ. I.7)}$$

And the result is:

$$NBE = 0.21\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = 0.21\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = 0.51$$

As shown in Table I-4 above, increased removals generate changes to the *GROW* term because removals increase while growth does not. Removals begin to exceed growth—and result in a net atmospheric contribution of biogenic $CO_2$ emissions from our hypothetical EGU using roundwood—

when current removals are increased by more than 3.3 billion cubic feet. However, it should be noted that current removals are already at a high level (4.8 billion cu ft/yr) relative to net growth compared with other regions nationwide.

## 2.4. Equation Term Analysis

For the equation term analysis, the assessment factor was calculated using base levels of the biogenic landscape attributes (Table I-5) with and without certain equation variables (e.g., *GROW* and *SITETNC*). As for the base case above, for roundwood, $L$ = 1.1 and $P$ = 1, while *AVOIDEMIT* is equal to 0. By calculating the assessment factor with and without certain variables in the equation, this analysis illustrates the relative importance of those terms.

**Table I-5. Biogenic Attributes for Term Analysis.**

| Feedstock/Region | Growth ($G$) (billion cu ft/yr) | Removals ($R$) (cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* |
|---|---|---|---|---|---|
| **Roundwood/Southeast** | 7.60 | 4.38 | −0.74 | 0 | −0.024 |

### Scenario 1: Without the *GROW* Term

In this equation term analysis, *GROW* is excluded from the equation to evaluate its impact on the assessment factor.

$$\boldsymbol{NBE = (PGE)(AVOIDEMIT + SITETNC)(L)(P)} \qquad \textbf{(EQ. I.8)}$$

Inserting the illustrative values for relevant equation terms into this equation results in:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(0 - 0.024)(1.1)(1)}$$

And the result is:

$$\boldsymbol{NBE = -0.01\ mmtCO_2e}$$

For this application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$\boldsymbol{BAF = NBE/PGE}$$

$$\boldsymbol{BAF = -0.01\ mmtCO_2e/0.42\ mmtCO_2e}$$

$$\boldsymbol{BAF = -0.03}$$

Excluding the *GROW* term results in an increase in the negative assessment factor compared with the base case because only the changes in non-tree pools represented in the *SITETNC* term are represented in the *BAF*. The resulting assessment factor remains negative, however, because the *SITETNC* term pools have been increasing through the reference period.

### Scenario 2: Without the *SITETNC* Term

In this equation term analysis, *SITETNC* is excluded from the equation to evaluate its impact on the assessment factor.

$$\boldsymbol{NBE = (PGE)(GROW + AVOIDEMIT)(L)(P)} \qquad \textbf{(EQ. I.9)}$$

Inserting the illustrative values for relevant equation terms into this equation results in:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.74 + 0)(1.1)(1)}$$

And the result is:

$$\boldsymbol{NBE = -0.34\ mmtCO_2e}$$

For this application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

$$\boldsymbol{BAF = NBE/PGE}$$

$$\boldsymbol{BAF = -0.34\ mmtCO_2e/0.42\ mmtCO_2e}$$

$$\boldsymbol{BAF = -0.81}$$

Excluding the *SITETNC* term also results in an increase in the assessment factor compared with the base case because only the changes in tree biomass pools that are represented in the *GROW* term are represented in the *BAF*. The resulting assessment factor remains negative because the *GROW* term pools have been increasing through the reference period.

## 2.5. Working Forest

Forest inventory estimates (such as growth and removals) can be expressed for different land areas or definitions of forest. In general, FIA distinguishes between forestland (all land covered with forest as defined by FIA) and timberland (forest meeting certain minimum productivity thresholds and not reserved from timber harvest by law). Forest owned by public entities can be further differentiated from forest owned by private entities (because, for example, private timberland accounts for 98.8% of all removals or harvests within the Southeast). The land under consideration can be further restricted to the "working forest," which can be defined as accessible lands not constrained by steep slopes or wet soils or other criteria that would serve to limit the ability of these lands to produce commercial wood fiber.

For the purposes of this working forest analysis, *GROW* estimates are developed for six categories of land for the Southeast United States:

1. All forest lands (all lands meeting the FIA definition of forest);
2. All timberlands (forest land above productivity thresholds not reserved from harvest);
3. Private forest lands (#1 above for private ownerships);
4. Private timberlands (#2 above for private ownerships);

5. Working timberlands (#2 above, further screened to eliminate steep slopes, wet soils, etc.); and
6. Private working timberlands (#4 above, further screened to eliminate steep slopes, wet soils, etc.).

These categories are summarized in Table I-6.

**Table I-6. Land Base Categorization for the Working Forest Definition Case Study.**

| Land Base | Public Lands | Private Lands | Reserved Land | Low Productivity Land | Steep Slopes, Hydric Soils, etc. |
|---|---|---|---|---|---|
| **All Forest Lands** | Yes | Yes | Yes | Yes | Yes |
| **All Timberlands** | Yes | Yes | No | No | Yes |
| **Private Forest Lands** | No | Yes | Yes | Yes | Yes |
| **Private Timberlands** | No | Yes | No | No | Yes |
| **All Working Timberlands** | Yes | Yes | No | No | No |
| **Private Working Timberlands** | No | Yes | No | No | No |

Applying these different land definitions to the southeastern U.S. FIA data from the 2010 survey period, we obtained different values for growth/removals ratios as depicted in Table I-7.

**Table I-7. Biogenic Attributes for the Working Forest Definition Case Study Sensitivity.**

| Roundwood/Southeast | Growth (G) (billion cu ft/yr) | Removals (R) (billion cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* | *BAF* |
|---|---|---|---|---|---|---|
| **All Forest Lands** | 8.24 | 4.43 | −0.86 | 0 | −0.022 | −0.97 |
| **All Timberlands** | 8.14 | 4.45 | −0.83 | 0 | −0.024 | −0.94 |
| **Private Forest Lands** | 7.6 | 4.4 | −0.75 | 0 | −0.022 | −0.85 |
| **Private Timberlands** | 7.6 | 4.4 | −0.74 | 0 | −0.024 | −0.84 |
| **All Working Timberlands** | 7.2 | 4.1 | −0.74 | 0 | −0.024 | −0.84 |
| **Private Working Timberlands**[1] | 6.8 | 4.1 | −0.66 | 0 | −0.024 | −0.76 |

[1]Because the private "working forests" tend to incur harvests more frequently (they account for 91.6% of harvest removals) and yet account for only 82% of growth, the *GROW* term decreases as the land base used in the computation becomes more restrictive.

This section calculates one of the above alternative land base equation as an example, using the All Forest Land category.

$$\boldsymbol{NBE = (PGE)(GROW + AVOIDEMIT + SITETNC)(L)(P)} \qquad \textbf{(EQ. I.10)}$$

Inserting the illustrative values for the relevant equation terms into this equation:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(-0.86 + 0 -\ 0.02)(1.1)(1)}$$

And the result is:

$$NBE = -0.40\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = -0.40\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = -0.97$$

## 2.6. Temporal Scale

Because forest growth and removals are dynamic processes, the *GROW* term can be expected to change over time. Some of this change may be due to sampling error, some due to changes in inventory processes over long spans of time, and some due to changing rates of utilization and changing age-class distributions of forests. Because of its small value and the potential for variation due to inventory methodology, *SITETNC* is held constant across the temporal scales evaluated. To demonstrate the levels of change in *GROW* that have occurred in the past in these measures, we can use data from the periodic assessments of the U.S. forest land base conducted under the Resources Planning Act (RPA) by the USDA Forest Service. From Smith et al. (2009; tables 33, 34, and 35), growth, mortality, and removals data were extracted for the southeastern United States for RPA years prior to 2010, and FIA data were used for 2010. All estimates are based on all timberlands (see analysis for Working Forest in Section 2.5).

Table I-8 shows that the growth estimate has fluctuated from a minimum of 5,587 million cu ft/yr in 1986 to a maximum of 8,142 million cu ft/yr in 2010—a 46% increase in the Southeast—while the removals estimate has fluctuated from a minimum of 3,031 million cu ft/yr in 1976 to a maximum of 4,449 million cu ft/yr in 2010—a 47% increase. The ratio of growth/removals has also fluctuated, decreasing from a maximum of 1.98 in 1976 to a minimum of 1.34 in 1996, followed by an increasing trend to 1.83 in 2010. As previously mentioned, growth/removals is expected to change over time for a variety of reasons and will reflect changing rates of roundwood utilization and changing age-class distributions of forests, among other factors.

**Table I-8. Biogenic Attributes over Forest Inventory Time Frames for the Temporal Scale Case Study.**

| Roundwood/Southeast/ All Timberlands | Growth (*G*) (billion cu ft/yr) | Removals (*R*) (billion cu ft/yr) | *GROW* | *AVOIDEMIT* | *SITETNC* | *BAF* |
|---|---|---|---|---|---|---|
| **1976** | 5.99 | 3.03 | −0.98 | 0 | −0.024 | −1.10 |
| **1986** | 5.59 | 3.67 | −0.52 | 0 | −0.024 | −0.60 |
| **1996** | 5.96 | 4.46 | −0.34 | 0 | −0.024 | −0.40 |
| **2006** | 7.31 | 4.31 | −0.70 | 0 | −0.024 | −0.79 |
| **2010** | 8.14 | 4.45 | −0.83 | 0 | −0.024 | −0.94 |

### Scenario 1: Changed Time Frame 1966–1976

Inserting the illustrative values for relevant equation terms into the equation results in:

$$NBE = (0.42\ mmtCO_2e)(-0.98 + 0 - 0.02)(1.1)(1) \qquad \textbf{(EQ. I.11)}$$

And the result is:

$$NBE = -0.46\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = -0.46\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = -1.10$$

### Scenario 2: Changed Time Frame 1977–1986

Inserting the values for $L$ and $P$ into this equation results in:

$$NBE = (0.42\ mmtCO_2e)(-0.52 + 0 - 0.02)(1.1)(1)$$

And the result is:

$$NBE = -0.25\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$$BAF = NBE/PGE$$

$$BAF = -0.25\ mmtCO_2e/0.42\ mmtCO_2e$$

$$BAF = -0.60$$

### Scenario 3: Changed Time Frame 1987–1996

Inserting the illustrative values for relevant equation terms into the equation results in:

$$NBE = (0.42\ mmtCO_2e)(-0.34 + 0 - 0.02)(1.1)(1) \qquad \textbf{(EQ. I.12)}$$

And the result is:

$$NBE = -0.17\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$BAF = NBE/PGE$

$BAF = -0.17\ mmtCO_2e/0.42\ mmtCO_2e$

$BAF = -0.40$

### Scenario 4: Changed Time Frame 1997–2006

Inserting the illustrative values for relevant equation terms into the equation results in:

$$NBE = (0.42\ mmtCO_2e)(-0.70 + 0 - 0.02)(1.1)(1) \qquad \textbf{(EQ. I.13)}$$

And the result is:

$$NBE = -0.33\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e.$$

Therefore:

$BAF = NBE/PGE$

$BAF = -0.33\ mmtCO_2e/0.42\ mmtCO_2e$

$BAF = -0.79$

### Scenario 5: Changed Time Frame 2010

Inserting the values into this equation results in:

$$NBE = (0.42\ mmtCO_2e)(-0.83 + 0 - 0.02)(1.1)(1) \qquad \textbf{(EQ. I.14)}$$

And the result is:

$$NBE = -0.39\ mmtCO_2e$$

For this application:

$$PGE = 0.42\ mmtCO_2e$$

Therefore:

$BAF = NBE/PGE$

$BAF = -0.39\ mmtCO_2e/0.42\ mmtCO_2e$

$BAF = -0.94$

## 3. Logging Residues in the Pacific Northwest

This case study calculates the net biogenic $CO_2$ emissions from a hypothetical electricity facility with an EGU that uses logging residues from the Pacific Northwest as a biogenic feedstock. The case study illustrates how estimated values for biogenic attributes and facility-specific attributes would be combined to calculate the *NBE* and *BAF* using a reference point baseline. This case study also examines alternative scenarios including (1) equation term analysis (i.e., investigation of the impact of various terms on the final result as they are added or subtracted from the assessment factor calculation); and (2) alternative fate (i.e., investigation of the impact of assuming either a decay or combustion fate if the logging residues were not removed as biogenic feedstock).

As with the SE roundwood case study, feedstock carbon losses during storage, transport, and processing ($L$) are held constant at 1.1, feedstock carbon embodied in products ($P$) is also constant at 1, and leakage associated with feedstock production (*LEAK*) values are not included in this case study application.

## 3.1. Base Case

For all of the case study scenarios, it was assumed that the electricity facility has an output of 30 MW, a capacity factor of 95%, and efficiency of 26% (consumes 1 BDT of roundwood per MWh of electricity produced ), thus requiring an input of 250,000 BDT of roundwood per year. If we assume that 50% of the 250,000 BDT of feedstock is carbon and convert the short tons to tonnes and carbon to $CO_2$, we end up with a *PGE* of 0.42 mmt$CO_2$ for the hypothetical 30 MW plant.

In Appendix H, *SITETNC* for logging residues in the Pacific Northwest was estimated as 1 mt$CO_2$e per ton of feedstock removed. *GROW* is 0 and *AVOIDEMIT* represents an alternative fate of decomposition on site.

**Table I-9. Biogenic Landscape Attributes: Logging Residues in the Pacific Northwest.**

| Feedstock/Region | *GROW* | *AVOIDEMIT* | *SITETNC* |
|---|---|---|---|
| **Logging Residues/Pacific Northwest** | 0 | −0.98 | 1 |

**Table I-10. Process Attributes: Logging Residues in the Pacific Northwest.**

| Feedstock/Process | PRODC | L |
|---|---|---|
| **Logging Residues/EGU** | 1 | 1.1 |

Inserting the illustrative values for relevant equation terms into the equation results in:

$$\boldsymbol{NBE = (0.42\ mmtCO_2e)(0 - 0.98 + 1)(1.1)(1)} \quad \textbf{(EQ. I.15)}$$

And the result is:

$$\boldsymbol{NBE = 0.01\ mmtCO_2e}$$

For this application:

$$\boldsymbol{PGE = 0.42\ mmtCO_2e}$$

Therefore:

***BAF = NBE/PGE***

$BAF = 0.01\ mmtCO_2e / 0.42\ mmtCO_2e$

$BAF = 0.02$

*SITETNC* equals 1 in the base case to represent a ton of logging residues being removed or emitted from the feedstock production site. As described in Appendix H, the *AVOIDEMIT* value of −0.98 represents the percentage of emissions that would have occurred on the production site had those logging residues remained on site rather than been combusted at a stationary facility less the 2% that would have remained sequestered in the soil pool long term. In other words, using logging residues that would have been left at the production site following a harvest would result in a loss (or emission) of the 2% of that feedstock that would have remained on site in the long run.

# 4. Corn Stover in the Corn Belt

This case study calculates the net biogenic $CO_2$ emissions from a hypothetical electricity facility with an EGU that uses corn stover from the Corn Belt as a biogenic feedstock. The case study illustrates how estimated values for biogenic attributes, and process attributes would be combined to calculate the *NBE* and *BAF* using a reference point baseline.

This case study also examines alternative scenarios including (1) equation term analysis (i.e., investigation of the impact of various terms on the final result as they are added or subtracted from the assessment factor calculation); and (2) the influence of including fluxes of nitrous oxide ($N_2O$) at sites where corn stover is removed.

## 4.1. Base Case

For all of the case study scenarios, it was assumed that the electricity facility has an output of 30 MW per year, a capacity factor of 95% efficiency, converts 1.1 BDT of corn stover per MWh of electricity produced, and would consume an input of 275,000 BDT of corn stover per year. Note, this estimate of BDT has been revised upward from the estimates presented in the roundwood and logging residue case studies to account for the lower carbon fraction in corn stover (0.44) compared with roundwood/logging residue (0.50). Converting the 275,000 BDT of feedstock to carbon and converting the short tons to metric tonnes and carbon to $CO_2$ we end up with a *PGE* of 0.44 mmt$CO_2$ for the hypothetical 30 MW plant.

In Appendix H, *SITETNC* for corn stover in the Corn Belt was estimated as +0.0026 mt$CO_2$e per ton of feedstock removed. *GROW* is set to 0 because the ratio of net growth to removals is 0. *AVOIDEMIT* is also 0 because all emissions would have occurred anyway in the absence of residue removals. Therefore:

**Table I-11. Biogenic Landscape Attributes: Corn Stover in the Corn Belt.**

| Feedstock/Region | *SITETNC* | *GROW* | *AVOIDEMIT* |
|---|---|---|---|
| **Corn Stover/Corn Belt** | .0026 | 0 | 0 |

**Table I-12. Process Attributes: Corn Stover in the Corn Belt**

| Feedstock/Process | *P* | *L* |
|---|---|---|
| **Corn Stover/EGU** | 1 | 1.1 |

To investigate the relative impact of each of the variables on the assessment factor result, the assessment factor was calculated with and without certain equation variables (i.e., *SITETNC*).

Inserting the illustrative values for the relevant equation terms into the main biogenic assessment factor equation from the main report, this equation is now:

$$\boldsymbol{NBE = (0.44\ mmtCO_2e)(0 + 0.0026 + 0)(1.1)(1)} \qquad \textbf{(EQ. I.16)}$$

And the result is:

$$\boldsymbol{NBE = 0.0013\ mmtCO_2e}$$

For this case study application:

$$\boldsymbol{PGE = 0.44\ mmtCO_2e}$$

Therefore:

***BAF = NBE/PGE***

$\boldsymbol{BAF = 0.0013\ mmtCO_2e/0.44\ mmtCO_2e}$

***BAF = 0.0029***

## 4.2. *NBE* Results with $N_2O$ Emissions

In Appendix H, *SITETNC* for corn stover in the Corn Belt with $N_2O$ emissions was estimated as +0.0123 mt$CO_2$ equivalent (e) per ton of feedstock removed.[1,2] Once again, *GROW* is set to 0 because the ratio of net growth to removals is 0. *AVOIDEMIT* is also 0 because all emissions would have occurred anyway in the absence of residue removals. Therefore:

**Table I-13. Biogenic Landscape Attributes: Corn Stover in the Corn Belt with $N_2O$ Emissions.**

| Feedstock/Region | *SITETNC* | *GROW* | *AVOIDEMIT* |
|---|---|---|---|
| **Corn Stover/Corn Belt** | .0123 | 0 | 0 |

Inserting the illustrative values for the relevant equation terms into the main biogenic assessment factor equation from the main report, this equation is now:

$$\boldsymbol{NBE = (0.44\ mmtCO_2e)(0 + 0.00123 + 0)(1.1)(1)} \qquad \textbf{(EQ. I.17)}$$

And the result is:

[1] A detailed methodology for estimating *SITETNC* for soil carbon and $N_2O$ emissions changes can be found in Appendix H.
[2] $CO_2$ equivalence is used for *SITETNC* as $N_2O$ emissions are converted to $CO_2$ terms.

$$NBE = 0.006\ mmtCO_2e$$

For this case study application:

$$PGE = 0.44\ mmtCO_2e$$

Therefore:

***BAF = NBE/PGE***

$$BAF = 0.006\ mmtCO_2e / 0.44\ mmtCO_2e$$

***BAF* = 0.0135**

# 5. References

Smith, W.B., P.D. Miles, C.H. Perry, and S.A. Pugh. 2009. *Forest Resources of the United States, 2007*. Gen. Tech. Rep. WO-78. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 336 p.

# Appendix J: Anticipated Baselines: Background and Modeling Considerations

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# 1. Introduction

This appendix describes the anticipated baseline approach and the purpose of, and potential components needed when, applying an anticipated baseline approach prospectively to estimate the land use and biogenic carbon-based emissions and sequestration implications of U.S. biogenic feedstock consumption at stationary sources. Baseline specification can vary in terms of what entity/groups are being analyzed (e.g., industries, economic sectors), time scale, geographic resolution, and, depending on context, environmental issues/attributes (EPA, 2010a).[1] Determination of the most appropriate baseline approach and consequently appropriate modeling approaches and level of detail depends largely on the goals of the assessment.

Establishing a baseline creates a point of comparison necessary for evaluating changes to a system.[2] However, the choice of approach largely depends on the question being asked. Applications of the framework may require a baseline or baselines against which changes of landscape carbon stocks can be measured. Other applications may necessitate a baseline against which the emissions and sequestration associated with the production and use of additional biogenic feedstocks at stationary sources can be estimated and analyzed. Alternative baseline assumptions can yield different results and should be finalized after careful consideration of the specific context in which the framework is applied.

This appendix first highlights what an anticipated baseline is in general and how it compares with other baseline approaches, such as the reference point baseline. The appendix then presents the rationale for using this type of forward-looking approach, followed by a discussion on the rationale for using a prospective, or future, anticipated baseline approach for assessing biogenic emissions at stationary sources. Next is a discussion of data needs, model constructs, and model attributes that should be considered when constructing a future anticipated baseline analysis. The last section briefly describes the model chosen for constructing the baseline and alternative scenarios for the illustrative framework applications of the future anticipated baseline used in this report—the U.S. Forest and Agricultural Sector Model with Greenhouse Gases (FASOM-GHG).

[1] Guidelines for Preparing Economics Analyses (NCEE), Chapter 5: http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0568-05.pdf/$file/EE-0568-05.pdf

[2] Definitions for baseline vary, including "the reference for measurable quantities from which an alternative outcome can be measured" (IPCC AR4 WGIII) or "the baseline (or reference) is the state against which change is measured. It might be a 'current baseline,' in which case it represents observable, present-day conditions. It might also be a 'future baseline,' which is a projected future set of conditions excluding the driving factor of interest. Alternative interpretations of the reference conditions can give rise to multiple baselines" (IPCC AR4 WGII).

# 2. Anticipated Baselines: Definitions and Applications

## 2.1. Definitions and Comparison of Baseline Approaches

Two potential baseline approaches for assessing the biological carbon cycle effects related to biogenic feedstock production and use at stationary sources are evaluated in this appendix. The first baseline approach assesses the estimated net change in carbon between two points in time (i.e., between reference points) (e.g., Fargione et al., 2008; UNFCCC, 2009). This approach allows for estimation of net carbon changes relative to particular points in time. It establishes as the baseline the aggregate carbon stock related to a specific feedstock type on a given land base at a given point in time. It is against this measured reference point that aggregate stocks at another point in time will be compared, and then determines if those aggregate stocks are rising or falling between the two points in time.

In contrast, the second approach, the anticipated baseline, is a comparison between two distinct scenarios, not two points in time. While there is a time element to the anticipated baseline approach, the basis of comparison for evaluating emissions changes is the difference between two modeled scenarios (i.e., between a business-as-usual [BAU] scenario and an alternative or counterfactual scenario with changes in environmental, economic, and/or policy conditions) (e.g., Searchinger et al., 2009; Sohngen and Sedjo, 2000). This approach uses information regarding carbon stocks and the carbon balance of biogenic feedstock production systems at a certain point in time to determine either the landscape carbon profile related to specific feedstock use at stationary sources or the marginal impact of specific feedstock use at a stationary source. Both of these baseline approaches, reference point and anticipated, can be prospective (e.g., assessing the impact of a particular policy or change in biomass utilization on future carbon stocks) or retrospective (e.g., finding what the impact of an existing policy or biomass utilization has been on carbon stocks).

A primary difference between the two baseline approaches is the ability of the anticipated baseline to evaluate the additional emissions associated with biomass consumption at stationary sources. This “additionality” component is vital to determining the net contribution of additional biogenic feedstock consumption at stationary sources relative to a baseline scenario in which that consumption did not occur (additionality is discusses further in the next section). Another major difference between the reference point and anticipated baseline is the latter’s ability to incorporate potential fundamental changes in a system to gain insights about potential outcomes. In an evolving bioenergy market, such an ability can be useful to test various market and policy conditions. Finally, while a reference point approach can assess what has been taking place on the landscape in terms of emissions fluxes, an anticipated baseline approach provides information about what level of “adjustment” may be appropriate given current and expected future market and production system changes.

One purpose of applying an anticipated baseline is to provide information on potential impacts (positive or negative) of a policy, activity, or other decisions. An anticipated baseline describes the expected “business-as-usual” (BAU) conditions absent a project, policy or other “shock” to a system, against which the potential impacts of a policy or change in markets or other behavior can be described and used to understand the directionality and magnitude of possible impacts.

When anticipated baselines are applied prospectively, they can be used to evaluate scenarios of potential future outcomes relative to an estimated future BAU baseline (e.g., with and without a policy), given a set of assumptions about technologies, markets, and biophysical conditions. It is the difference between these two possible futures that provides insight regarding potential policy impacts. Prospective anticipated baselines are especially important when analysis of existing or recent historical trends to assess potential future conditions or impacts may not be appropriate. For example, it would be inappropriate to construct a retrospective baseline if there is limited or no historical data or experience from which to draw inferences (e.g., national GHG reduction incentives, bioenergy production goals, the use of currently noncommercial technologies such as new agricultural production technologies, or feedstocks not grown at the commercial scale). Use of an anticipated baseline prospectively allows for evaluation of not only future market and policy impacts, but also provides insight into how those impacts deviate from, or are additional to, the BAU trajectory.

Also, an application of a future anticipated baseline approach provides a means to estimate the potential additional emissions and sequestration changes over time in response to changes in biogenic feedstock demand. An assessment over long future time frames is particularly important in the case of the production of long rotation feedstocks such as forest-derived feedstocks as well as in cases of land use or land use management practices that may have long-term effects on landscape fluxes, including deforestation, afforestation to provide woody feedstocks, or fluxes from soil carbon pools.

### 2.1.1. Additionality

One of the primary purposes for applying an anticipated baseline approach is to ascertain whether an activity or policy has or will have resulted in GHG emission reductions or removals in addition to what would have occurred in the absence of such an action. The difference in net atmospheric carbon dioxide ($CO_2$) emissions with and without changes in biogenic feedstock use is known as additionality (Murray et al., 2007). Additionality can be determined by assessing the difference in potential net atmospheric $CO_2$ emissions of a specific level of biogenic feedstock use over a certain period of time (in many cases the BAU baseline) versus the net atmospheric $CO_2$ emissions that would have occurred over the same time period with a different level of biogenic feedstock use (counterfactual scenario), holding other factors and assumptions consistent between scenarios. When an anticipated BAU baseline consists of no biomass consumption at stationary sources, the counterfactual anticipated scenario allows evaluation of the aggregate, or average, potential market and landscape-level effects of all biogenic feedstock consumption at stationary sources. Similarly, when applied prospectively and compared with a BAU of a specific level of biogenic feedstock usage at stationary sources, an anticipated baseline allows evaluation of the incremental, or marginal, future potential market- and landscape-level effects.

This ability to assess potential additionality is particularly useful in capturing the complex interactions between biogenic feedstock production and forest product markets, including: biogenic feedstock demand; market-driven changes in planting, management, harvest regimes; market substitution effects; and direct land use change and related GHG implications. It also allows for consideration of alternate fates (i.e., what would happen to the feedstock if not combusted for energy), regional differences, and behavioral responses to market incentives. By estimating the impact of a change in policy (or other action) holding all other conditions and assumptions constant, the resulting complex interactions between markets and land use management decisions (e.g., planting regimes, land use) can be attributed to that change in policy (or other action).

## 2.2. Review of Relevant Literature

There is an extensive body of literature on the potential GHG implications of the expanded use of biogenic feedstocks for energy, many of which use some form of future anticipated baseline for analysis. Most studies have focused on annual, often agricultural, crops as feedstock for bioenergy production, mostly in the context of liquid biofuels (see reviews conducted in Gerber et al., 2008 and Pérez Domínguez and Müller, 2008). A seminal article by Searchinger et al. (2008) questioned the GHG benefits of corn ethanol given the potential for land use change and large increases in GHG emissions as an indirect market response to the biofuel demand stimuli.

More recently, increased attention has been paid to the net GHG consequences of forest bioenergy—either for transportation fuels or for electricity generation (Searchinger et al., 2009; Sedjo and Sohngen, 2012). Researchers have also begun modeling forest bioenergy pathways to better understand the GHG implications of forest bioenergy expansion (Daigneault et al., 2012; Mosnier et al., 2013; Latta et al., 2013). Although this issue mirrors many of the land use and market change concerns of the annual feedstock literature, there are unique challenges in the treatment of feedstocks with long rotations. Evaluation of these feedstocks, such as roundwood, would entail investment dynamics and interactions with traditional forest product markets.

A subset of the agricultural and forest bioenergy literature has applied an intertemporal optimization modeling approach—which explicitly assumes perfect foresight of anticipated future market and policy conditions. Several published manuscripts have applied intertemporal optimization to evaluate the impacts of bioenergy expansion. The Regulatory Impact Analysis of the Renewable Fuel Standard 2 (EPA, 2010b) applied a full suite of models to examine fuel pathway lifecycle analysis of U.S. biofuel expansion, including land use and GHG implications.

Daigneault et al. (2012) analyzed the impacts of forest biomass electricity generation in the United States using a global forest and land use model developed in earlier work and updated to reflect demand for biomass-based energy. Several scenarios were developed to test the sensitivity of results to basic assumptions of land use competition. In cases where conversion of agricultural land to new forest biomass production was unconstrained, carbon stock immediately increased. When land use was constrained, this resulted in a net increase in emissions (decrease in forest carbon stock).

Latta et al. (2013) applied an intertemporal partial equilibrium model of the U.S. forest and agricultural sectors to assess the market, land use, and GHG implications of biomass electricity expansion. Results showed how intertemporal optimization procedures can yield different biomass feedstock portfolios and GHG performance metrics at different points in time. They also evaluated the impacts of restricting feedstock eligibility, land use change, and commodity substitution. The authors highlighted the importance of dynamic considerations and forest and agricultural sector interactions on projecting the GHG effects of biomass electricity expansion in the United States.

Another prominent study is the U.S. Billion-Ton Update (DOE, 2011), which relied on the generation of future biomass supply estimates derived from given price paths for biomass feedstocks. That study estimated forest and agriculture biomass supply (in physical units) and bioenergy supply (in gallons of biofuel and kilowatt-hours [kWh]) at $40, $50, and $60 per dry ton from 2012 to 2030. At $40 and $50 per dry ton, the majority of the biomass supply is derived from agricultural residues and wastes, while at $60 per dry ton bioenergy crops dominate the supply in later years (2022 and 2030).

In the U.S. Energy Information Administration's (EIA's) Annual Energy Outlook (AEO) 2012 projections, biomass energy is one of the fastest growing renewable fuel sources, with expected growth of more than 3% per year until 2035 in the Reference case. The AEO reports both total production of energy from biomass (including wood and wood waste, biomass for liquid fuels, and nonelectric energy demand from wood) and generation in the electric generation and end-use sectors.[3]

There is a currently an expanding pool of research focused specifically on accounting for biogenic emissions, especially in the context of biogenic emissions from forest-related electricity and industrial sector stationary sources. As researchers have not reached agreement regarding the appropriateness of a standard baseline approach, literature addressing this topic will become progressively more available.

### 2.3. Application of Future Anticipated Baseline Approach in this Report

The goal of prospectively applying an anticipated baseline in this report is to assess the potential future net biogenic $CO_2$ contributions to the atmosphere from changes in biogenic feedstock consumption for energy generation at stationary sources. This future anticipated baseline approach addresses the question "Is more or less carbon stored in the system over time compared to what would have been stored in the absence of changes in biogenic feedstock use?" Thus, the future anticipated baseline approach requires a means to estimate the potential incremental impact of changes in biogenic feedstock production and use at stationary sources under specific scenario assumptions into the future. The future anticipated baseline approach accomplishes this by first establishing a BAU baseline scenario (with established levels of biogenic feedstock demand from stationary sources) and uses this as an emissions benchmark of biogenic feedstock use based on a specific set of anticipated future environmental and socioeconomic conditions. The BAU projection is then compared with a simulated alternative future scenario (or scenarios) that incorporates the same set of anticipated future environmental and socioeconomic conditions as the BAU baseline scenario and only a single specific change (e.g., increase or decrease) in biogenic feedstock demand.

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[3] AEO projections are used as part of the baseline and scenario construction in other appendices of this report. However, it is important to remember that estimates of future possible biomass supply results from AEO as well as other reports cited here cannot be compared directly with the results produced using the proposed method in this appendix because of the evaluation of different feedstocks, different end users, and other evaluation parameters.

The resulting difference between these scenarios indicates possible impacts of biogenic feedstock use.

### 2.4. Limitations and Implications of the Future Anticipated Baseline Approach

Although models are useful for gauging the responsiveness of complex economic systems and providing insights into potential responses to policy or actions, all models have associated uncertainties and limitations. This is especially true for intertemporal optimization models in which optimal economic decisions are influenced by expectations of future market and policy conditions. Results from economic optimization models are sensitive to the selection of model functions and parameters (e.g., biophysical yield parameters or demand elasticities) as well as scenario assumptions. There are also inherent uncertainties regarding input data, parameters, and model structure (in the historical data as well as future expectations). In addition, model scenario results are not predictions of the future. Instead, they should be viewed as providing insights as to what may happen under scenarios of plausible potential futures.

Future anticipated baselines allow for consideration of potential intermediate and distant futures where economic drivers may fall outside of historic ranges of data or where there is no experience with specific policies. In the case of forestry, a longer-term analysis (i.e., 40 years or greater) is often necessary to capture biophysical considerations (e.g., growth rates, rotation periods) and related investment behavior. Simulation models are often designed to consider long-run potential outcomes that tend to fall outside of historic experience.

Although longer time frames offer the advantage of capturing long-term investments for natural resource systems with long biological growth intervals, economic and physical uncertainties grow with longer time frames. Given the size and complexity of many intertemporal optimization models, characterizing uncertainty through Monte Carlo analysis or stochastic dynamic programming creates computational difficulty. Thus, uncertainty is often evaluated through sensitivity analysis by adjusting key parameters across multiple scenarios (i.e., evaluating multiple future anticipated baselines or deviations from a common baseline). Sensitivity analysis can be used to test the impacts of different assumptions employed in the model, including assumptions about future economic conditions or policies.

## 3. Key Design Elements to Consider

A future anticipated baseline approach application requires information about biomass production systems and associated $CO_2$ emissions as well as the demand system and related economic factors. The data inputs, model parameters, and assumptions about the BAU and counterfactual trajectories all play significant roles in determining results. The analysis in this section includes consideration of desired model framework/function types, future macroeconomic conditions (i.e., population, gross domestic product), and relevant sector representation. It also considers capabilities for representing future potential biophysical conditions, the land use and energy sectors, land use and commodity competition, and GHG accounting. If all of these components and model functions cannot be included in the assessment, there will be trade-offs and resulting implications.

The list below illustrates the key components to be considered in a modeling platform for simulating anticipated baselines. Each of these components is discussed in the following subsections:

- Model function types and model dynamics (economic optimization, intertemporal or recursive dynamic);
- Anticipated future conditions (macroeconomic, biophysical);
- GHG emissions representation;
- Forest sector representation;
- Agricultural sector representation;
- Land use competition;
- Energy sector representation; and
- International representation.

## 3.1. Model Function Types and Model Dynamics

### 3.1.1. Modeling Scope: Economic Optimization

In the context of this framework, a model should have a detailed biophysical component to evaluate biogenic emissions, be well grounded in economic theory, and represent the benefits, costs, and opportunity costs associated with land management and biomass processing alternatives. A suitable model or set of models should allocate resources and economic inputs to production and consumption processes that achieve the highest net economic return. This attribute is essential because it is consistent with rational economic behavior, and allows for the inclusion of autonomous adaptation to changing conditions through incentives. By optimizing net returns to economic welfare-producing activities, a model allocates resources efficiently to produce final economic goods and services. For models with a direct linkage to land use systems, optimization helps reflect the opportunity costs of different land use and/or management alternatives, with implications for future management decisions.

Different modeling frameworks are appropriate for different research questions. Generally speaking, if a broader look at the overall economy is necessary for an analysis, then an economy-wide computable general equilibrium (CGE) model may be used. If specific sectors should be evaluated in detail, such as forestry and/or agriculture, then a sectoral partial equilibrium (PE) model may be chosen.

A number of studies evaluate the trade-offs and implications of employing either CGE or PE models to evaluate the impacts of biomass (liquid or solid) production for energy. General trade-offs for using different modeling types for climate and land use policy evaluation are examined in Van der Werf and Peterson (2009). Kretschmer and Peterson (2008) present a thorough review of CGE models integrating bioenergy as well as some discussion of PE models (mostly those focusing on the agriculture sector in the context of transportation fuels).

In the context of bioenergy, CGE models can evaluate the direct and indirect impacts of activities or policy shocks across a variety of sectors and, in the case of global models, across countries. This modeling approach can be useful for analyzing overall impacts of bioenergy but may not have the ability to look at the level of geographic, biophysical, or other sector-specific details that may be necessary. PE models can offer detailed information for the sector(s) being evaluated (in this context, agriculture and forestry sectors) but do not include all sectors of the economy and cannot provide a complete picture of macroeconomic impacts. There are variations between CGE and PE models as well to consider. For example, as Kretschmer and Peterson (2008) pointed out, FASOM-GHG differs from other PE models in that it covers both agriculture and forestry (most other PEs currently focus on one or the other) and includes uses for biomass other than liquid biofuels. Latta et al. (2013) applied FASOM-GHG to assess the market, land use, and GHG implications of biomass electricity expansion. The authors evaluated different CGE and PE models used to model forest-derived biomass use for energy.

#### 3.1.2. Temporal Dynamics

A future anticipated baseline simulation would ideally be based on expectations of future market conditions. To evaluate future biogenic feedstock production trends and potential impacts, a model will need to optimize resource allocation and provide projections over long time horizons. This ability is especially important in order to capture the market and GHG effects of long rotation cycles of forest-derived feedstocks, in addition to other landscape GHG effects that can take decades to unfold (e.g., soil carbon pool impacts, decay rates). The need for long time frames implies use of either intertemporal optimization or recursive dynamic models. Static (single time period) models are not considered for use here, although in certain contexts they could provide key insights.

***Intertemporal optimization models***, or models that optimize over a dynamic interval, can be useful for an anticipated future baseline evaluation. Intertemporal models incorporate expectations of future market conditions, and the model solves over a user-defined planning horizon. Economic agents in intertemporal models are forward looking, and management decisions in the present are based on expectations of current and future market conditions. Intertemporal PE or CGE models require that all markets clear simultaneously for all years in the simulation horizon. For instance, a dynamic forestry sector model would choose rotations, product supply, land management intensity, and equilibrium market conditions in order to maximize economic welfare over the full time horizon.

***Recursive dynamic models*** produce projections 1 year at a time, building on the conditions established in the previous year. For example, land deforested in time period *t-1* would be reflected in the initial forestland endowment in period *t*. Anticipated changes in demand and/or prices can be incorporated such that a model continuously faces new biophysical, technological, or economic environmental parameters in each time step. However, decisions in current time periods are not made with expectations of future conditions in mind, so the decision variables in each time period are static in nature. Given the importance of expectations over long planning horizons for forest investment and management decisions, many existing forest models incorporate intertemporal optimization. However, recursive dynamic models could also potentially be used for analyses of forestry, ideally with key anticipated baseline conditions introduced in the model, and forest management in time *t+1* tied to management decisions in time period *t* (e.g., through the incorporation of additional equations based on expected landowner behavior or linkages to other models).

Both intertemporal optimization and recursive dynamic approaches use idealized scenarios of the future. These scenarios are built to reflect perfect competition and information about future markets and conditions as well as optimized markets (where supply equals demand). It is important to recognize that simulations of future scenarios are meant to provide insights about the potential directionality and, in some cases, magnitude of market responses to an activity, target, or other policy shock. Such approaches are best suited for comparing base optimized solutions (i.e., the BAU) with scenario optimal solutions with a shock (alternative scenarios) in order to look at the market impacts between the model solutions.

## 3.2. Anticipated Future Conditions (Macroeconomic, Biophysical)

Anticipated baseline and alternative scenario analysis include expectations of anticipated macroeconomic and, in the context of accounting of GHG emissions, biophysical conditions. BAU and scenario projections should use published and reputable economic data forecasts, extrapolating only when necessary. Economic projections of population and income exogenously influence demand over time. For example, it makes sense that timber demand is represented not only as a function of price, but also as a function of population and income, and these factors can vary by region. Gross domestic product (GDP) and population estimates can also be introduced to a model to shift the demand curve over time, thus requiring additional timber from the system.[4]

There are several well-known and widely used projections of socioeconomic and physical data to choose from, including those from U.S. EIA and Bureau of Economic Analysis (BEA). Alternatively, one could adopt any number of country-specific projections of key economic variables. At a minimum, simulating an anticipated baseline requires the following economic data:

- GDP projections;
- Population growth projections;
- Demand functions tied to population/income;
- Technological progress assumptions, especially if applicable to stationary sources where efficiency improvements are possible and anticipated;
- Energy market forecasts, especially if energy market data are exogenous to the forestry/land use model; and
- Representation of current and anticipated energy, environmental, or other policies that can constrain BAU trajectories such as renewable portfolio standards (RPS) or the Regulatory Impact Analysis of the Renewable Fuels Standard (RFS2) legislation.

Because some of these data sources are typically only projected over medium time horizons (2035 in the case of the AEO), a consistent method for extrapolating to long-term horizons is required. Defining the relevant time horizon is a determination that will affect the choice of modeling

[4] Note that this could also reduce total demand for timber if regional timber demand is negatively correlated with per capita income for a particular country/region.

procedures to estimate the impacts of using biogenic feedstocks for energy. For woody biomass, a future anticipated baseline methodology would require a long time horizon (40 years or greater) that fully captures forest rotation time frames. For agriculture, a shorter time frame might be sufficient, although a longer time frame would allow one to examine perennial biomass or short rotation woody crops that take several years to reach maturity.

Future anticipated baseline and alternative scenario analysis also requires information about future expected biophysical conditions, such as forestry and agricultural productivity, and landscape and carbon cycle dynamics. Specifically, this information includes parameterization of forest growth curves by class and region as well as biophysical limits to productivity, which can include climate change or environmental conditions (e.g., anticipated shifts in temperature, atmospheric $CO_2$, and water availability). Changing environmental conditions could be reflected in yield curves reflective of natural and anthropogenic disturbance risks, due to climate change, increased urbanization, land use changes, or ecological encroachment. Although climate change projections or natural disturbance risks may not be applicable to all accounting applications, it may be necessary to include them in some instances.

## 3.3. GHG Emissions Representation

Future anticipated baseline modeling requires an assessment of landscape-level emissions changes from an increase or decrease in biogenic feedstock consumption relative to the anticipated baseline. Thus, a modeling framework should directly account for emissions from land management activities across a variety of biogenic feedstock and landscape carbon pools. The choice of modeling framework in large part dictates the complexity available for GHG accounting. The degree of detail in GHG pools as well as the emissions associated with land use and land use change activities depends on what sectors of the economy are included in the model and what degree of detail is available in those sectors. Generally speaking, if a broader economy-wide CGE is used, the GHG accounting will be coarse yet comprehensive across the full macro-economy. If a PE model focusing on specific sectors, such as forestry and/or agriculture, should be employed, the level of detail in GHG pools and emissions consequences of specific land use and land use change actions can be included, but at the expense of comprehensive economy-wide coverage.

## 3.4. Forest Sector Representation

Anticipated future baseline analyses offer the ability to construct and evaluate long-run projections where structural changes are more likely. In the case of forestry, longer-term analysis is often necessary to capture biophysical considerations (e.g., growth rates, rotation periods) as well as the related investment behavior. In the forest sector, decisions about planting and harvest schedules depend in part on expectations about future markets (Sedjo and Tian, 2012).

Forest sector representation should, to the extent possible, be based on observed demand rates for forest products. Future supply and demand can then be determined endogenously through intertemporal optimization under anticipated future macroeconomic conditions. Underlying datasets include historic price and quantity demanded data for forestry products such as timber, pulp/paper, biomass, pellets, etc., as well as demand projections for calibration. Note that, in some instances, it might be preferable to model specific biomass demand assumptions as given in the baseline. The modeling framework should also have comprehensive wood product market representation (including raw timber, pulp/paper, and processed wood products). Ideally, product demand would vary by production region. In addition, the modeling platform should depict growth rates and harvest schedules that vary by region, species, and management regime. Management regime (or intensity) is especially important because intensification of forest stands can alter the carbon intensity of the biomass. Depicting the costs and utilization potential of forest logging and milling residues is another important modeling attribute for future anticipated baseline modeling in this context. Forest residues are an important source of biogenic feedstock supply, and the emissions profile of residues post-harvest may need to be tracked (as discussed in Appendices H and K).

## 3.5. Agricultural Sector Representation

Similar to the forestry sector, the modeling framework should represent agricultural sector production possibilities, markets, and land management options. Output prices modeled endogenously would allow for supply-side responses and reallocation of resources in response to demand shocks. The livestock sector should also be represented, with explicit linkages to the crop sector through the market for animal feed sourced from agricultural crops and by-products. In addition to conventional commodity production possibilities, models should be representative of dedicated energy crop possibilities and crop residuals as potential biomass feedstock sources. This would include representation of the land requirement, costs of harvesting, transporting, and storing the energy biomass prior to combustion.

## 3.6. Land Use Competition

Additionally, the modeling framework should depict land use competition among alternative uses (timberland, cropland, grazing land), allowing for endogenous land use shifts in response to changing market and policy conditions. Furthermore, a model should be able to simulate regional or global supply-side responses to changes in biomass demand, feedstock prices, or renewable energy prices from baseline levels. For instance, the modeling framework would allow for projections to simulate how requiring *X* tons of biomass energy per year in the Southeastern United States might affect forest land use decisions locally, in the Pacific Northwest (regionally), and/or globally. For an economic model, supply-side response potential should be reasonably constrained according to the biophysical nature of the system. That is, although supply might be particularly responsive to price/demand changes, physical constraints such as land availability, biophysical growth capacities, or infrastructure limitations could constrain regional responses to a market change. Developing land constraints that reflect current activities can help account for these factors, though such constraints limit flexibility when projecting into the future. Such functionality allows for evaluation of overall potential landscape land use changes (direct and/or indirect land use change) and related GHG impacts. Latta et al. (2013) highlighted the importance of land use competition between the forestry and agricultural sectors for projecting the potential GHG effects of biomass electricity expansion in the United States.

## 3.7. Energy Sector Representation

Forestry and/or land use models typically do not endogenously capture energy market impacts of different policy scenarios that may affect biomass demand. Forestry and land use models can be excellent tools for estimating biomass supply (and subsequent environmental impacts) under exogenously defined policy scenarios (volumetric mandate or price incentive). However, such models currently do not capture the spillover effects of increased biomass energy demand/consumption in other sectors of the economy (including energy sectors). Without explicitly accounting for these sectors, one cannot reflect the true demand for woody biomass and the competition between biomass and other renewable energy sources under policy-induced shifts from an anticipated baseline. That is, although existing forestry and land use models can simulate biomass production pathways, this may not accurately reflect the fuel mix that will result from a change in policy or energy market conditions. Therefore, it is important to calibrate land use models to projected energy market conditions or existing energy market models.

Also, if a policy or other exogenous change favors investment in a particular non-biomass generation technology, and costs begin to decline because of this investment, the demand for biomass energy would presumably fall. The recent rise in natural gas electricity generation is an example of this scenario and illustrates why modeling deviations from an anticipated baseline in the electricity system are important. If one assumes that biomass demand is tied to the price of renewable energy, then competition among alternative sources matters. This is an important factor that can ultimately affect the estimation procedure for future anticipated baselines in several ways, including those highlighted below.

The rebound effect is an unanticipated policy consequence that has been discussed in the energy economics literature (Greening et al., 2000). Once demand initially declines, the price of fossil energy dips, leading consumers to increase consumption, which reduces the net benefit of the original efficiency improvement. In the case of renewable energy, a policy mechanism that increases the demand for renewable energy can elicit a rebound effect if fossil fuel replacement is high enough to decrease the price of fossil energy.

Another shortcoming of using forestry and land use models with limited energy market interactions is that they likely do not capture competition between competing sources of renewable energy, including different biomass feedstock types. This capability is important because estimation of values for future landscape GHG implications could vary depending on the total amount, and source, of biomass demanded from the system. For instance, if net emissions increase nonlinearly with consumption of a particular feedstock source, it is important to know the total projected demand of biomass energy and the potential emissions impacts. Similar logic flows for systems in which biogenic emissions fall with the level (and price) of bioenergy demanded because of increased terrestrial carbon storage.

## 3.8. International Representation

An international scope allows for the interaction of U.S. forestry and agricultural markets with the rest of the globe, acknowledging international commodity production, demand, prices, and the associated GHG emissions of that production and consumption. Because many forestry and agricultural markets are globally traded, international commodity prices can affect U.S. land use decisions. Conversely, in the case of some commodities where the United States is a large contributor to the global market, U.S. land use decisions affecting commodity supply can impact global prices and thus international land use decisions and related GHG fluxes. Therefore, a global modeling framework would allow for evaluation of international impacts of changes in U.S. biogenic feedstock demand, including emissions leakage effects. Appendix E provides a detailed discussion of leakage and previous literature that has applied international models to project indirect land use emissions from domestic (U.S.) policies.

## 3.9. Consideration of Scenario Development

In addition to depicting a future anticipated BAU baseline, the modeling framework should be able to easily calibrate to alternative future scenarios for simulation analysis. That is, if an assessment needs to evaluate the GHG emissions effects of a deviation from anticipated baseline woody biomass harvest regime, what is the appropriate projected baseline and how should an alternative scenario be designed such that a comparison between the two will provide the necessary insights? Numerous examples for how to model future anticipated baselines and alternative scenarios exist, including:

- Define an anticipated baseline with a projected amount of biomass energy demand and compare this against an alternative scenario where total projected biomass energy demand is different (higher, lower, or no biomass consumption) relative to the baseline, holding everything else constant.
- Similarly, define an anticipated baseline with a projected amount of biomass energy demand, but compare that amount against alternative scenarios that exhibit changes (higher or lower) in projected demand for a *single feedstock* (e.g., roundwood) relative to the level of consumption for that feedstock in the anticipated baseline.
- Model a major new policy and compare it to the baseline. This shift could include an aggressive GHG reduction strategy, such as a carbon tax, renewable energy portfolio standard (RPS), or clean energy standard (CES).

Regardless of the scenario implemented, calculation of results will focus on the difference between the future anticipated baseline and the alternative scenario. Furthermore, the chosen modeling approach could vary depending on the alternative scenario evaluated.

For example, one scenario could require an RPS-type policy framework in which a minimum percentage of electricity must be met through renewables. In this example, an energy model could project the proportion of biomass used to help meet the policy-mandated renewable energy demand. Forestry sector models would use this information to determine the final feedstock mix resulting from this demand from the energy sector. Alternatively, one could develop policy scenarios that mandate specific amounts of individual bioenergy feedstocks, and then focus on the GHG implications of each, one at a time. The first approach is somewhat consistent with recently published modeling efforts (Daigneault et al., 2012; Galik and Abt, 2012; Mosnier et al., 2012; Latta et al., 2013), while the latter is consistent with RFS2 legislation. For the purposes of this framework, there is not a specific policy being modeled, nor is there a specific price path (e.g., for energy or $CO_2$ emissions allowance) or renewable energy target to be met. Therefore, the approach described below differs somewhat from most previous studies in its construction of baselines and alternative scenarios.

# 4. Modeling Approach and Tools Chosen for Illustrative Framework Applications

The most appropriate modeling approaches and level of detail depend largely on the goals of the assessment. It is important to assess candidate models against considerations appropriate for that application. Ideally, the modeling approach should handle all of the complex interactions necessary to address this complicated issue (many of which have been discussed in previous sections). However, the reality is that a single ideal model does not currently exist. All existing models have various advantages as well as shortcomings that must be taken into account, and the use of one model over another implies trade-offs.

The remainder of this appendix explains the basic methodology developed for this report, focusing on the model attributes used to evaluate biogenic emissions, and the choice of a land use optimization model calibrated to existing energy market projections for this assessment.

This study first uses the most recent biomass consumption rates for energy generation at electricity sector and some industrial sector stationary sources available through the EIA and Department of Energy (from the EIA-923 database, in short dry tons [DOE, 2011]). This analysis then applies the FASOM-GHG model, because it includes most of the key functions and components described in the above section (these functions are described in Section 4.3).

Although FASOM-GHG is not explicitly tied to an energy sector model, biomass projections and energy market assumptions are calibrated to the AEO 2012 forecasts, as described in brief below and in detail in Appendix K that discusses the future anticipated baseline construction methods.

The next subsections briefly describe the anticipated future baseline scenarios and the alternative biogenic feedstock production scenarios to provide context for the application of FASOM-GHG. Details on the baseline scenario development can be found in Appendix K and on the alternative scenario case study applications in Appendix L. The following section describes the key components and capabilities of the FASOM-GHG model, followed by a subsection focusing on specific datasets and functions within FASOM-GHG that are pertinent to this report. Full FASOM-GHG documentation is available online.[5]

## 4.1. Brief Overview of Baseline Construction

Although there are numerous AEO model scenarios to choose from, the alternative baselines in this study were developed using four AEO 2012 projections and two additional cases:

[5] http://www.cof.orst.edu/cof/fr/research/tamm/forest_and_agriculture_sector_op.htm

- **Reference case:** The Reference case is the baseline AEO 2012 model, which assumes real GDP grows at a 2.4% average annual rate from 2008 to 2035, buoyed by a 1.5% per year growth in productivity in nonfarm businesses and a 0.6% growth in nonfarm employment. All the other scenarios pivot off of the Reference case scenario, changing specific assumptions.
- **High GDP Growth case:** The AEO 2012 High Economic Growth case assumes that real GDP grows by 3%, supported by productivity growth of 2.4% and employment growth of 1.2%.
- **Low GDP Growth case:** The AEO 2012 Low Economic Growth case assumes that real GDP grows by 1.8%, supported by productivity growth of 1.5% and employment growth of 0.5%.
- **Low Renewable Technology Cost case:** The AEO 2012 Low Renewable Technology Cost case assumes annual levelized cost for non-hydropower renewables is 10% lower than the Reference case in 2010 and is 35% lower by 2035 compared with Reference case values.
- **Zero Biomass case:** A constraint is imposed that restricts all biomass consumption for energy generation in each region.
- **Constant Biomass case:** The Constant Biomass scenario begins with total biomass demand constraints set to 2009 consumption levels.

Using these scenarios to derive biogenic feedstock demand projections for stationary sources, regional constraints are imposed requiring supply-side utilization of biomass for energy generation that matches these projections exactly. No restrictions are imposed on regional feedstock mixes used to meet these overall biomass requirements. The model minimizes the costs of providing the requisite biomass. It is important to acknowledge that this study uses the 2009 EIA AEO information to be consistent with the other databases used (the most recent Emissions and Generation Resource Integrated Database [eGRID] at the time of this work was from 2009).

Most dynamic models, particularly energy sector models, are calibrated to existing economic forecasts. This analysis adopts key forecasts from the AEO. Thus, anticipated market and policy conditions are consistent with assumptions underlying the AEO and the National Energy Modeling System (NEMS). Energy price projections are calibrated to the AEO 2012 to reflect anticipated energy market conditions. Biogenic feedstock consumption projections are also calibrated to growth rates in renewable energy demand in the industrial and electricity sectors. An advantage of calibrating scenarios to the AEO is that existing state policies (RPS, CES, etc.) are already accounted for to the extent possible. Thus, growth parameters for renewable energy demand used in this analysis assume that state-level policies encouraging growth in renewable electricity will hold. Additional discussion on the specific scenarios chosen for this report can be found in Appendix K.

## 4.2. Brief Overview of Regional Feedstock Case Studies

The following illustrative case studies are developed to focus on the net biogenic $CO_2$ effects of an increase in biogenic feedstock consumption for a specific feedstock within a particular region, relative to the future anticipated future baseline. These case studies, representing different feedstock types and different regions in the United States in order to illustrate regional and feedstock differences, offer insight into the potential landscape emissions impact of increased consumption of a single biogenic feedstock:

- Southeast roundwood;
- Corn Belt corn stover; and
- Pacific Northwest logging residues.

In each region, the demand for the specified feedstock is increased 1 million tons per year above biomass demand in the AEO Reference case level by 2030. For example, roundwood demand in the Southeast increases by 1 million tons in relation to the AEO-based baseline. All feedstock scenario results are calculated relative to the AEO Reference case and Zero Biomass case. Additional information on how these case study scenarios were developed and executed in FASOM-GHG, as well as variations on these case study evaluations, can be found in Appendices K and L. Some of the feedstock-specific case study sensitivities were chosen to maintain consistency with the reference point case studies evaluated in this report and are reported in Appendix L.

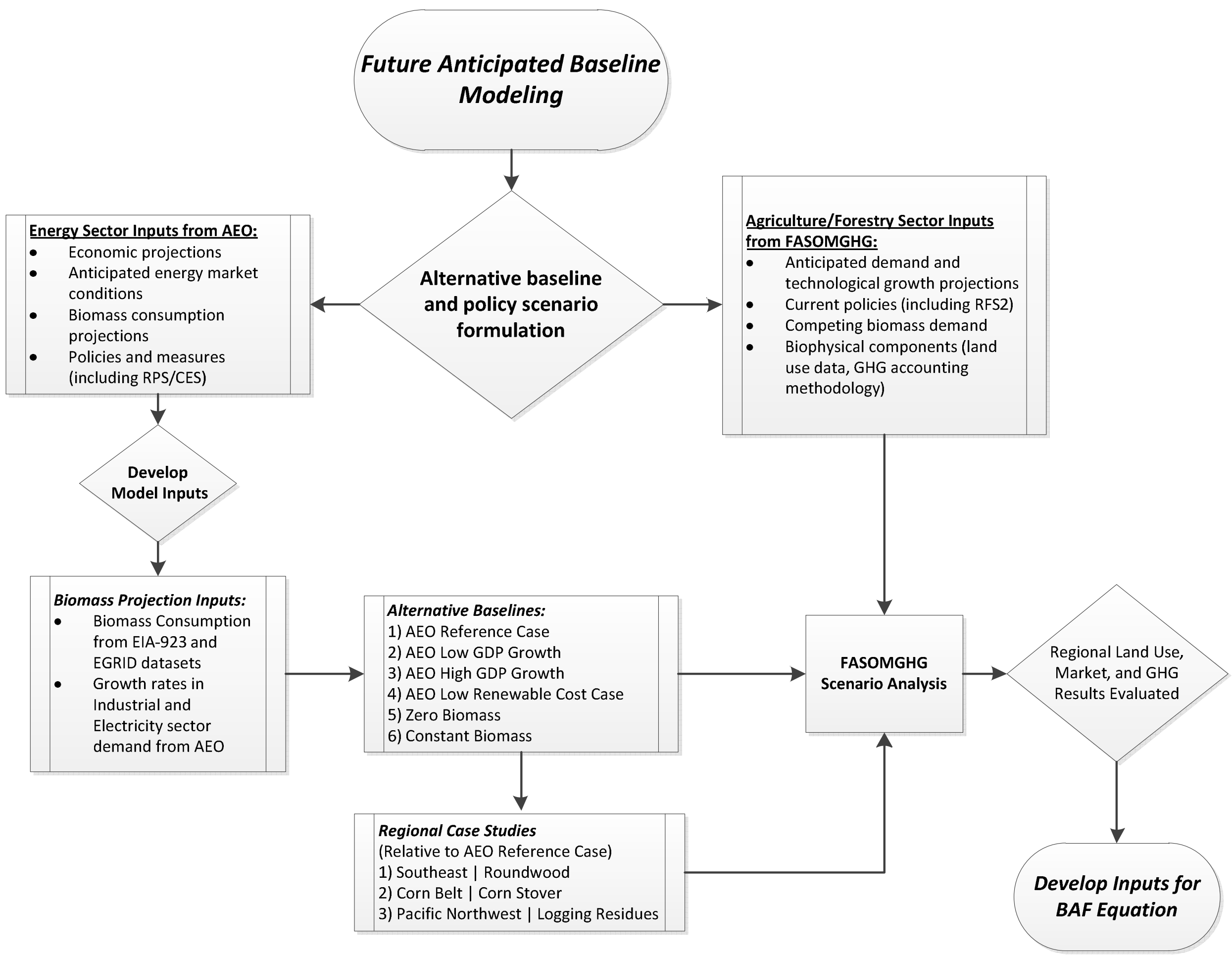


**Figure 1. Conceptual Diagram Outlining the Basic Anticipated Future Baseline Modeling Approach Presented in this Appendix.**

## 4.3. Use of FASOM-GHG

FASOM-GHG has the basic necessary capabilities and characteristics to satisfy the requirements of this study. Table 1 highlights these abilities, which are discussed in detail in the following sections.

**Table 1. FASOM-GHC Attributes.**

| Attribute | FASOM-GHG |
|---|---|
| Model framework/function type | Intertemporal partial equilibrium |
| Anticipated future conditions | AEO 2012 energy market conditions, USDA baseline (2010), and 2005 RPA Assessment |
| GHG representation | $CO_2$, $CH_4$, and $N_2O$ accounting across forest, crop, and livestock management activities; $CO_2$ accounting includes biogenic feedstock (forest carbon) and non-biogenic feedstock (soil) pools |
| Forest sector representation | Logs from timber harvest and secondary wood products, forest residues |
| Agricultural sector representation | 40 primary crop commodities; 25 primary livestock products; 32 domestic and imported forest logs; 12 categories of forest and agricultural residues; 17 secondary crop products; 17 secondary livestock products; 10 processing by-products; 40 processed forest products |
| Land use competition | Endogenous competition between cropland, forestland, and grazing lands |
| Energy sector representation | Ethanol (first and second generation), biodiesel, and biopower (100% biomass generation or co-firing levels of 5%, 10%, 15%, and 20%) with exogenous price or quantity constraint |
| International representation | 18 regions for seven agricultural traded commodities; forest sector includes endogenous activities for trade with Canada as well as other significant trade flows (e.g., softwood lumber trade with non-Canadian regions) |

### 4.3.1. Overview of Key FASOM-GHG Attributes and Functions

This analysis applies an updated and enhanced version of FASOM-GHG. FASOM-GHG is a dynamic partial equilibrium economic model of the U.S. agricultural and forestry sectors and has been applied in a wide range of policy settings. FASOM-GHG explicitly models GHG mitigation strategies, including many bioenergy processing options (Murray et al., 2005; Schneider and McCarl, 2003).

FASOM-GHG uses a price-endogenous mathematical programming approach developed by Judge and Takayama (1973) and McCarl and Spreen (1980). The model maximizes total intertemporal welfare across the U.S. agricultural and forestry sectors, or the sum of producer surplus (area below the equilibrium price) and consumer surplus (area above the equilibrium price). Commodity and most factor prices are endogenous, determined by the supply and demand relationships in all markets included within the model. The framework accounts for market adjustments over time to systematic policy shocks by depicting changes in equilibrium prices and quantities supplied of all primary and secondary commodities. Because commodity markets within agriculture and forestry are highly interdependent, a systematic shock that disrupts the optimal production portfolio of one commodity (e.g., corn) can cycle through other primary or secondary commodity markets (such as ethanol and livestock, which use corn as a critical factor input, or corn substitutes such as alternative feed grains), through competition for production inputs, consumers, trading partners, and land.

FASOM-GHG accounts for a comprehensive range of land use categories consistent with land classifications from multiple resources including the Natural Resources Inventory (NRCS, 2003), Major Land Use Database (USDA-ERS, 2010), and agricultural census (USDA-NASS, 2010). The model allows for explicit land use competition between cropland, grazing lands, and conservation lands (CRP) and forestland based on expected returns to alternative uses. This allows us to simulate potential land use change impacts of policy drivers that increase the relative value of land holdings in a particular use over time (Alig et al., 1998, Alig et al., 2010) and is a departure from the static CGE modeling approach that assumes conversion costs or elasticities of substitution between alternative land uses.

FASOM-GHG is disaggregated into 63 minor production regions in the lower 48 states and 11 main agro-forestry regions. Table 2 displays all major regions with accompanying production units. All major regions include crop and forestry production opportunities except for the Great Plains and Southern Plains (which includes most of Texas and Oklahoma). Land use change between forestry and agriculture is restricted to lands that fall within a certain land suitability class, thus ensuring that land transfers remain within realistic bounds (Alig et al., 2010).

**Table 2. Definition of FASOM-GHG Production Regions and Market Regions.**

| Key | Market Region | Production Region (States/Subregions) |
|---|---|---|
| NE | Northeast | Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, West Virginia |
| LS | Lake States | Michigan, Minnesota, Wisconsin |
| CB | Corn Belt | All regions in Illinois, Indiana, Iowa, Missouri, Ohio (IllinoisN, IllinoisS, IndianaN, IndianaS, IowaW, IowaNE, IowaS, OhioNW, OhioS, OhioNE) |
| SE | Southeast | Virginia, North Carolina, South Carolina, Georgia, Florida |
| SC | South Central | Alabama, Arkansas, Kentucky, Louisiana, Mississippi, Tennessee, Eastern Texas |
| SW | Southwest (agriculture only) | Oklahoma, all of Texas but the eastern part (Texas High Plains, Texas Rolling Plains, Texas Central Blacklands, Texas Edwards Plateau, Texas Coastal Bend, Texas South, Texas Trans Pecos) |
| RM | Rocky Mountains | Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah, Wyoming |
| PSW | Pacific Southwest | All regions in California (CaliforniaN, CaliforniaS) |
| PNWE | Pacific Northwest-East side | Oregon and Washington, east of the Cascade mountain range |

| | | |
|---|---|---|
| PNWW | Pacific Northwest- West side (forestry only) | Oregon and Washington, west of the Cascade mountain range |


**Figure 2. Map of the FASOM-GHG Regions (Source: Beach et al., 2010b).**

Land to development transfers are modeled on a regional basis by land type and drawn from data prepared for the 2010 Resources Planning Act (RPA) Assessment (Alig et al., 2009). These parameters help depict an anticipated future land base that is decreasing in the baseline due to development pressures. Accounting for land to development pressures in an agriculture and forestry sectoral modeling framework is important because varying levels of development pressures can affect land use competition between agriculture and forestry, GHG mitigation potential, and commodity prices (see Alig et al., 2010 for additional discussion).

FASOM-GHG encompasses a suite of GHG mitigation options, including biological sequestration of carbon in agricultural soils and forest stands, alternative crop and livestock production practices to reduce emissions, and bioenergy feedstock substitutes for fossil fuels. The gases represented are carbon dioxide, methane, and nitrous oxide. Forest carbon balances are tracked using a methodology consistent with the Forest Carbon accounting system, FORCARB (Birdsey et al., 2000).

Forest carbon is tracked in trees, soils, understory, and end products. Forest management offset opportunities are endogenously modeled in FASOM-GHG and include avoided deforestation, rotation extensions, altered species mix, partial thinning, and reforestation. For a discussion of GHG accounting and mitigation options, as well as forestry and agricultural management options in FASOM-GHG, see Beach et al. (2010b).

The model allows for intensive and extensive margin shifts for both crop and forestry production activities (as discussed in Baker et al., 2013). Furthermore, land use competition and product substitution between the two sectors is a key model component, missing from other partial equilibrium models of the agriculture and forestry sectors. The inclusion of such a function has been found to have a dramatic impact on GHG emissions trajectories relative to less inclusive modeling approaches (Latta et al., 2013). Additional information on how FASOM-GHG depicts intensive and extensive margin production opportunities in forestry and agriculture can be found in Beach et al. (2010a) and Adams et al. (2008).

FASOM-GHG incorporates endogenous international trade effects, such as international supply regions (18 regions) for seven agricultural traded commodities with import supply functions (Adams et al., 2008). The forest sector includes endogenous activities for virtually all forms of trade with Canada as well as other significant trade flows to offshore regions (e.g., softwood lumber trade with non-Canadian regions). Details on FASOM-GHG's international components are discussed in the supplemental online documentation (Beach et al., 2010b; Adams et al., 2008). FASOM-GHG cannot conduct detailed analysis of global GHG impacts of changes in U.S. biogenic feedstock production and consumption. However, the model could be linked with a global model, including forestry and agricultural trade components with related land use and GHG accounting components.

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# Appendix K: Future Anticipated Baseline Construction: Methodology and Results

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# 1. Introduction

This appendix describes a methodology for constructing alternative future anticipated baseline scenarios that can be used to evaluate the potential net atmospheric contribution of biogenic carbon dioxide ($CO_2$) from increased consumption of biogenic feedstocks at stationary sources. The purpose of this analysis is to illustrate how landscape $CO_2$ balances (emissions fluxes net of carbon sequestration in biogenic feedstocks and soils) could respond to changes in land management associated with alternative biogenic feedstock demand projections, and how baseline formation can affect emissions projections estimates.

Biogenic feedstock consumption will likely grow over time as the demand for renewable electricity increases (driven in part by state renewable portfolios, clean energy standards or other policies and incentives). Thus, it is important to consider anticipated growth in stationary source biogenic feedstock demand in addition to current consumption levels. Using a compilation of different energy sector datasets as inputs to a dynamic land use model, several potential future anticipated baseline scenarios are constructed to project biogenic $CO_2$ emissions from the U.S. forest and agricultural sectors as well as emissions intensity values for biogenic feedstock consumption for electricity generation at stationary sources. These potential future baseline scenario projections are developed to show a range of potential future conditions, illustrating how baseline scenario projections can be sensitive to different macroeconomic inputs.

The first section of this appendix discusses how current biogenic feedstock consumption estimates are combined with regional energy market projections to generate six alternative future anticipated baseline scenarios, representing alternative biogenic feedstock demand trajectories. Next, the U.S. Forest and Agricultural Sector Optimization Model with Greenhouse Gases (FASOM-GHG) is used to simulate future biogenic $CO_2$ emissions fluxes estimated for each alternative baseline scenario. These projected emissions trajectories are then compared with a projected future with no biogenic feedstock consumption for electricity generation. Then, results from the alternative future baseline scenarios are used to project cumulative landscape emissions associated with each baseline's biogenic feedstock consumption. This appendix concludes with a discussion of key findings, uncertainties, and limitations. Two baseline scenarios from this appendix are then used for further application and analysis in the future anticipated baseline case studies (Appendix L).

# 2. Methodology for Projecting U.S. Biogenic Feedstock Consumption Scenarios

A prospective analysis of $CO_2$ emissions from biogenic feedstock consumption at stationary sources requires two primary pieces of information: current and anticipated future biogenic feedstock usage. The primary data sources used to estimate current facility-level biomass energy consumption are EIA-923 Annual Electric Utility data from December 2009.[1] This information serves as the basis for developing projections using data derived from the Energy Information Agency's (EIA's) Annual Energy Outlook (AEO) models from 2012. This section provides the methodology and presents estimated current and future biogenic feedstock consumption under five alternative future scenarios. One of the outcomes for this section is a table representing current biogenic feedstock consumption for both forest- and agriculture-derived feedstock types delineated according to regions appropriate for use in the FASOM-GHG model.

## 2.1. Estimates of Current Consumption

To arrive at the biogenic feedstock consumption estimates used for this analysis, three basic steps were required. In Step 1, the December 2009 version of the Form EIA-923 survey representing current facility-level data was queried for total biogenic feedstock consumption at industrial, electricity, and commercial stationary sources. Step 2 involved filtering the data to remove biogenic feedstocks such as black liquor and municipal solid waste, which are not included in the FASOM-GHG model. Finally, in Step 3, common plant ID codes were obtained for each EIA-923 power generation unit by matching the units to EPA's Emissions & Generation Resource Integrated Database (eGrid) 2009 database to obtain latitude and longitude coordinates. These coordinates were then used to map the stationary sources to the 11 primary FASOM-GHG agroforestry regions.

### 2.1.1. Step 1: Querying EIA-923 Feedstock Consumption Data

EIA-923 contains detailed monthly and annual electric power data on electricity generation, fuel consumption, fossil fuel stocks, and receipts at the stationary source level (EIA, 2012). The dataset contains information on the feedstock type used as well as the different generation processes. Specifically, the data splits plants into three sectors: electricity, industrial, and commercial. Electricity sector entities use biogenic feedstocks to generate electricity for an external electric grid. Industrial sector entities, such as pulp and paper mills, use biomass for internal industrial production processes and electricity generation purposes with residual bioelectricity sold back to the grid. Finally, commercial sector entities are primarily small-scale electric generators, burning

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[1] The EIA-923 database is updated annually. The 2009 dataset is used for this analysis to represent starting conditions for the "2010" simulation period in FASOM-GHG.

biomass to supply electricity to a single installation. A hospital with a boiler that co-fires biomass is a good example of a commercial plant.[2]

In addition to categorizing biogenic feedstock demand by the electricity and industrial sectors, EIA-923 further disaggregates consumption by specific biogenic feedstock sources, as shown in Table K-1. Biomass-derived energy can come from a multitude of feedstocks, including raw biomass sources, waste streams, by-products of silvicultural practices and/or agricultural cultivation, or by-products of industrial processes.

**Table K-1. Description of Biomass Sources in EIA-923.**

| | EIA 923 Code | Biomass Description |
|---|---|---|
| Solid Renewable Fuels | AB | Agricultural crop by-products/straw/energy crops |
| | MSB | Municipal solid waste—biogenic component |
| | OBS | Other biomass solids |
| | WDS | Wood/wood waste solids (paper pellets, railroad ties, wood chips, etc.) |
| Liquid Renewable Fuels | OBL | Other biomass liquids |
| | BLQ | Black liquor |
| | SLW | Sludge waste |
| | WDL | Wood waste liquids excluding black liquor |

Figure K-1 provides estimates of current biogenic feedstock consumption by biomass sources for each of the three sectors with the industrial sector further disaggregated to differentiate pulp and paper from other industrial entities. Electric utilities consumed the most biomass in 2009, more than 40 million short tons, with the largest share coming from wood solids and municipal solid waste. Pulp and paper mills are the second largest consumers of biomass for energy, with the majority of consumption coming from black liquor and wood solids that are forest-derived industrial by-products of pulp and paper production processes.

[2] This analysis focuses on the major biomass consuming industries, thus, commercial facilities such as hospitals and schools are dropped from the underlying dataset and this analysis.

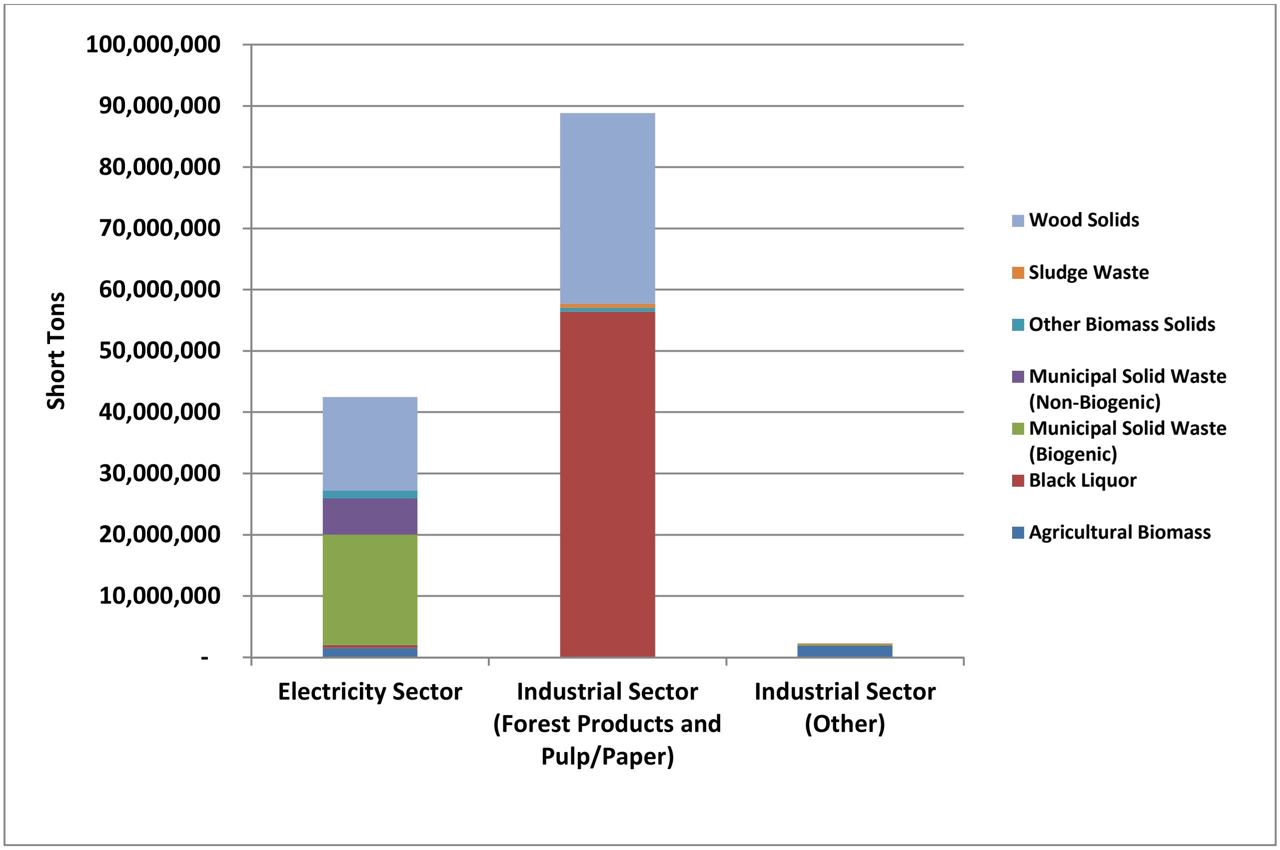


**Figure K-1. Biogenic Feedstock Consumption for Energy Generation (Short Tons) by Sector and Source in 2009 (Source: EIA-923).**

### 2.1.2. Step 2: Filtering the Data for FASOM-GHG Biogenic Types

With the EIA-923 biogenic feedstock consumption for 2009 identified, it is now necessary to process the data so that these can be used as part of the FASOM-GHG modeling approach. This means that biogenic feedstocks handled in other parts of the framework and those not included in FASOM-GHG, as well as coinciding stationary sources, must be filtered out of the dataset.

First, municipal solid waste and other waste-derived feedstocks receive a different treatment than forestry and agriculture-derived biogenic feedstocks (see Appendix N). Second, the FASOM-GHG model does not currently depict black liquor, an industrial processing byproduct from pulp and paper milling. Thus, for the purposes of this study, the estimates of current biogenic feedstock consumption are restricted to the following EIA-923 biomass types: agricultural crop by-products, straw, and energy crops (AB); wood and wood waste solids (WDS); and other biomass solids (OBS). This subset of biomass represents forest and agricultural biomass and excludes any liquids or municipal solid waste used for electricity generation; it thus accounts for approximately 37% of all biogenic feedstocks currently consumed for energy generation.[3]

[3] It is important to remember that because of the data filtering necessary for this specific study, the biogenic feedstock consumption projections provided in subsequent sections are lower than AEO or other bioenergy

Next, stationary sources that use these biogenic feedstocks that have been filtered out of the dataset are removed, eliminating all of the sources in the commercial sector, and resulting in two remaining power generation sectors: the electricity sector and the industrial sector. For the purposes of this study, the electricity sector dataset was created by excluding any stationary source that does not have "electric utility" as the sector name or does not have the appropriate NAICS code 22. The industrial sector dataset excludes any stationary source that does not have "industrial NAICS cogen" or "industrial NAICS non-cogen" as the sector name. This 2009 feedstock consumption estimate represents the base-level value from which all projections presented for the future anticipated baseline approach in this and related appendices (Appendix L) are simulated.

Figures K-2 and K-3 provide a geographic depiction of biogenic feedstock consumption in short tons in the electricity and industrial sectors, respectively, in 2009. Biomass consumption in the electricity sector is primarily confined to the Northeast, Florida, California, and Minnesota. Within the industrial sector, biomass is consumed mostly in the Southeast, Northeast, and Pacific Northwest, where much of the pulp and paper industry is located. These 2009 consumption rates are what determine the "Constant Biomass Consumption case" scenario discussed in subsequent sections.

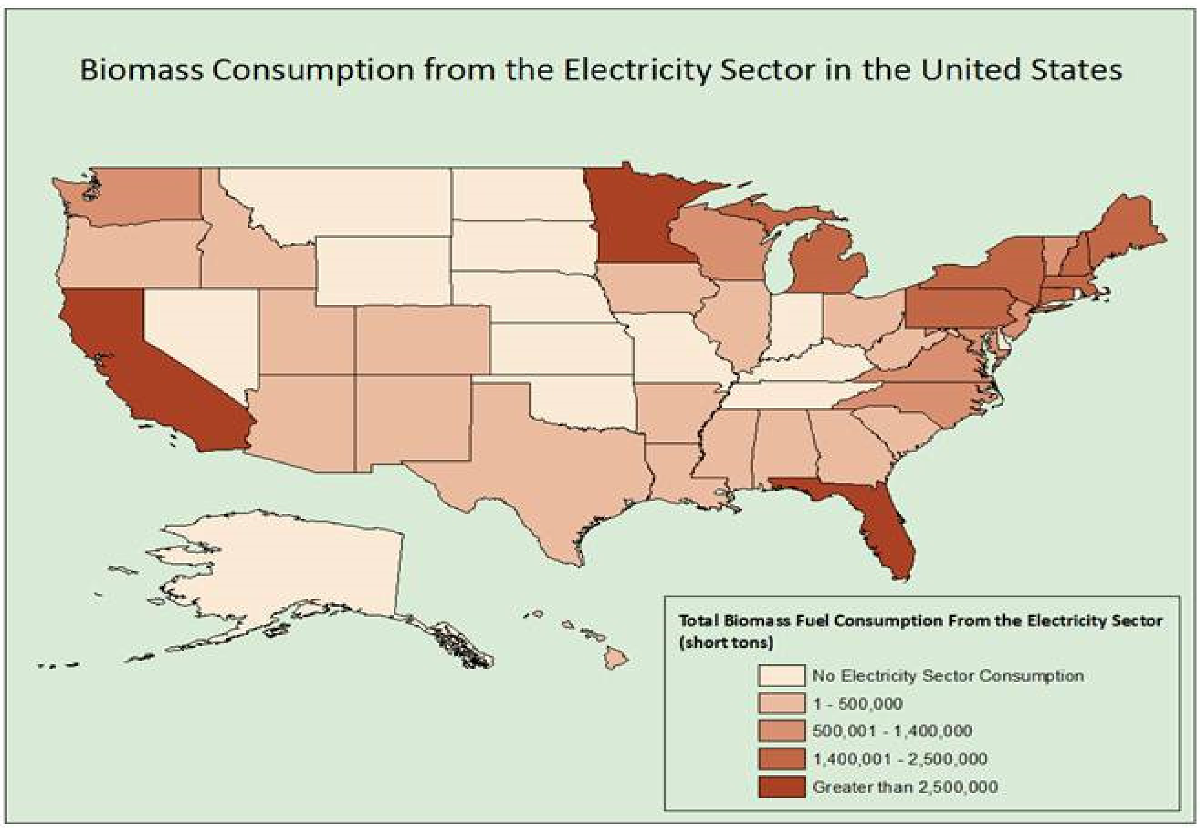


**Figure K-2. Biogenic Feedstock Consumption by Electric Utilities in 2009 (Source: EIA-923).**

projections that would include municipal solid waste and other important biogenic feedstock types.

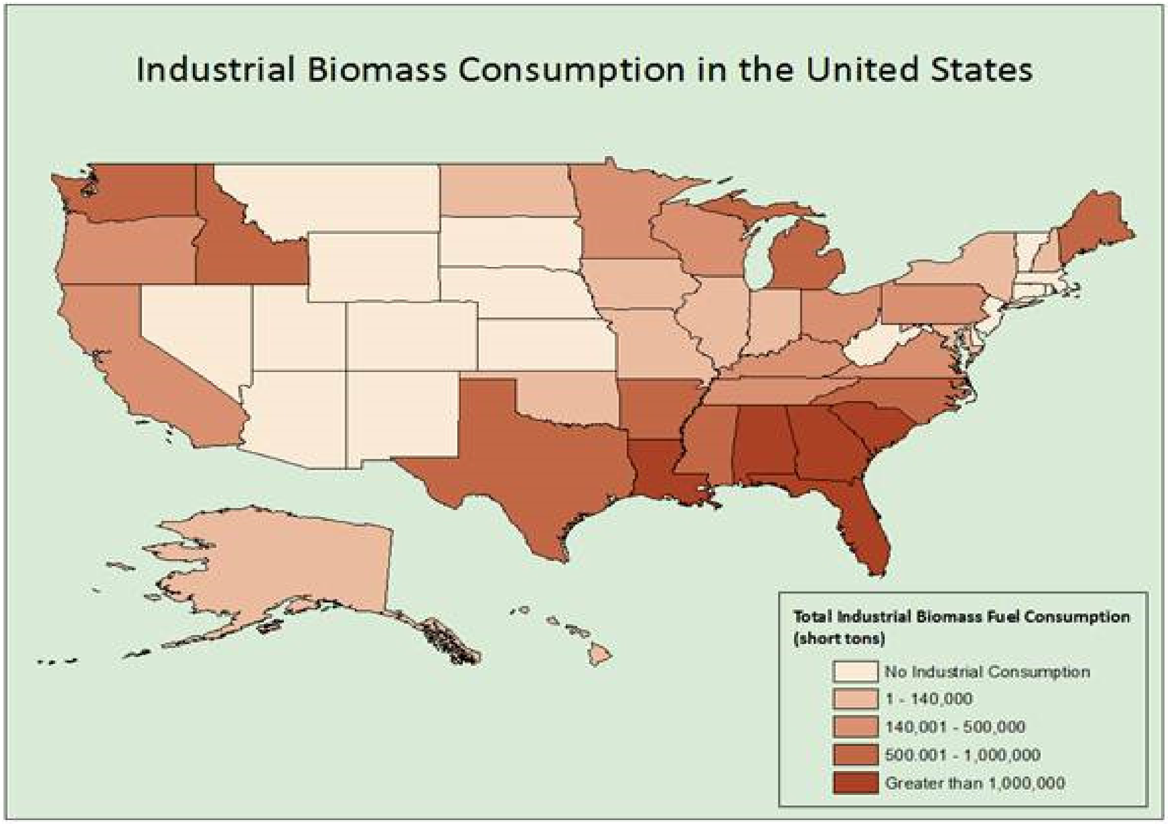


**Figure K-3. Total Biogenic Feedstock Consumption for Electricity Generation at Stationary Sources in the Industrial Sector in 2009 (Source: EIA-923).**

### 2.1.3. Step 3: Mapping EIA-923 Data to FASOM-GHG Regions

Once the biogenic feedstock types and sectors were filtered, EIA 923 data were merged with FASOM-GHG regions. EIA-923 data are collected at the stationary source level, but the geographic coordinates are not published. However, EPA's publicly available eGRID 2009 database uses EIA-923 data and includes a common plant ID code to link the data sources as well as latitude and longitude coordinates for the stationary sources included in this analysis. For industrial sector sources, only forest product and paper manufacturing facilities are included. Figure K-4 displays the EIA-923 biogenic feedstock consumption data at the eGRID stationary source locations overlaid with a map of the FASOM-GHG regions.[4]

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[4] **Region Definitions for Figure K-4:** CB = Corn Belt; GP = Great Plains; LS = Lake States; NE = Northeast; PNWE = Pacific Northwest East; PNWW = Pacific Northwest West; PSW = Pacific Southwest; RM = Rocky Mountains; SE = Southeast; SC = South Central; SW = Southwest

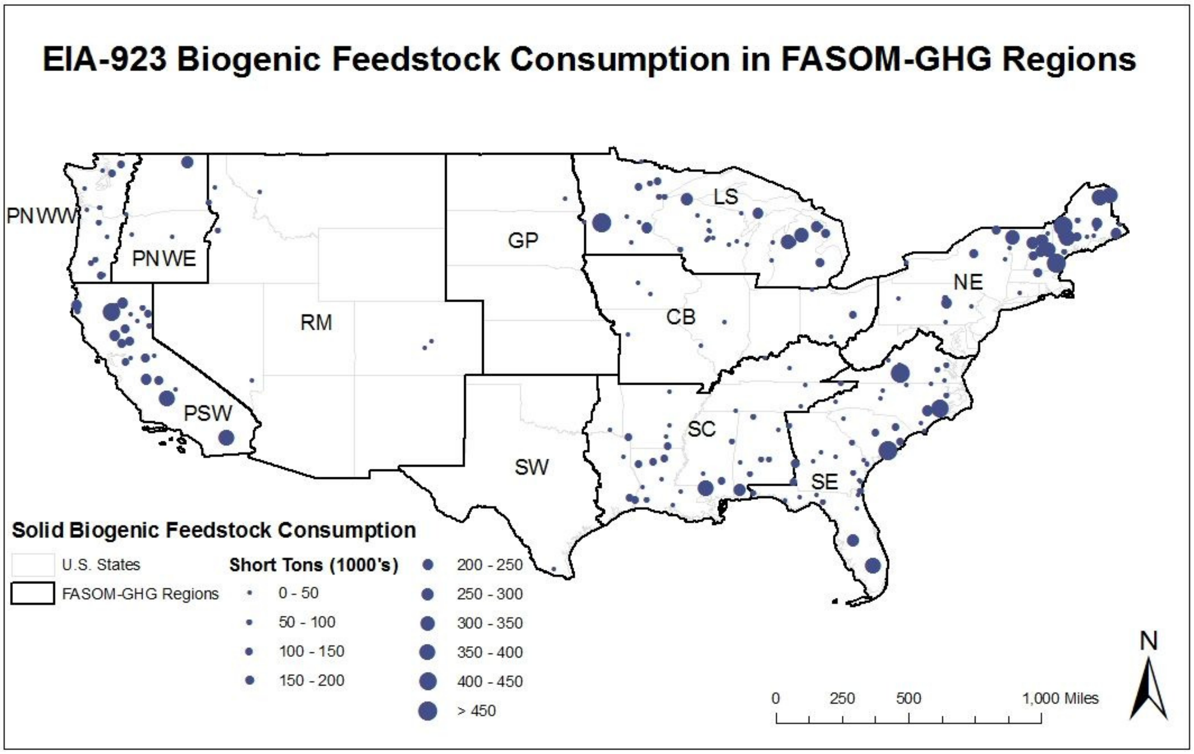


**Figure K-4. Map of EIA-923 Biogenic Feedstock Consumption at Electric and Industrial Plants and FASOM-GHG Region.**

Table K-2 provides the results of the data processing exercise by sector and FASOM-GHG region in addition to percentages of agricultural versus nonagricultural biogenic feedstock categories consumed within the combined electricity and industrial sectors. The majority of biogenic feedstocks used (once forest and agricultural biomass sources have been filtered out of the EIA-923 data)—96.7% of U.S. solid biomass feedstock consumption—originates from nonagricultural sources. Thus, forest-derived biomass in solid and liquid form makes up the majority of biomass consumption at stationary sources in the filtered EIA-923 dataset. Forest biomass includes the categories “Roundwood” and “Logging Residues,” plus a set of “Forest Derived Industrial Products or Processing By-products,” as defined in Appendix D. Agricultural feedstocks include the categories “Conventional Agricultural Crops”, “Dedicated Energy Crops,” and “Agricultural Crop Residues,” which are also defined and discussed in Appendix D.

## 2.1. Estimates of Future Consumption

EIA’s AEO focuses on factors that influence the U.S. energy system in the long run (energy demand, supply, and prices). AEO projections assume that current laws and regulations remain unchanged throughout the projections, unless explicitly changed for a policy scenario case (for instance, certain AEO scenarios include GHG mitigation policies, including $CO_2$ emissions allowance fees). These laws and regulations include mandatory state renewable or clean energy standards, which are applied to the underlying model used to produce the AEO (the National Energy Modeling System) to the extent possible.

**Table K-2. EIA-923 Biogenic Feedstock Consumption by Sector and FASOM-GHG Region as well as Regional Proportion Derived from Woody Biomass (EIA-923, 2009).**

| FASOM Region | Electric Fuel Consumed (short tons) | | | Percentage Derived from Woody Biomass |
|---|---|---|---|---|
| | Nonelectric Stationary Sources | Electric Stationary Sources | Total | |
| **CB** | 192,185 | 73,292 | 265,477 | 95% |
| **GP** | 3,477 | 4,308 | 7,785 | 0% |
| **LS** | 525,588 | 3,151,535 | 3,677,123 | 99% |
| **NE** | 470,644 | 5,336,594 | 5,807,238 | 100% |
| **PNWE** | 8,437 | 290,853 | 299,290 | 100% |
| **PNWW** | 437,863 | 326,206 | 764,069 | 100% |
| **PSW** | 312,445 | 3,629,679 | 3,942,124 | 95% |
| **RM** | 91,502 | 229,078 | 320,580 | 100% |
| **SC** | 2,170,840 | 363,043 | 2,533,883 | 96% |
| **SE** | 1,739,475 | 2,556,641 | 4,296,116 | 92% |
| **SW** | 162,325 | 0 | 162,325 | 0% |
| **U.S. Total** | 6,114,781 | 15,961,229 | 22,076,010 | 97% |

The AEO projections also include key macroeconomic factors that significantly influence the energy market, including population and gross domestic product (GDP). According to the AEO website, the AEO Reference case "provides the basis for examination and discussion of energy production, consumption, technology, and market trends and the direction they may take in the future. It also serves as a starting point for analysis of potential changes in energy policies." In addition to the Reference case, EIA presents a number of other alternative cases to illustrate uncertainties associated with the Reference case projections.

The same exercise is done in this study. To account for uncertainty in future anticipated biogenic feedstock consumption, multiple anticipated future baseline scenarios are developed that calibrate directly to AEO 2012 scenarios (discussed below). The 2012 AEO projections used in this report are carried out until 2035, and all biogenic feedstock consumption beyond this period is held constant in FASOM-GHG simulation periods beyond 2035.

AEO scenario projections are used to build biogenic feedstock consumption trajectories off of the 2009 feedstock consumption values calculated for this analysis. There are numerous AEO scenarios available from EIA, with deviations in economic growth assumptions, policy variables, and fuel prices (the AEO 2012 report included 29 total scenarios): the discussion here focuses on the following four baseline scenarios: Reference, High Economic Growth, Low Economic Growth, and Low Renewable Technology Cost. In addition to the AEO scenarios, a fifth baseline scenario was developed in which 2009 biogenic feedstock consumption levels are held constant. The Reference case is the baseline AEO (2012) scenario, which assumes real GDP grows at a 2.4% average annual rate from 2008 to 2035, buoyed by a 1.5% per year growth in productivity in nonfarm businesses and 0.6% growth in non-farm employment. All other AEO baseline scenarios pivot off this Reference baseline scenario by changing specific assumptions. The High Economic Growth baseline assumes that real GDP grows by 3%, supported by productivity growth of 2.4% and employment growth of 1.2%. The Low Economic Growth baseline assumes that real GDP grows by 1.8%, supported by productivity growth of 1.5% and employment growth of 0.5%. The Low Renewable Energy Technology Cost baseline assumes annual levelized cost for non-hydropower renewables is 10% lower than the Reference baseline in 2010 and drops 35% by 2035 compared to Reference baseline values.

To generate biogenic feedstock consumption projections for each AEO baseline scenario, index variables were created using 2009 as the base year; these values reflect the rate of growth in projected renewable electricity consumption by Electricity Market Module (EMM) region in quadrillion British thermal unit (Btu). For industrial sector stationary sources, growth rates are equal to the change in industrial sector renewable energy consumption by EMM region. For electricity sector stationary sources, the change in total renewable electricity generation by EMM region is used. Note that this assumes that the proportion of biomass energy to total renewable energy would stay constant over time. Thus, it does not factor in potential declining costs of alternative renewable energy technologies such as wind, solar, or geothermal, and such declining costs could reduce the share of renewables coming from biogenic feedstocks.

These projections are multiplied by EIA-923 electricity consumption data (i.e. total 2009 biomass consumption for each facility) to produce facility-level biogenic feedstock projections from 2009 to 2035. This facility-level data is then mapped to FASOM-GHG regions using the eGRID latitude and longitude coordinates. This methodology provides a justifiable set of unique biogenic feedstock consumption projections calibrated to standard energy market projections.

### 2.1.1. AEO Baseline Scenario Projections

This section provides graphical representation of the current and future alternate estimates of biogenic feedstock consumption derived above at the regional scale. The biogenic feedstock consumption baseline projections are shown in Figure K-5. The results in this figure include both industrial and electricity sector biogenic feedstock consumption as well as the combined total. Of the five scenarios, the Low Renewable Energy Technology Cost case (in which renewable fuels are lower cost), exhibits the largest amount of biogenic feedstock consumption in 2035, with growth accelerating after 2025. It is important to note that in this case biogenic feedstock consumption growth is driven by growth in renewables generally. Because these projections are derived from 2009 numbers, the renewable portfolio is fixed in 2009 and is not allowed to change. The High and Low Economic Growth cases represent potential upper and lower bounds for the AEO Reference baseline scenario because they include exogenous shifts on renewable demands without any endogenous changes in underlying technology.

Shifting focus to the individual sector graphs in Figure K-5, high demand growth is seen for the Low Renewable Technology Cost case in the electricity sector after 2025. This growth occurs because large capacity exists for increases in renewable electricity generation in that sector. This exponential rise is contrasted with the relatively modest rise seen for the industrial sector graph. The industrial sector does not have as much capacity for fuel increase because many of the industrial facilities included in this dataset already use biogenic feedstocks as a primary fuel source, such as pulp and paper mills.

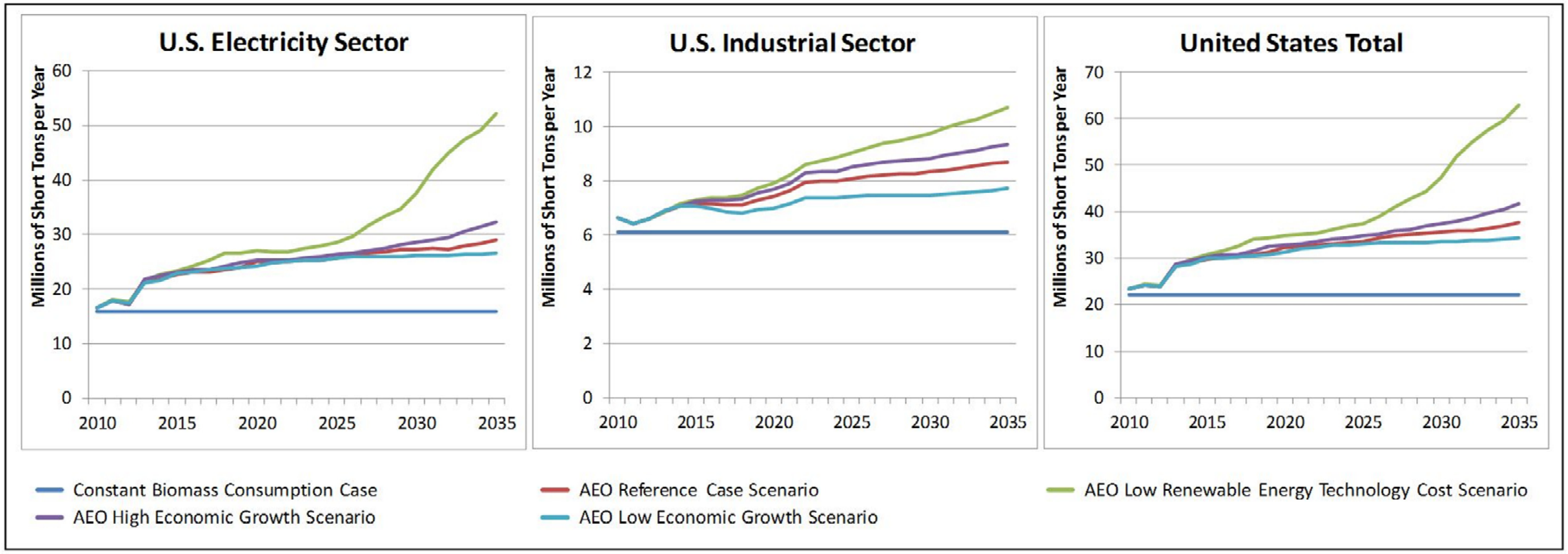


**Figure K-5. Biogenic Feedstock Consumption Projections by Sector.**

Breaking down the scenario projections by FASOM-GHG regions allows insights into the regional distribution of growth in biomass consumption. Regional baseline scenario projections are provided in two groups of graphs at the end of this appendix. Figure K-10 (located in the addendum of this appendix) shows the biogenic feedstock consumption projections for each baseline scenario grouped by FASOM-GHG region. In general, the Low Renewable Technology Cost case represents the maximum growth path for any given FASOM-GHG region. However, the magnitude of the effect of this growth rate varies greatly between regions, with large growth in the Southeast (SE) and Lake States (LS) regions, relative to other baseline scenarios. This variation is further seen in the Rocky Mountains (RM) and Pacific Southwest (PSW) regions, where the high economic growth case yields the greatest amount of biogenic feedstock consumption for a number of years within the projection. This illustrates the regional variability in biogenic feedstock consumption projections presented in the alternative future anticipated baseline scenarios.

Figure K-11 presents the same data showing the biogenic feedstock consumption projections for each FASOM-GHG region grouped by AEO scenario. The top four consumers of biogenic feedstocks for all the cases are the PSW, Northeast (NE), LS, and SE. However, for the Reference case and High Economic Growth case, the PSW has the highest feedstock consumption, whereas the Low Renewable Energy Technology Cost case shows the SE biogenic feedstock consumption expanding rapidly. This Low Renewable Technology Cost case shows there is large capacity potential for renewable energy expansion in the SE, given a reduction in the price of renewables generally, or policies that encourage renewable energy development. With few existing state energy policies requiring or incentivizing renewable energy in the SE, much of the additional renewable capacity potential exists in these states.

# 3. Biogenic Feedstock Consumption Baseline Scenario Projections: Results and Analysis

This section analyzes the GHG emissions and terrestrial carbon sequestration estimates produced from the alternative baseline scenarios created in the previous section. The purpose of this analysis is to illustrate how landscape $CO_2$ balances (emissions flux net of carbon sequestration in biogenic feedstocks and soils) could respond to changes in land management associated with alternative biogenic feedstock demand projections. This evaluation considers total soil and feedstock-related biogenic $CO_2$ emissions from agricultural and forestry land management decisions across multiple future anticipated baselines. A detailed discussion of the emissions fluxes evaluated in this analysis is found in Appendix L. Results illustrate how baseline scenario formation can have a large impact on emissions projections.

FASOM-GHG is used to simulate market equilibrium in the forest and agricultural sectors by maximizing net economic surplus (consumer and producer) over 17 5-year periods (2000–2080), along with a terminal period valuation. Several key assumptions made for this baseline scenario formulation are highlighted below (the basic structure and key underlying datasets of the FASOM-GHG model are described in the Supplemental Information section of the future anticipated baseline case studies appendix, Appendix L):

- For the alternative baseline scenarios, constraints are imposed requiring that a specific volume of biomass be consumed for electricity generation on a regional basis. These regional constraints are all that vary across simulation runs, and serve as the primary basis of comparison for examining GHG implications of changes in biomass energy demand.
- For biofuels production, all alternative baseline scenarios assume that the RFS2 legislation binds and the mandated levels of biofuel are supplied. Feedstock-specific constraints are imposed and are based on the supplemental control case assumptions from the EPA Regulatory Impact Analysis of the RFS2.[5]
- Other than biofuel feedstock restrictions, there are no constraints on feedstock choices for bioenergy across the alternative baseline scenarios, allowing the model to choose an optimal feedstock portfolio to achieve regional biomass requirements.
- Agricultural productivity rates are linear growth rates in agricultural productivity growth and demand growth; these parameters are calibrated to USDA (2009) projections of yield and demand growth for key commodities forestry data (yields, species mix, etc.) are from USFS, calibrated to the forest inventory and assessment (FIA) and other relevant forestry sector datasets.

In addition to FASOM-GHG details included in Appendices J and L, further details regarding the model structure, regional detail, commodity representation, and GHG accounting can be found in Beach et al. (2010).

The discussion of results below focuses on four baseline scenarios. Two of the four AEO baseline scenarios discussed above are used—the Reference baseline and the Low Renewable Technology Cost baseline. The Low and High Economic Growth baseline scenarios were not simulated in FASOM-GHG as these do not deviate greatly from the AEO Reference case scenario. The AEO Reference and Low Renewable Technology Cost baseline scenarios provide a reasonable range of potential biomass energy expansion. In addition to these AEO-based baseline scenarios, two other counterfactual baseline scenarios are also simulated: one with constant 2009 biogenic feedstock consumption (as derived above—this scenario is referred to as the Constant baseline scenario throughout the remainder of this analysis) and another with no biogenic feedstock consumption at stationary sources ("Zero Biomass Consumption" scenario). The difference between the Zero Biomass Consumption baseline scenario and each of the other baseline scenarios (alternate AEO-based and Constant scenario) indicates the additional calculated emissions associated with that level of biogenic feedstock consumption compared to no biogenic feedstock consumption. The purpose of simulating both the Constant and Zero Biomass baseline scenarios is to respond directly to the SAB review of the previous accounting framework, which noted (Swackhamer and Khanna, 2011):

> *Estimating additionality, i.e., the extent to which forest stocks would have been growing or declining over time in the absence of harvest for bioenergy, is essential, as it is the crux of the*

[5] This includes growth in domestic (U.S.) biofuel production up to approximately 30 billion gallons in the 2020 simulation period (including 15 billion gallons of corn ethanol, 13.7 billion gallons of cellulosic ethanol, and 1.3 billion gallons of biodiesel, produced primarily from soybean oil).

> *question at hand. To do so requires an anticipated baseline approach… [the] Framework would need to model a "business as usual" scenario along some time scale and compare that carbon trajectory with a scenario of increased demand for biomass… In general the Framework should provide a means to estimate the effect of stationary source biogenic feedstock demand, on the atmosphere, over time, comparing a scenario with the use of biogenic feedstocks to a counterfactual scenario without the use of biogenic feedstocks.*

With this in mind, baseline scenarios are constructed so that business-as-usual (BAU) projections, which include anticipated growth in biogenic feedstock consumption, can be compared relative to alternative baseline scenarios that include no new growth in biogenic feedstock demand (the Constant scenario), and no future consumption of biomass. Thus, this appendix considers a range of possible anticipated future baselines and alternative counterfactuals, allowing for a detailed assessment of potential biogenic emissions estimates for illustrative purposes.

The following sections continue with more detail describing the baseline scenario results, with a brief look at periodic net emissions and cumulative net emissions for each of the scenarios. Finally, the model results for additional emissions in each AEO baseline scenario are presented and discussed at both the national and regional levels.

## 3.1. FASOM-GHG Simulation Results

This section presents and compares results for the various baseline scenarios. First net $CO_2$ emissions are presented followed by cumulative net emissions, and finally additional emissions relative to a counterfactual scenario in which no biogenic feedstocks are consumed at stationary sources.

### 3.1.1. Net Emissions Flux per Time Period

For each 5-year period, an annual emission or sequestration value was calculated using the FASOM-GHG equations and parameters (presented in the supplemental section at the end of Appendix L), then a total net emissions flux for that 5-year period was calculated by aggregating the individual fluxes. Figure K-6 illustrates projected $CO_2$ emissions flux trajectories across the different baseline scenarios using atmospheric GHG accounting (a positive value represents net emissions, while a negative value represents net carbon sequestration on the landscape). Note that the difference between scenarios is not as significant as the change in net emissions between periods (i.e., over time). The cyclical shape of these trajectories is driven by periodic shifts in forest management; harvest emissions and forest biomass growth can vary period-to-period, leading to high net emissions totals in some periods and net sequestration in others. Land use change emissions can also contribute to this cycle. Assumptions related to management practices for the various feedstocks considered are described in Appendix H. Periods with high agricultural land use conversion (such as pasture or forest conversion to cropland) can result in increased emissions, while afforestation can increase terrestrial carbon uptake.

Differences between baseline scenario projections are subtle, as the overall shape of these trajectories is similar. However, the absolute difference in annual emissions could be significant (for instance, this difference ranges 3–5 million $tCO_2e$ per year for the 2010 simulation period).

Furthermore that each biogenic feedstock consumption scenario results in an immediate increase in emissions in the 2010 time step, coinciding with the first year of the biogenic feedstock demand shock applied to each alternative baseline scenario. Thus, biomass demand initially increases emissions relative to the Zero Biomass demand scenario, driven by changes in management in response to the new feedstock demand.

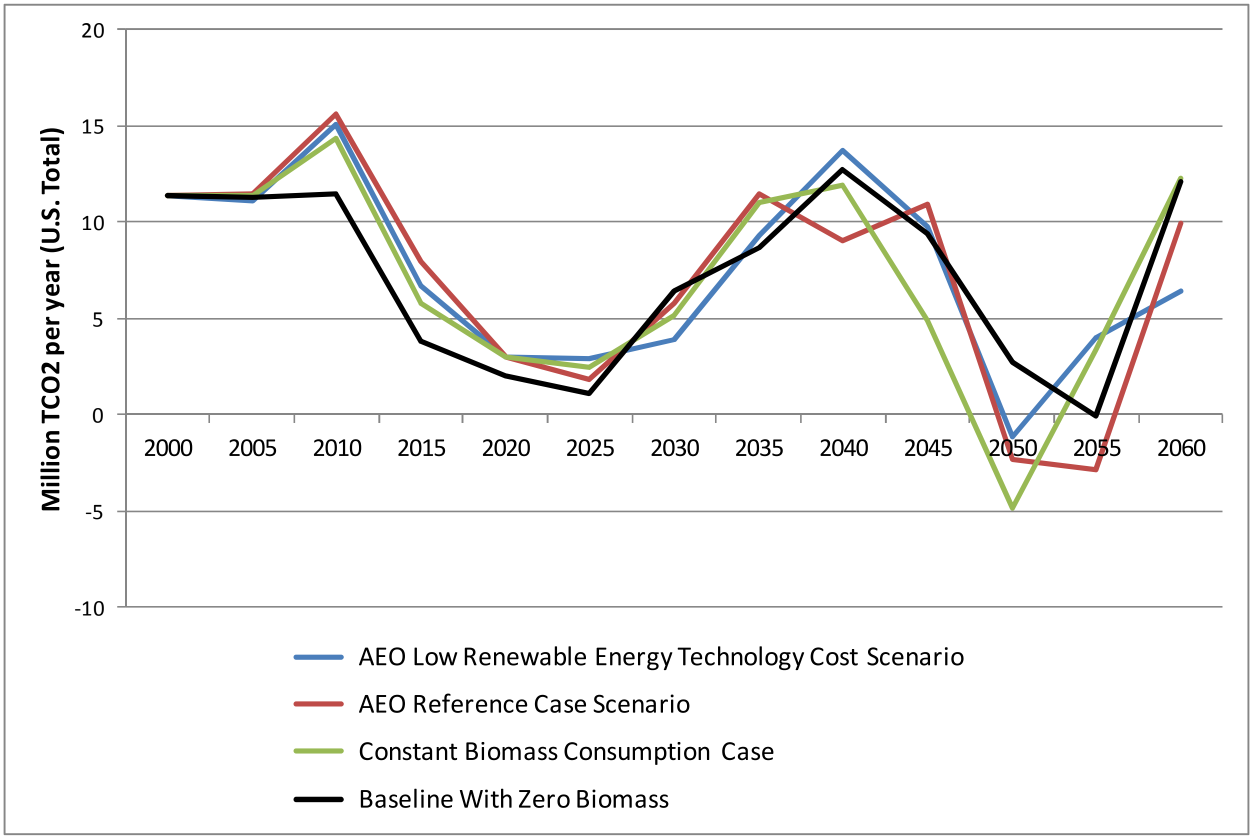


**Figure K-6. Terrestrial Carbon Flux from U.S. Forest and Agricultural Sectors (Excluding Non-$CO_2$ Emissions, Emissions from Fossil Fuel Consumption, and Carbon Stored In Wood Products).**

### 3.1.2. Cumulative Net Emissions

Although the periodic net emissions flux trajectories are similar over time, minor differences in annual fluxes can result in large differences in cumulative emissions over time. Total emissions for each 5-year period are converted to a cumulative emissions total over the time horizon, displayed in Figure K-7.

In 2035 (when biogenic feedstock demand peaks), cumulative emissions for the AEO Reference baseline scenario are 22% higher than cumulative emissions for the Zero Biomass baseline. However, over time, projected cumulative emissions for the Zero Biomass baseline begin to converge with the alternative biogenic feedstock demand baselines. There is little difference between the Zero and Constant Biogenic Feedstock baseline scenarios toward the end of simulation horizon, and less than 7% difference across all scenarios. Thus, after immediate and medium-term emissions effects of increased biogenic feedstock demand, physical carbon stocks begin to recover and the cumulative difference in emissions from Zero Biomass to the alternative baselines begins to subside. One implication of these results is that the choice of time scale is important and can have a large impact on the cumulative emissions difference between scenarios (i.e., with a shorter timeframe used for this illustration, the observed convergence would not occur).

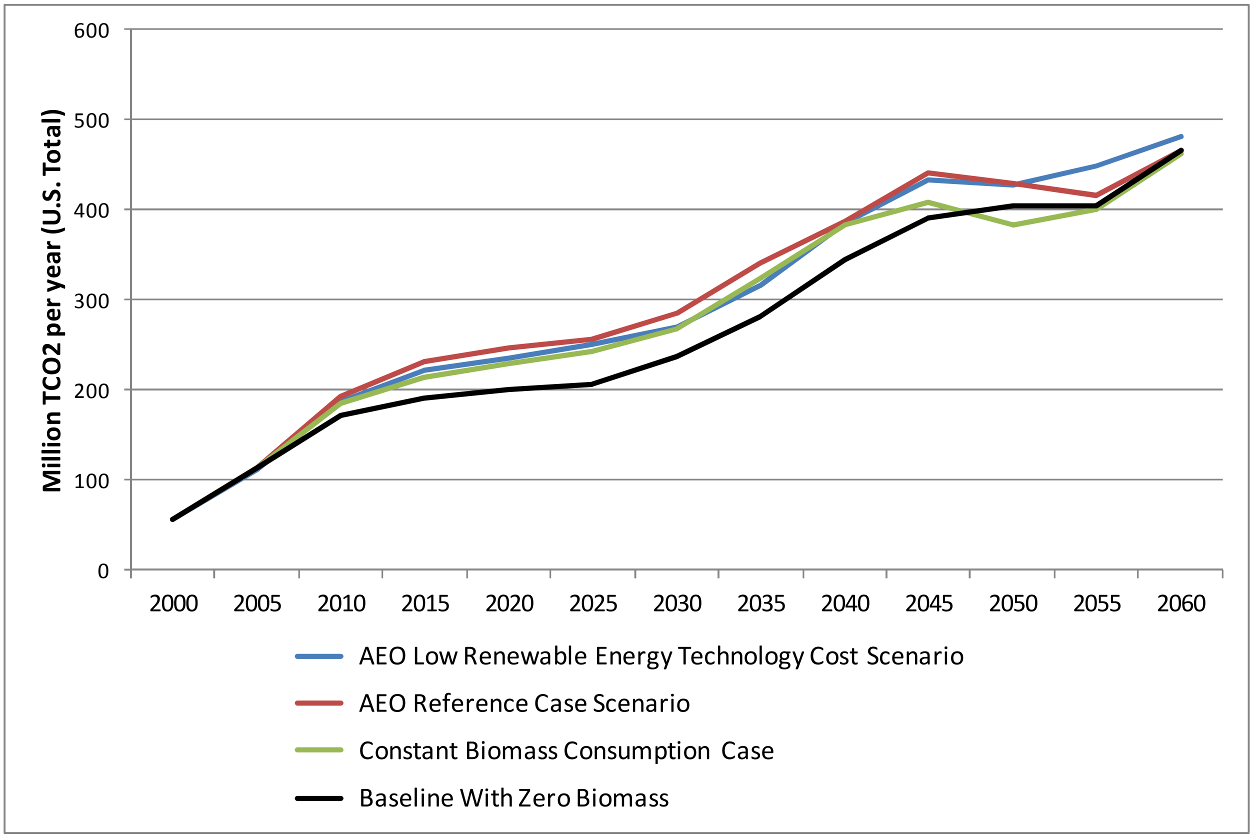


**Figure K-7. Cumulative Emissions Over Time by Biomass Scenario.**

### 3.1.3. Additional Estimated Emissions

Additional biogenic $CO_2$ consumption and landscape-level emissions are calculated relative to the Zero Biomass baseline scenario. This is done in order to capture all additional emissions effects of biomass consumption under an anticipated future baseline projection relative to an alternative future in which there is no biomass consumption. Thus, all additional current and expected growth in biogenic feedstock consumption and the relative change in terrestrial $CO_2$ emissions are captured. This approach allows one to consider changes in emissions for relative changes in future biomass demand (captured by the Constant [no growth in consumption], AEO Reference, and AEO Low Renewable Technology Cost Baselines).

Emissions intensities for additional biogenic feedstock consumption relative to the Zero Biomass baseline scenario are computed (as this includes all biogenic feedstock users). An emissions intensity is the ratio of net biogenic emissions (net of emissions and landscape-level sequestration) to net biogenic $CO_2$ in biomass consumed for energy ($tCO_2$ equivalent emissions/ $tCO_2$ equivalent in consumed biogenic feedstocks). In this context, emissions intensity represents the ratio of emissions relative to a unit of biogenic feedstock used for energy (i.e., the portion of biogenic feedstock carbon emitted to the atmosphere). A value of 0 implies no net emissions, meaning that biogenic $CO_2$ emissions from the use of biogenic feedstocks in energy production would be balanced with carbon uptake in the feedstock and on the landscape where the feedstock was produced. A value of 0.5 would imply that half of the biogenic $CO_2$ emissions are displaced by carbon uptake on the landscape. This metric provides insight into potential emissions intensity of biogenic feedstock consumption by current and future stationary sources under these specific anticipated baseline scenarios and related parameters. This metric does not take into account stationary source process emissions, including combustion efficiencies, feedstock losses during processing, or other possible components related to feedstock procurement or processing.

For the three alternative biogenic feedstock consumption baseline scenarios (Constant, AEO Reference baseline, and AEO Low Renewable Technology Cost Baseline), cumulative additional emissions are calculated as the difference from the Zero Biomass baseline scenario and each alternative scenario for each period of the simulation horizon. Cumulative additional biogenic $CO_2$ from biogenic feedstock consumption is calculated by converting annual biogenic feedstock consumption requirements for each scenario to a cumulative value, and then converting to $CO_2$ equivalence (assuming that each dry ton of feedstock is 50% carbon). This cumulative additional emissions value is then divided by total $CO_2$ equivalence of biogenic feedstock consumption to derive the emissions intensity per unit of biogenic feedstock (Figure K-8).

In general, emissions intensity projections show that biogenic $CO_2$ emissions are not entirely displaced by terrestrial carbon sequestration early in the simulation horizon. When compared with the Zero Biomass baseline, additional emissions per-unit of additional biogenic $CO_2$ consumption ranges 0.35–0.47 ton $CO_2e$ once the biogenic feedstock requirements are imposed in 2010. Note, however, that emissions intensity declines steadily over time for each baseline scenario, approaching a net carbon balance of 0 for the AEO Reference and AEO Low Renewable Technology Cost cases. Emissions intensity reaches values below 0 for the Constant Biomass case, indicating that cumulative biogenic $CO_2$ emissions are more than balanced by emissions changes on the landscape. This decline in emissions intensity is driven by several factors:

1) A shift in land use/management early in the simulation horizon that increases tree carbon uptake over the long term (afforestation of cropland and pastureland);
2) Declining market effects of initial biomass demand shocks;
3) Improved agricultural productivity over time due to exogenous yield growth assumptions and endogenous yield growth responses to the biomass requirements (including regional crop mix changes); and
4) A shift in biogenic feedstock composition (as seen in Figure K-9).

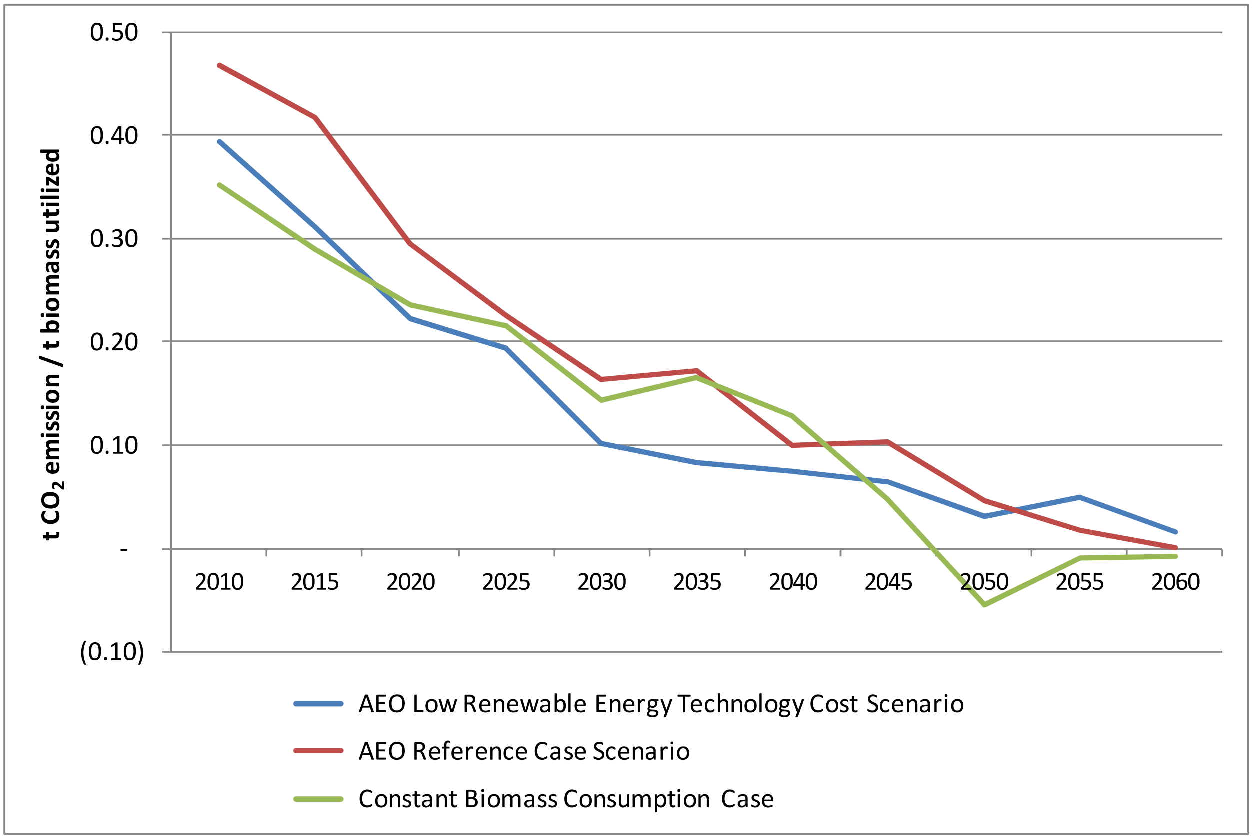


**Figure K-8. Emissions Intensity of Biomass Energy Relative to the Zero Biomass Scenario (Cumulative Additional Emissions Divided by Cumulative Biogenic Carbon from Additional Biogenic Feedstocks).**

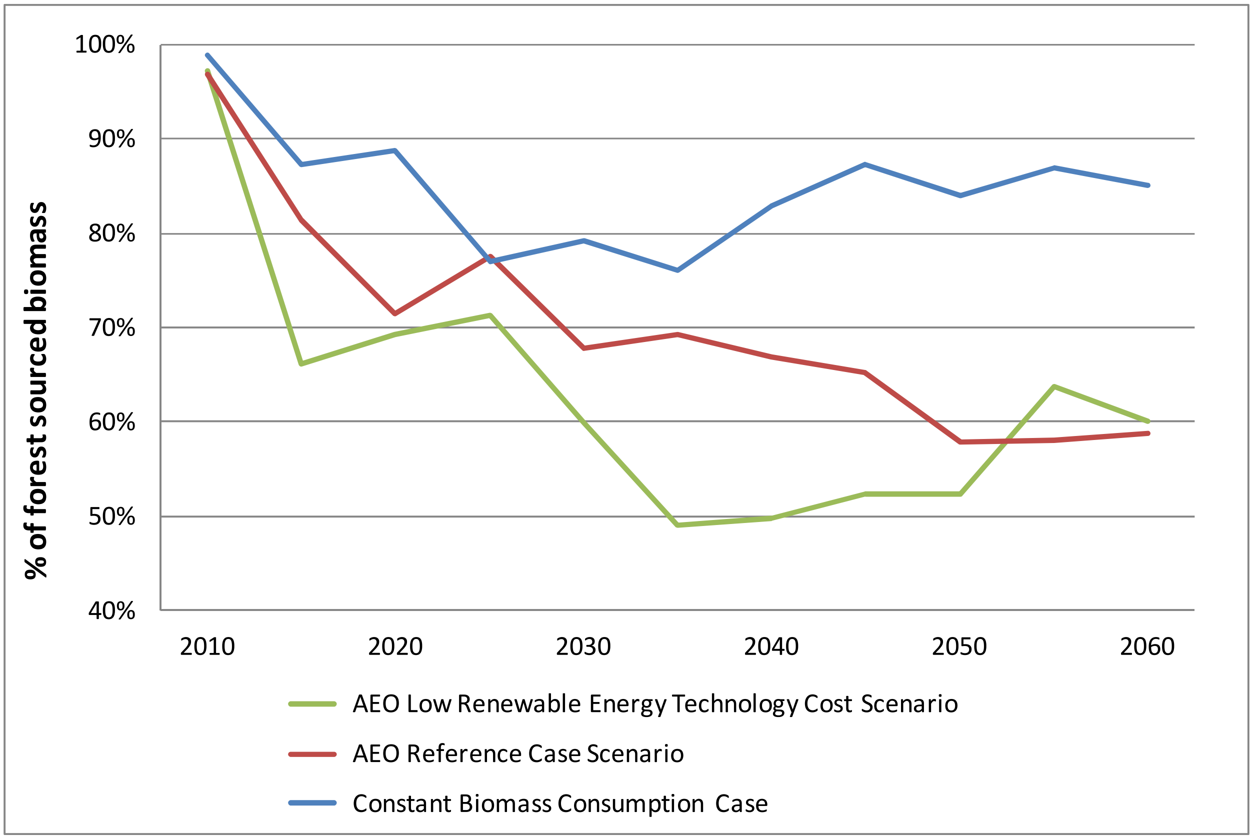


**Figure K-9. Proportion of Biogenic Feedstock from Woody Sources by Scenario Relative to the Zero Biomass Scenario.**

In general, use of forest biogenic feedstocks for energy generation generates greater direct emissions and less energy output per-unit area than agricultural feedstocks or dedicated energy feedstocks, such as switchgrass (for more discussion, see Latta et al., 2013). Initially, the overall feedstock portfolio is required to be approximately 96% derived from forestry feedstocks to match observed biomass consumption patterns, hence the higher emissions per unit of biomass. However, this constraint is relaxed after the initial time period, giving the model flexibility to choose a biogenic feedstock portfolio mix that minimizes the costs of meeting the total biogenic feedstock requirement. Over time, as greater amounts of biogenic feedstocks are required for the AEO Reference and Low Renewable Technology Cost Baseline scenarios, the portfolio shifts to a higher proportion of agricultural feedstocks, which decreases overall emissions intensity.

This decline is more pronounced for the Low Renewable Technology Cost baseline as it adopts a much higher proportion of dedicated energy feedstocks to meet total biogenic feedstock demand. Although the Constant Biomass scenario continues to consume a relatively high proportion of forest-derived feedstocks, total biogenic feedstock requirements do not increase from the base year, so land management, markets, and terrestrial carbon stocks adjust more rapidly, causing emissions to fall.

# 4. Conclusion

This appendix developed biogenic feedstock demand projections with an initial (2009) value calibrated to observed consumption patterns at electricity and industrial sector stationary sources. Future projections were then calibrated to projected growth rates in renewable energy demand (by sector and region) from the AEO (2012). A range of anticipated future baseline scenarios were created, representing a range of possible biogenic feedstock consumption futures. These projections were mapped to FASOM-GHG agroforestry regions, representing regional biogenic feedstock requirements for simulation analysis.

Emissions trajectories across the alternative future baseline scenarios were compared to a scenario in which no agricultural and forestry biomass is consumed at stationary sources for energy generation. Results of the simulation analysis revealed that emissions from biogenic feedstock consumption are not fully balanced by initial landscape $CO_2$ uptake. However, over time, emissions intensity decreases, approaching or surpassing a net carbon balance for all alternative anticipated future baseline scenarios assessed here.

In general, these results are consistent with previous studies that have shown that there are GHG consequences associated with biogenic feedstock production, especially immediately following an increase in biogenic feedstock demand (Latta, et al., 2013; Daigneault et al., 2012). However, this analysis shows that carbon dioxide emissions associated with biogenic feedstock production and use are at least partially balanced by changes in sequestration on the landscape and that, over time (in this case decades), an increasing share of these carbon dioxide emissions is balanced at a national level and in most regions as well. When disaggregated regionally, emissions intensity trajectories also approach net carbon balances for most FASOM-GHG regions, with a few clear exceptions (CB, PSW). For the CB and PSW, land use change early in the simulation horizon (afforestation) leads to net sequestration, which causes negative emissions intensity in the near term.

Three of the future anticipated baseline scenarios presented in this appendix are utilized within the case study appendix (Appendix L): Zero Biomass, Constant Biomass (existing sources in 2009), and AEO Reference. Appendix L develops feedstock- and region-specific demand shocks in addition to the AEO Reference case scenario. Emissions projections from these feedstock case study scenarios are then compared to the three alternative baselines above to evaluate the emissions effect of a marginal increase in consumption of one feedstock (comparison to AEO Reference), an average effect relative to current consumption levels (Constant), and an average effect relative to no biogenic feedstock consumption (Zero).

The baselines and estimated values derived in this appendix and in Appendix L are intended to illustrate the functionality of a future anticipated baseline approach method and do not reflect EPA findings in the context of specific policies or programs. As with all modeling studies, there are a number of uncertainties present in the baseline scenario assumptions and parameters adopted for this analysis. These uncertainties include future environmental conditions and the biophysical emissions accounting parameters, future economic or policy conditions, and technological growth (both for agricultural/forestry feedstock yield and commodity-processing technologies). However, model projections provide key insight into the potential market and land use consequences of possible shifts in the demand for biogenic feedstocks at stationary sources. Furthermore, this study does not include full coverage of possible feedstocks from agricultural and forestry production processes. Most notably, FASOM-GHG does not include production of black liquor as an industrial processing by-product of pulp and paper production.

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# 6. Supplemental Data and Information

## 6.1. Graphics

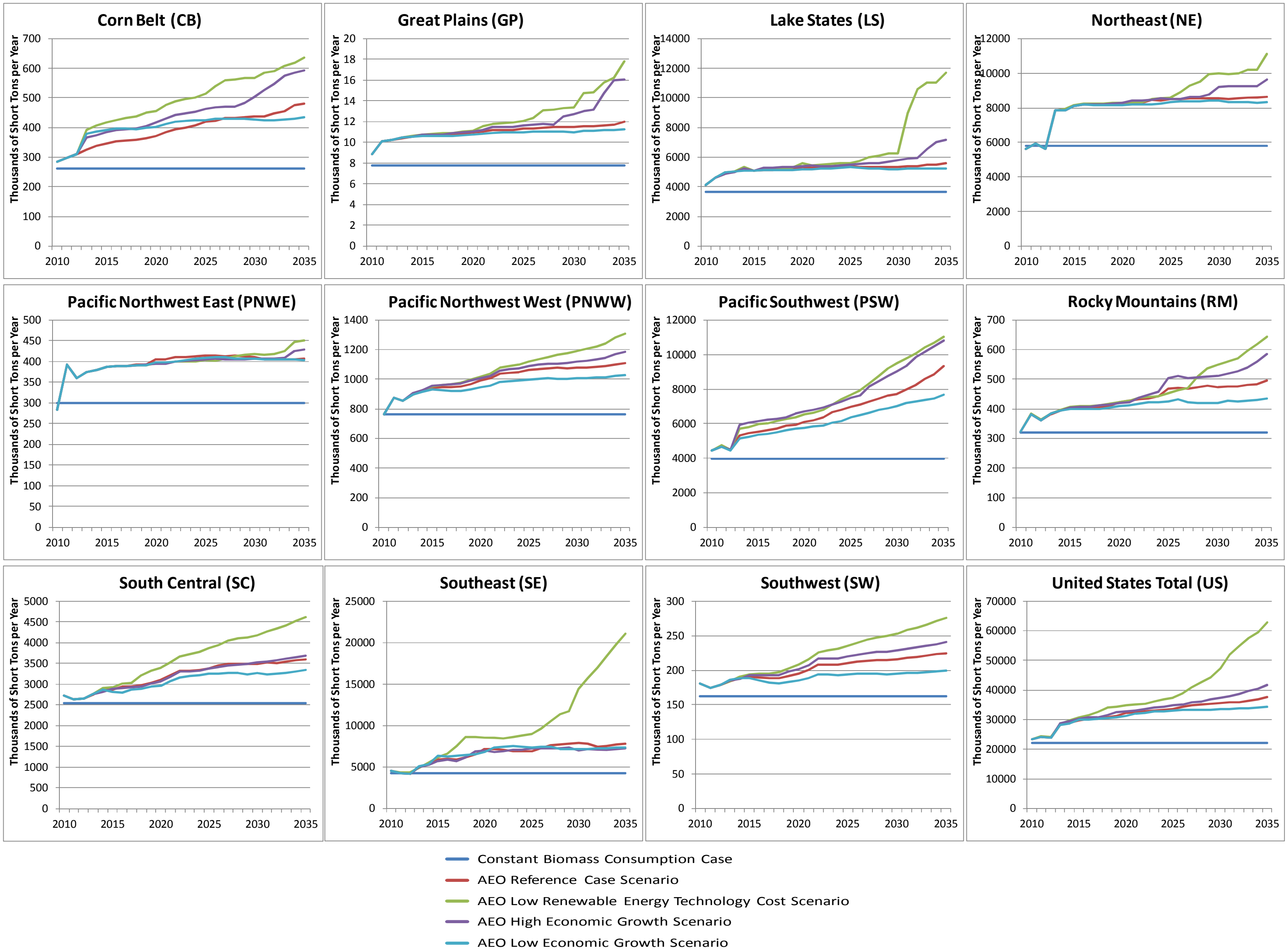


**Figure K-10. Biogenic Feedstock Consumption Projections by AEO Scenario Grouped by FASOM-GHG Region.**

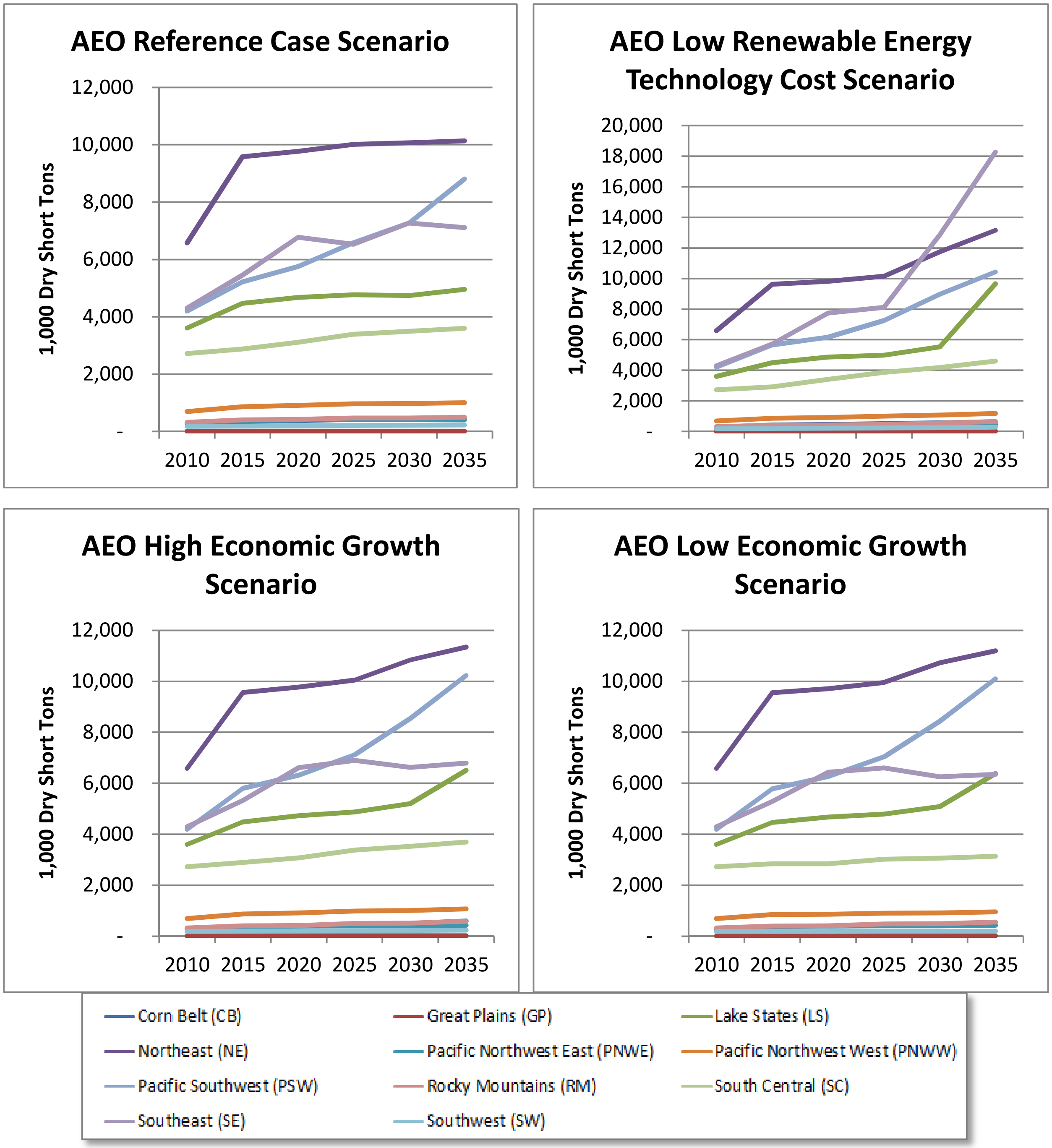


**Figure K-11. Biogenic Feedstock Consumption Projections by FASOM-GHG Region Grouped by AEO Scenario.**

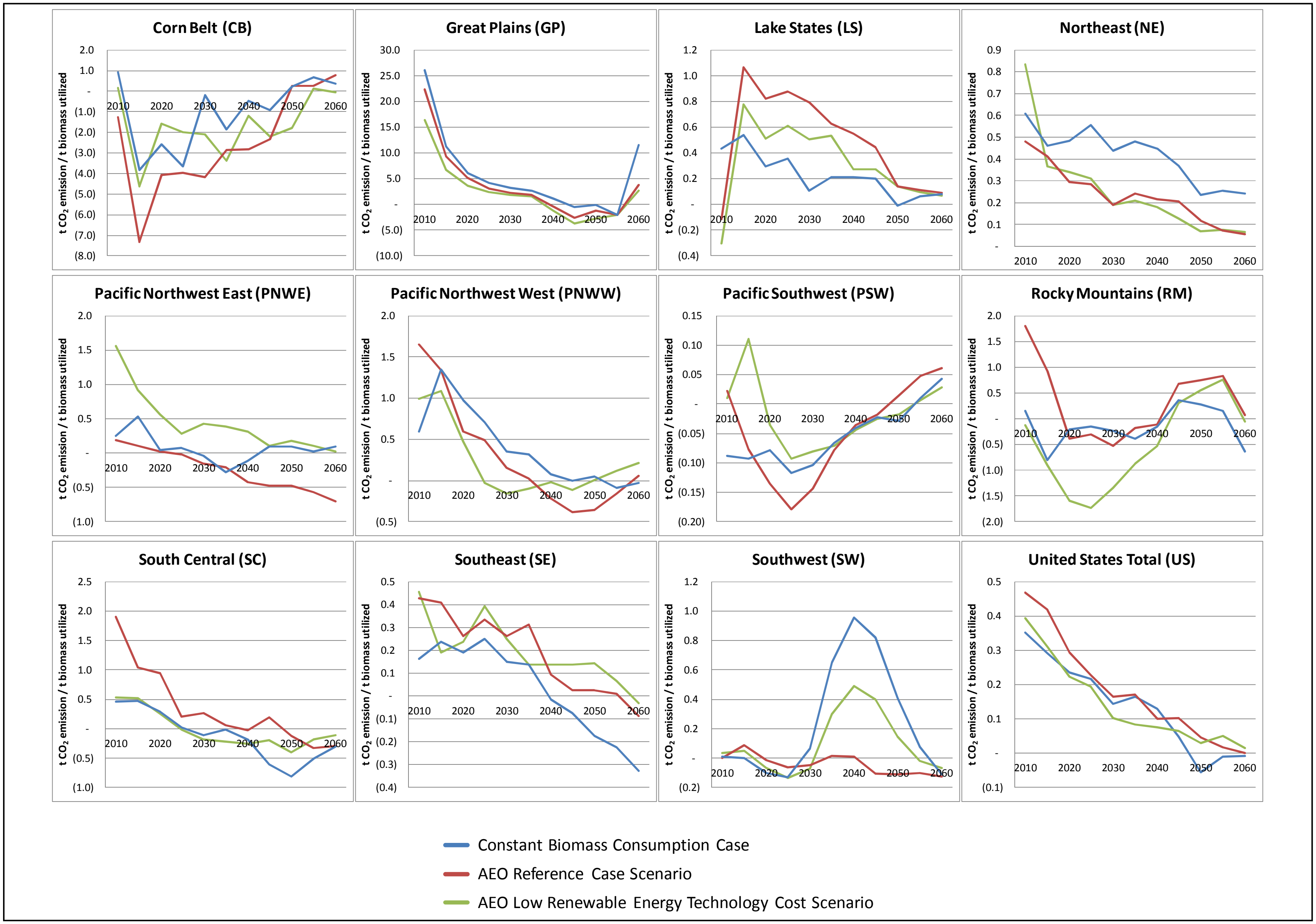


**Figure K-12. Additional Emission per Ton of Biogenic Feedstock Utilized.**

# Appendix L: Illustrative Forestry and Agriculture Case Studies Using a Future Anticipated Baseline

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# 1. Introduction

This appendix demonstrates the functionality of the future anticipated baseline approach through three illustrative region- and feedstock-specific case studies. These case studies use the baseline scenarios constructed in Appendix K as the basis for comparison of alternative biogenic feedstock production scenarios per specific feedstocks and specific regions. The application of the future anticipated baseline approach within these case study constructs allows for the calculation of illustrative values for the pertinent framework equation terms and ultimately generation of illustrative biogenic assessment factors (*BAF*) specific to the individual case study parameters and assumptions.

Results show aggregate emissions estimates on a specific regional scale per case study. Results can be interpreted as the projected emissions intensity of specific biogenic feedstocks consumed at existing or anticipated stationary sources across multiple baseline projections of biogenic feedstock consumption. However, results do not reflect the net emissions *contribution* of a particular feedstock within a particular region but rather illustrate potential net biogenic emissions effects *associated with increased consumption* of a specific feedstock in a specific region under specific conditions. To maintain consistency with the reference point approach, region- and feedstock-specific simulation scenarios were developed to isolate the landscape-level carbon-based emissions fluxes related to a demand shift for an individual feedstock relative to the AEO Reference baseline (as presented in Appendix K).

The three case studies are:

- Roundwood in the Southeast (SE);
- Corn stover in the Corn Belt (CB); and
- Logging residues in the Pacific Northwest (PNW).

This appendix first provides an overview of the methods used to construct the case study parameters, an explanation of the how landscape-level biogenic emissions fluxes are mapped to *BAF* equation terms, a discussion of how to interpret results (using different assessment methods), and a presentation of illustrative case study results and analysis.

The values derived in this appendix are intended to illustrate the future anticipated baseline approach and do not reflect EPA findings in the context of specific policies or programs. As with all modeling studies, a number of uncertainties are present in the baseline assumptions and parameters adopted for this analysis. These uncertainties include historical input data, future environmental conditions and the biophysical emissions accounting parameters, future economic or policy conditions, and technological growth (both for agricultural/forestry feedstock yield and commodity processing technologies). However, model projections provide key insight into the potential market and land use consequences of possible shifts in the demand for biogenic feedstocks at stationary sources.

## 2. Method and Parameters Used to Calculate the Illustrative *BAF*s Using the Future Anticipated Baseline Approach

The intertemporal optimization approach used in these illustrative case studies captures investment behavior under anticipated changes in feedstock demand; thus, land management responds in advance of an anticipated change. This approach allows for a depiction of land use investment/management over the long term, which provides an improved projection of landscape-level biogenic $CO_2$ emissions under anticipated changes in biogenic feedstock consumption than static (one-time) models or recursive dynamic models that do not react to future expectations.

Ultimately, numerous assumptions and parameters can be varied to establish future anticipated baselines that differ from those presented here and in Appendix K. Furthermore, there are numerous possibilities for creating alternative feedstock scenarios relative to a future anticipated baseline. The primary goal of this appendix is to illustrate how the future anticipated baseline approach could be applied in practice to assess landscape-level emissions effects related to changes in demand for individual feedstocks. The secondary objectives of this appendix are to evaluate the potential direction and magnitude of biogenic $CO_2$ emissions from increased regional feedstock consumption, using the modeling assumptions and baseline constructs as presented in Appendix K.

## 2.1. Case Study Methods

Each feedstock case study was developed using the same underlying methodology. Each feedstock scenario is evaluated relative to the three alternative baseline scenarios, as introduced in Appendix K (Zero Biomass, Constant Biomass [existing sources in 2009], and AEO Reference). For each feedstock case study, the regional emissions intensity of additional biogenic feedstock consumption (additional biogenic $CO_2$ emissions divided by additional biogenic feedstock $CO_2$) is computed, similar to the approach outlined in Appendix K and described in more detail below.

Each of the case studies begins with regional biomass consumption trajectories from the AEO Reference baseline and then requires an additional 1 million short dry tons of specific biogenic feedstock consumption in the region under consideration. This additional biomass requirement is phased in linearly, beginning with 250,000 short dry tons in the 2015 simulation period, reaching 1 million tons in 2030. The feedstock requirement is phased in over time under the conservative assumption that it could take time for a new facility or demand point to build up a steady supply source of one particular feedstock given regional market dynamics. The additional biomass requirement is then held constant for the remainder of the simulation horizon[1] and must be met by the case study feedstock only. For the SE roundwood case, for example, the additional biomass requirement must come exclusively from hard and soft roundwood. This constraint is maintained throughout the simulation horizon to isolate the emissions effects of increased demand for a specific feedstock over the long term.

Comparison of the 1 million ton increased feedstock consumption scenario to the AEO Reference baseline scenario can be interpreted as the *marginal* effect of a new source of consumption that is fueled by a single feedstock, relative to the AEO Reference anticipated baseline. This increased consumption could be thought of as the estimated marginal effect of the additional demand from a stationary source that is expected to consume approximately 1 million tons of biogenic $CO_2$ annually for onsite energy generation over the long term.

Comparison of the 1 million ton increased feedstock scenario to the Zero Biomass baseline scenario provides an estimate of the *average* biogenic $CO_2$ emissions effect for all existing and planned biogenic feedstock consumption within a region (as defined by the eGRID/EIA dataset-derived 2009 existing users and AEO Reference baseline anticipated new users) plus the additional

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[1] The 2012 Annual Energy Outlook projections do not extend past 2030; thus, biomass consumption shock is held constant after this simulation period.

feedstock-specific requirement from the case study. For this comparison, the feedstock scenarios were constructed in exactly the same way (same volumes, same feedstocks), with the anticipated baseline feedstock portfolio exactly matching the AEO Reference baseline simulation results over time. The difference here is that biogenic $CO_2$ emissions intensity metrics are computed relative to the Zero Biomass simulation results. Thus, the numerator represents the net change in regional projected emissions (Feedstock Case Study − Zero Biomass), while the denominator represents regional biogenic $CO_2$ consumption (AEO Reference baseline biomass plus additional feedstock requirement).

Another possible method is the *augmented average* approach.[2] Comparison of the 1 million ton increased feedstock scenario to the Constant Biomass baseline can be interpreted as the *augmented average* biogenic $CO_2$ emissions effect of planned expansion in biogenic feedstock consumption (as defined by the AEO Reference baseline) above the eGRID/EIA dataset-derived 2009 existing users, plus the additional feedstock specific requirement from the case study. Essentially, this is the same methodology as the comparison to the Zero Biomass baseline, but all calculations are relative to an anticipated baseline that holds biomass consumption fixed to observed levels in 2009.

Illustrative equation term and estimated *BAF* results using the marginal and average methods above are presented for each case study.

## 2.2. Case Study Parameters

All feedstock case studies are simulated over an 80-year time horizon (2000–2080) to capture investment dynamics in the forestry sector over this period. The results are computed using key outputs for the 2010–2060 time horizon (in 5-year timesteps), which provides a 50-year time frame for evaluating land use and biogenic emissions changes (and avoids any terminal effects that may affect results in the last few time periods of results). Results generated within this overall time frame can be aggregated and evaluated in different ways (e.g., the *BAF* can be constructed for 10- to 50-year time frames as desired), and the 50-year mark should not be interpreted as an EPA decision on applying time frames in the contexts of specific programs and policies. The spatial scale of these regional case studies is represented by the 11 primary agroforestry regions of FASOM-GHG. Additional FASOM-GHG modeling details are provided in the Supplemental Information section at the end of this document.

Although FASOM-GHG offers full GHG accounting options (including $N_2O$ and $CH_4$ emissions from crop and livestock operations), this study focuses on changes in landscape-level biogenic $CO_2$ only (though a sensitivity evaluating the impact of including $N_2O$ is included in this analysis). This approach includes carbon in agricultural and forestry soils, and carbon stored in forest and agricultural biomass (additional details provided below).

[2] Discussed here as a possible method, but this method was not employed to avoid further complexity, as many different methods could be discussed and employed using a future anticipated baseline approach. Therefore, the illustrative results tables do not include this category.

# 3. Mapping and Interpreting Future Anticipated Baseline Data and Illustrative Results

This section presents which FASOM-GHG data components are mapped to *BAF* equation terms as well as how *BAF* equation terms are calculated using these data over a specified simulation time horizon. Specifically, FASOM-GHG projections are used to derive representative values for regional net growth (*GROW*), total net carbon change on the feedstock production region (*SITETNC*), and avoided emissions from feedstock harvest or collection (*AVOIDEMIT*) in each simulation period. These terms are aggregated into a net biogenic emissions (*NBE*) term, which is used along with the total additional biogenic $CO_2$ (calculated directly from the feedstock-specific biomass constraint) as a representative potential gross emissions (*PGE*) value to derive an estimated *BAF*.

A major difference in the illustrative *BAF* terms generated with the retrospective reference point and the future anticipated baseline approach is that the equation terms (*PGE, GROW, SITETNC, AVOIDEMIT*, and *NBE*) as defined and applied within the future anticipated baseline approach do not represent the absolute emissions associated with the terms but rather the additional, or relative, emissions compared with an alternate potential future.

## 3.1. FASOM-GHG Data Component Mapping to *BAF* Terms

Deriving *BAF* equation term values from FASOM-GHG output data components involves aggregating the various emissions components into a single value. Table L-1 lists specific carbon-based GHG flux categories from FASOM-GHG simulations and the *BAF* equation term associated with each carbon-based GHG flux account. Note that non-$CO_2$ emissions from crop and livestock management, carbon stored in wood products, and fossil fuel emissions from land management are not included in this analysis.

**Table L-1. FASOM-GHG Emissions Components Matched with *BAF* Equation Terms.**

| FASOM-GHG Emissions Component | Southeast Roundwood | Pacific Northwest Logging Residues | Corn Belt Corn Stover |
|---|---|---|---|
| **Agricultural LUC and Soil Management Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Logging Residue Decay Flux** | *AVOIDEMIT* | *AVOIDEMIT* | *AVOIDEMIT* |
| **Afforestation Harvest Flux** | *GROW* | *GROW* | *SITETNC* |
| **Afforestation Tree Carbon Flux** | *GROW* | *GROW* | *SITETNC* |
| **Existing Forest Harvest Flux** | *GROW* | *GROW* | *SITETNC* |
| **Existing Forest Tree Carbon Flux** | *GROW* | *GROW* | *SITETNC* |
| **Afforestation Litter and Understory Harvest Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Afforestation Soil Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Afforestation Litter and Understory Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Deforestation Soil Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Existing Forest Litter and Understory Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Existing Forest Litter and Understory Harvest Flux** | *SITETNC* | *SITETNC* | *SITETNC* |

| FASOM-GHG Emissions Component | Southeast Roundwood | Pacific Northwest Logging Residues | Corn Belt Corn Stover |
|---|---|---|---|
| **Logging Residue Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |
| **Existing Forest Soil Carbon Flux** | *SITETNC* | *SITETNC* | *SITETNC* |

Details about the underlying input data for the above FASOM-GHG elements and how they are calculated in the model are included in the Supplemental Information section in this appendix.

### 3.2. Potential Gross Emissions (*PGE*)

Because the *BAF* as calculated using the future anticipated baseline approach is a measure of emissions intensity, *PGE*, which also varies by time ($t$), is the estimated biogenic feedstock consumed (in terms of $CO_2$) in the case study ("*CS*") that is additional to the estimated biogenic feedstock consumed (in terms of $CO_2$) in the alternate baseline ("*AB*"):

$$PGE_t = \left(BIOGENIC\_CO2_{"CS",t} - BIOGENIC\_CO2_{"AB",t}\right) \qquad \textbf{(EQ. L.1)}$$

### 3.3. Net Growth (*GROW*)

Similar to the retrospective reference point baseline approach, the future anticipated baseline treats *GROW* as the net landscape biogenic $CO_2$ growth. The forest $GROWTH_{i,"CS",t}$ is indexed by $i$ to represent biogenic emissions fluxes contributing to *GROW* from Table L-1, representative scenario "*CS*" for the case study (or alternative baseline, "AB"), and $t$ for the time period. The following equation represents the calculation of $GROW_t$, where the $i$ index specifically represents the individual biogenic $CO_2$ fluxes from the SE roundwood column of Table L-1 labeled "*GROW*" (afforestation harvest flux, afforestation tree carbon flux, etc.) It should be noted that the fluxes considered in set $i$ include both forest growth and removals, thus yielding a net growth value for each time period, $t$. Under this approach, each flux account is the difference between the simulated values for the feedstock case study scenario ("*CS*") and the alternate baseline ("*AB*") for all time periods.

$$GROW_t = \sum_i \left(GROWTH_{i,"CS",t} - GROWTH_{i,"AB",t}\right) \qquad \textbf{(EQ. L.2)}$$

### 3.4. Total Net Change in Site Emissions (*SITETNC*)

*SITETNC* represents the difference in landscape-level biogenic $CO_2$ emissions fluxes not directly related to the actual biogenic feedstock growth (for each time period in the simulation). This factor includes changes in carbon stored in soils, non-harvested biomass, and potentially other pools. The change in site carbon, $SITE_{k,"CS",t}$, is the sum of a set of k biogenic $CO_2$ components from the SE roundwood column of Table L-1 labeled "*SITETNC*" for the case study scenario ("*CS*") in time period $t$. The following equation illustrates how periodic *SITETNC* values are computed under the future anticipated baseline framework as the relative difference in emissions between the case study, "*CS*," and an alternative baseline, "*AB*," for each time period, $t$.

$$\boldsymbol{SITETNC_t = \sum_k\left(SITE_{k,''CS'',t} - SITE_{k,''AB'',t}\right)} \quad \textbf{(EQ. L.3)}$$

### 3.5. Avoided Emissions (*AVOIDEMIT*)

A similar logic follows for *AVOIDEMIT. AVOIDEMIT* represents the avoidance of estimated biogenic emissions that could have occurred on the feedstock landscape without biogenic feedstock removal. In the context of the future anticipated baseline approach, *AVOIDEMIT* represents the relative difference in avoided biogenic emissions between scenarios. Each *"AVOID"* term in equation 4 below represents the avoided biogenic emissions within a particular scenario. Letting $AVOID_{h,''CS'',t}$ represent the sum of the set of *h* biogenic $CO_2$ components from the Southeast roundwood column of Table L-1 labeled *AVOIDEMIT*, for the case study scenario("*CS*") in time period *t.* The following equation illustrates how periodic *AVOIDEMIT* values are computed under the future anticipated baseline framework as the relative difference in emissions between the case study ("*CS*") and an alternative baseline ("*AB*") for each time period, *t*.

$$\boldsymbol{AVOIDEMIT_t = \sum_h\left(AVOID_{h,''CS'',t} - AVOID_{h,''AB'',t}\right)} \quad \textbf{(EQ. L.4)}$$

### 3.6. Net Biogenic Emissions (*NBE*)

*NBE* represents the difference in biogenic landscape-level $CO_2$ emissions (emissions from harvesting and using the biogenic feedstock) between scenarios (calculated as the sum of all landscape-level $CO_2$ emissions). This is represented as:

$$\boldsymbol{NBE_t = GROW_t + SITETNC_t + AVOIDEMIT_t} \quad \textbf{(EQ. L.5)}$$

### 3.7. Biogenic Assessment *Factor (BAF)*

Thus, the biogenic assessment factor is ratio of the net biogenic emissions ($NBE_t$) to the potential growth emissions ($PGE_t$), or simply put:

$$\boldsymbol{BAF_t = {}^{NBE_t}/_{PGE_t}} \quad \textbf{(EQ. L.6)}$$

## 4. Guide to the Case Studies

### 4.1. Understanding the Illustrative Results

The illustrative results provided below for the three case studies include positive and negative values. Positive values indicate a net flux of emissions (harvest or land use change emissions outweigh biogenic $CO_2$ sequestration on the landscape), whereas negative values indicate net sequestration (biogenic $CO_2$ sequestration on the landscape outweighs harvest or land use change emissions). However, determination of how and whether negative values would be applied in practice would depend on the policy or program being analyzed.

*BAF* results can be illustrated in a variety of contexts, relative to different counterfactual scenarios:

- **Marginal and average user effects**—As discussed in the Methods Section above, the average, augmented average, and marginal *BAF* results are a function of the comparison between the specified anticipated baseline scenario (Zero, Constant and AEO Reference) and the case study increased feedstock scenario.

- **Cumulative and per-period calculations and values**—Per-period values, calculated using the formulas from the section above, illustrate *BAF* values specific to an individual point in time. By using intertemporal models, these periodic *BAF*s can vary widely from period to period as land management and forest harvest intervals adjust to the new biomass demand shock. This explains the variable nature of the periodic calculations. A 2015–2060 average is calculated to represent the average periodic *BAF* over the entire time frame of the analysis. Cumulative *BAF*s are calculated by taking the cumulative value of each term in the *BAF* equation over time. The cumulative value offers insight into potential anthropogenic biogenic carbon-based emissions effects over a specified future time horizon relative to the future anticipated baseline, whereas a single value at a point in time only offers insight into periodic deviations from the baseline. Calculation of the *BAF* using cumulative and average values can smooth out the fluctuations in equation terms per period and provide a more stable estimate of net biogenic emissions over time.

Using the Zero versus the AEO or Constant Biomass baseline as the basis of analysis led to different *BAF* values. The Zero Biomass baseline comparison to the case study projection captures all anticipated biomass users, whereas the Constant Biomass comparison focuses on new users. Depending on the policy application of the framework, either of these approaches may be more appropriate. Also, the means for considering the results over time (averaged per-period *BAF*s versus cumulative) led to different *BAF* values. Per-period values, calculated using the formulas from the section above, illustrate *BAF* values specific to an individual point in time, which might be useful in some policy applications but not relevant for others. Given the nature of modeling methods employed, periodic *BAF*s can vary widely from period to period as land management and forest harvest intervals adjust to the new biomass demand shock.

The supplemental data and information section provides the illustrative results and discussion for the various feedstock- and region-specific case studies. Data presented in this supplemental section include projected equation term values for each simulation period for emissions fluxes and cumulative emissions, using the average and marginal counterfactual approaches.

## 4.2. Overview of the Illustrative Results

Table L-2 provides illustrative values for *NBE* and the *BAF*s for each of the three case studies. These values are based on cumulative emissions totals for a simulation horizon that extends to 2060. In each case, *NBE* and *BAF* values are calculated relative to the Zero Biomass counterfactual scenario and, thus, represent an average regional *BAF* for all current and anticipated expansion in biogenic feedstock consumption from the additional 1 million dry ton feedstock demand shock. All *BAF* equation terms presented in Table L-2 can be replicated based on the cumulative "average" value tables provided in the Supplemental Information section of this appendix, referencing the 2055–2060 simulation period.

Table L-2 presents estimated landscape attributes for each of the three case studies and concludes with two illustrative *BAF* values (with and without default process attributes *P* and *L*, which are assumed to be 1 and 1.1, respectively, for consistency with previous appendices). The third column of the table represents relative growth emissions, or the difference in cumulative forest carbon sequestration between the two scenarios. The fourth column represents relative removal (or harvest) emissions. Note that either of these columns could yield a positive or negative value, depending on the relative difference in these cumulative fluxes between the case study and Zero Biomass baseline scenario. For instance, a positive value in the relative removal column means that forest harvest emissions increase with the additional biogenic feedstock demand in the case study. Relative net growth (in the fifth column) is the sum of relative growth and relative removals. Dividing this absolute emissions change by the regional *PGE* term in the ninth column yields the *GROW* term for the *NBE* equation. Columns six and seven represent relative emissions changes for those fluxes captured by the *SITETNC* term (Table L-1 provides a list of all biogenic carbon-based fluxes included in *AVOIDEMIT* and *SITETNC* for this application).

*NBE* (eighth column) is the sum of all relative landscape attributes in columns five through seven. In this particular application, *PGE* represents the total *PGE* for the region of assessment. This value represents cumulative additional consumption of biogenic feedstocks for energy generation (in million $tCO_2e$) for the feedstock scenario over the future time horizon of assessment (2015–2059), and relative to the Zero Biomass case). Thus, this is a regional *PGE* term that could potentially be used to calculate the regional ratios for *GROW, AVOIDEMIT*, and *SITETNC* (depending on the policy program or context).

The final columns represent proof-of-concept *BAF* values for the region and feedstock case study. The first *BAF* value does not adjust for process attributes *P* and *L*. Both the roundwood and logging residue case studies find a long-term cumulative *BAF* value that is very close to 0 or slightly negative in the Southeast roundwood case. The Corn Belt corn stover simulations result in a projected long-term cumulative *BAF* of 0.15, which suggests that 85% of *PGE* released during conversion at a stationary source would be reabsorbed by the landscape.

**Table L-2. Illustrative *BAF* Values for the Future Anticipated Baseline Case Studies: Cumulative Average Results from 2015–2060.**

| | | *Relative* Growth & Removals | | *Relative* Carbon Fluxes | | | *Relative* Total Carbon Flux & Biogenic Emissions | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Scenario | Time Scale | *Relative* Growth Emissions (million $tCO_2e$) | *Relative* Removals Emissions (million $tCO_2e$) | *Relative* Net Growth (GROW = Relative growth – relative removals) (million $tCO_2e$) | *Relative* Avoided Emissions (AVOIDEMIT) (million $tCO_2e$) | *Relative* Net Landscape Emissions (SITE_TNC) (million $tCO_2e$) | Net Biogenic Emissions (NBE): Sum of all relative carbon fluxes (million $tCO_2e$) | Potential Gross Emissions (PGE): All Additional Biogenic Feedstock Consumption (million $tCO_2e$) | Assessment Factor (BAF) (Ratio of relative total carbon flux to relative feedstock flux) | Adjustment Factor (BAF) with Process-Based Equation Terms P and L |
| **SE Roundwood** | 2015–2060 | −37 | 17 | −20 | −0.6 | −3 | −24 | 672 | **−0.03** | **−0.03** |
| **PNW Logging Residues** | 2015–2060 | −14 | 16 | 2 | 0 | 4 | 7 | 155 | **0.04** | **0.04** |
| **CB Corn Stover** | 2015–2060 | NA | NA | 0 | 0 | 16 | 16 | 108 | **0.15** | **0.16** |

# 5. Case Study Details

## 5.1. Southeast Roundwood

It is important to consider the regional effects of additional feedstock expansion given regional differences in forest species composition, management techniques, hardwood/softwood mixes, and forest products industry. For example, softwood plantation pine systems are common in the Southeast, and such plantations involve more intensive management but shorter rotations than typical hardwood stands in other regions such as the Northeast. Thus, high levels of emissions from biomass removals could occur more frequently on the landscape in the Southeast, but the carbon payback period could be shorter.

### 5.1.1. Marginal Effects for the Southeast Roundwood Case Study

Table L-3 displays average periodic and cumulative biogenic $CO_2$ emissions results for the marginal estimated landscape factor calculations for three separate portions of the simulation horizon (2015–2029, 2015–2044, and 2015–2060).[3] As noted previously, the marginal effect refers to a net change in regional landscape-level emissions and biogenic $CO_2$ consumption for the feedstock case study relative to the AEO Reference baseline. Estimated per-period landscape factors vary over time for the marginal case, reflecting the cyclical nature of terrestrial $CO_2$ fluxes from forest management, though this variation is smoothed by averaging over time. Initially, emissions intensity is negative and relatively large in magnitude, reflecting a net increase in carbon

[3] Note that an estimated landscape factor has the same interpretation as the *BAF* without process attribute terms *P* and *L* (presented in Equation 6 of this appendix).

sequestration on the landscape driven by land-owner investment decisions (anticipatory planting) and harvest timing decisions in response to the anticipated long-term demand shift for roundwood-derived biomass. That is, landowners plant new trees and delay harvests in an effort to meet this long-term increase in demand.

Furthermore, the Southeast region is a unique region with historically high levels of observed land use exchanges between agriculture and forestry (Wear and Gries, 2002; Milesi et al., 2003). This phenomenon is evident in the Southeast case study results, as afforestation and pasture-to-cropland transitions occur in response to the added roundwood feedstock requirement leading to periodic fluctuations evident in the *BAF* equation terms and the estimated landscape factor itself. These land use changes can cause large periodic fluctuations in *SITETNC* emissions as new sources of carbon sequestration from afforested stands affect the projected terrestrial carbon balance (as seen in 2035 and 2040). In addition to land use change, differences in forest management techniques and shorter rotations in the Southeast relative to other regions lead to more variability in the *GROW* term as high levels of harvest emissions occur more frequently and forest carbon stocks recover more rapidly.

**Table L-3. Southeast Roundwood Landscape Factor Results (Marginal User).**

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | 2015–2030 | 2015–2045 | 2015–2060 |
| ***SE Roundwood Marginal User*** | ------ *additional emissions (t $CO_2$) from AEO Reference case baseline level* ------ | | | | |
| | **GROW** | **Per Period** | -478 | -444 | -587 |
| | **SITETNC** | | -129 | -116 | -118 |
| | **AVOIDEMIT** | | 1.7 | 1.5 | 1.3 |
| | **PGE** | | 917 | 1,375 | 1,528 |
| | **Estimated Landscape Factor** | | -0.66 | -0.41 | -0.46 |
| | *Cumulative additional (t $CO_2$) from AEO Reference baseline level* | | | | |
| | **GROW** | **Cumulative** | -6,363 | -12,505 | -25,610 |
| | **SITETNC** | | -2,920 | -4,456 | -6,299 |
| | **AVOIDEMIT** | | 25 | 44.7 | 56.3 |
| | **PGE** | | 13,750 | 41,250 | 68,750 |
| | **Estimated Landscape Factor** | | -0.67 | -0.41 | ***-0.46*** |

Cumulative *BAF*s are smoother and less variable overall when compared with the periodic *BAF*s. However, the average of all periodic *BAF* values over the simulation period through 2060 is extremely close to the cumulative landscape factor. Thus, expanded roundwood consumption in the Southeast results in a net reduction in biogenic $CO_2$ emissions relative to the AEO Reference baseline.

### 5.1.2. Average Effects for the Southeast Roundwood Case Study

Average effects are displayed in Table L-4. These results include the net change in biomass consumption and emissions for existing levels of consumption, planned expansion, and the additional case study feedstock requirement. Periodic landscape factors are more stable (less variable) under this approach than the marginal effects above, in part because the additional biogenic $CO_2$ in the denominator includes the biomass consumption already projected to take place. Changes in the denominator are not overwhelmed by the large landscape-level emissions changes present in the numerator. Estimated landscape factors for the "average user" are positive at the beginning of the simulation horizon when the increase in biomass consumption has its greatest effect but decrease over time as landscape biogenic carbon balances recover. Like the "marginal" periodic landscape factors, average periodic landscape factors fluctuate over time and the average by 2060 is less than 0 at −0.07.

**Table L-4. Southeast Roundwood Landscape Factor Results (Average User).**

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | 2015–2029 | 2015–2044 | 2015–2060 |
| *SE Roundwood Average User* | *------ additional emissions (t $CO_2$) from Zero Biomass baseline level ------* | | | | |
| | **GROW** | **Per Period** | 2,315 | 138 | -769 |
| | **SITETNC** | | 651 | 45 | -155 |
| | **AVOIDEMIT** | | -12.3 | -9.4 | -12.5 |
| | **PGE** | | 12,378 | 13,670 | 14,069 |
| | **Estimated Landscape Factor** | | 0.24 | 0.01 | -0.07 |
| | *Cumulative additional emissions (t $CO_2$) from Zero Biomass baseline level* | | | | |
| | **GROW** | **Cumulative** | 49,031 | 18,433 | -20,308 |
| | **SITETNC** | | 13,211 | 4,802 | 3,519 |
| | **AVOIDEMIT** | | -228 | -325.3 | -604.8 |
| | **PGE** | | 225,072 | 449,508 | 672,489 |
| | **Estimated Landscape Factor** | | 0.28 | 0.05 | ***-0.03*** |

Cumulative landscape factors end with a similar total in 2060 to the periodic average (−0.03). Note that this result differs from the previous "marginal" user landscape factor. An "average" value includes the landscape-level emissions effect of all biomass users (current, planned, and the additional roundwood consumption source), whereas the marginal case captures only the change in roundwood consumption (relative to all current and planned sources). The key difference here is that the marginal result is capturing land management changes early in the simulation horizon (afforestation, longer forest rotations) in anticipation of the long-term increase in roundwood demand. Much of the emissions effect of moving from zero biomass consumption to the feedstock case study is captured in the AEO Reference baseline, so the resulting change from AEO Reference to the roundwood feedstock case is only capturing the additional emissions and biomass consumption attributable to the additional roundwood demand source. Figure L-1 compares this case study's cumulative average user trend *BAFs* with the average regional *BAF*s for the AEO Reference case baseline scenario presented in Appendix K (when comparing the AEO Reference case baseline to the Zero Biomass Baseline scenario).

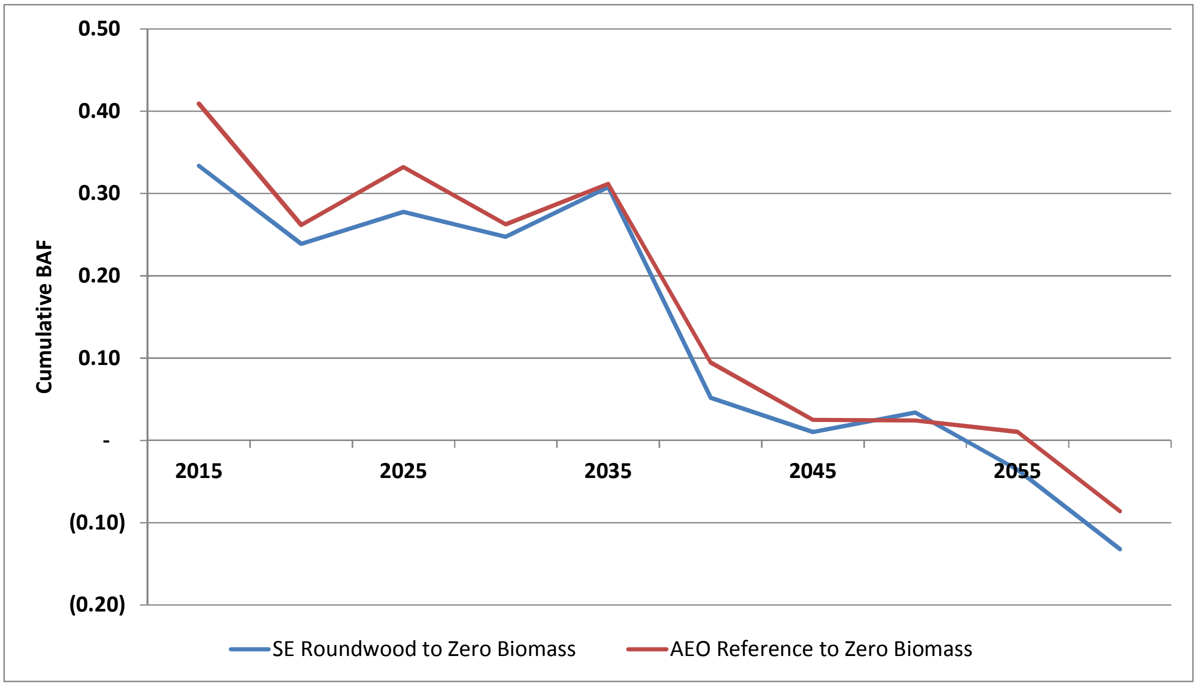


**Figure L-1. Comparison of Average Cumulative Landscape Factors in the Southeast Region for the AEO Reference Case Baseline and Roundwood Case Study Relative to the Zero Biomass Scenario.**

## 5.2. Pacific Northwest (PNW) Logging Residues

The PNW logging residues case study simulates a demand for additional feedstock that is met entirely from soft and hardwood logging residues. This case study helps illustrate the potential biogenic $CO_2$ effects of increased demand for logging residues as a bioenergy feedstock. It is important to note that the FASOM-GHG model divides the PNW into western and eastern regions, reflecting differences in ecological, environmental, and production processes on either side of the Cascade Range in Oregon and Washington. This analysis only includes the western portion of the PNW where cool, relatively dry summers and mild, wet winters yield highly productive Douglas-fir, hemlock, and spruce forests. A full evaluation of the PNW region would require including the eastern portions of Oregon and Washington, which are primarily agricultural regions with limited market interaction with the area included in this assessment.

### 5.2.1. Marginal Effects for the PNW Logging Residue Case Study

Table L-5 displays landscape factors for the PNW logging residue case study (marginal user case). Unlike the roundwood case studies previously examined, projected marginal *BAFs* are positive at the beginning of the analysis, quickly become negative through the near to medium term, and then become positive toward the end of the simulation horizon. This trend holds for both the periodic and cumulative marginal user calculations. Early in the simulation horizon, the model projects that the additional biomass requirement leads to increased forest harvest emissions. An increase in logging residue demand leads to a net increase in roundwood harvests for other products in order to meet the additional residue demand. Then, afforestation and forest management responses to the feedstock requirement lead to an increase in biogenic carbon sequestration (hence, large negative values for *GROW*), resulting in negative landscape factors from 2020–2040. Over the long term, however, this effect flips as harvest emissions outweigh growth in landscape-level biogenic carbon sequestration.

The large emissions increase in *GROW* leads to high positive values for the periodic landscape factors (greater than 1) and flips the sign for the cumulative landscape factor by 2050. The average periodic landscape factor from 2015–2055 is 0.25, and the cumulative landscape factor in 2055 is slightly higher at 0.3. Thus, relative to the AEO Reference baseline, isolated expansion in logging residue consumption in the PNW would lead to a slight increase in biogenic $CO_2$ emissions.

**Table L-5. PNW Logging Residue Landscape Factor Results (Marginal User).**

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | 2015–2030 | 2015–2045 | 2015–2060 |
| *PNW Logging Residue Marginal User* | ------ *additional emissions (t $CO_2$) from AEO Reference case baseline level* ------ | | | | |
| | **GROW** | **Per Period** | -400 | -141 | 261 |
| | **SITETNC** | | -36 | 96 | 126 |
| | **AVOIDEMIT** | | 1.0 | 0.3 | -0.2 |
| | **PGE** | | 917 | 1,375 | 1,528 |
| | **Estimated Landscape Factor** | | -0.47 | 0.03 | 0.25 |
| | *Cumulative additional emissions (t $CO_2$) from AEO Reference case baseline level* | | | | |
| | **GROW** | **Cumulative** | -3,531 | -1,748 | 14,197 |
| | **SITETNC** | | 214 | 3,630 | 6,408 |
| | **AVOIDEMIT** | | 15 | 8.5 | -7.5 |
| | **PGE** | | 13,750 | 41,250 | 68,750 |
| | **Estimated Landscape Factor** | | -0.24 | 0.05 | ***0.30*** |

### 5.2.2. Average Effects for the PNW Logging Residue Case Study

For the PNW Logging Residue case, average user landscape factors, or the combined effects of current consumption, planned expansion, and the additional feedstock consumption source, trend toward 0 over time (Table L-6). Net emissions decrease rapidly initially due to additional tree planting and changes in forest management in response to the anticipated feedstock demand. Figure L-2 provides a comparison of cumulative landscape factor values for the AEO Reference and PNW logging residue scenarios, respectively, relative to the Zero Biomass case. The additional logging residue feedstock demand leads to a slight reduction in emissions intensity over the medium term due to anticipatory land management, but a slight increase in emissions over the long term due to sustained harvest emissions that increase with the demand for logging residues. The cumulative landscape factor is close to 0 (0.04) but positive in the 2050–2060 assessment period.

**Table L-6. PNW Logging Residue Landscape Factor Results (Average User).**

| Case Study | Term | Emissions Projection Method | Time Period 2015–2030 | 2015–2045 | 2015–2060 |
|---|---|---|---|---|---|
| *PNW Logging Res Average User* | ------ *additional emissions (t $CO_2$) from Zero Biomass baseline level* ------ | | | | |
| | **GROW** | **Per Period** | -80 | -784 | -195 |
| | **SITETNC** | | -29 | -33 | 39 |
| | **AVOIDEMIT** | | 0.7 | 0.0 | -0.2 |
| | **PGE** | | 2,584 | 3,119 | 3,301 |
| | **Estimated Landscape Factor** | | -0.04 | -0.26 | -0.05 |
| | *Cumulative additional emissions (t $CO_2$) from Zero Biomass baseline level* | | | | |
| | **GROW** | **Cumulative** | 9,923 | -12,398 | 2,355 |
| | **SITETNC** | | 2,158 | 1,591 | 4,358 |
| | **AVOIDEMIT** | | 9 | -0.4 | -8.5 |
| | **PGE** | | 45,098 | 99,893 | 154,896 |
| | **Estimated Landscape Factor** | | 0.27 | -0.11 | ***0.04*** |

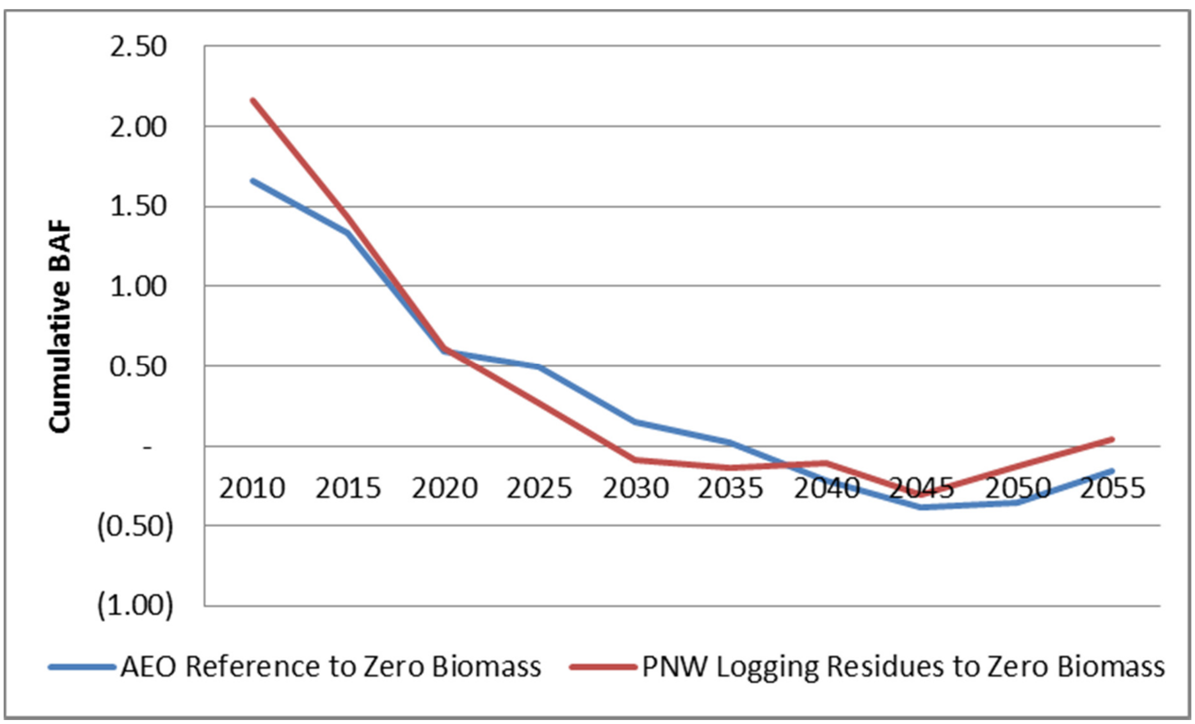


**Figure L-2. Comparison of Average Cumulative Landscape Factors in the PNW Region for the AEO Reference Case and PNW Logging Residue Case Study Relative to the Zero Biomass Scenario.**

## 5.3. Corn Belt Corn Stover

The Corn Belt corn stover case study applies the same additional 1 million ton biomass shock over time to the Corn Belt region and requires this additional biomass demand to be met exclusively with corn residues from this region. However, there are some accounting differences between the corn stover case studies and the previous two roundwood scenarios. The *GROW* term defaults to 0 for agricultural biomass sources in this methodology. The assumption is that, with annual crops, biogenic $CO_2$ "growth" in this context equals what is harvested (removed) from the system for energy generation. However, because this effort seeks to also capture changes in landscape-level emissions, forest tree carbon and harvest emissions changes engendered by the increase in corn stover removal are included in the *SITETNC* term.

One important point regarding the Corn Belt case study is that for each alternative future baseline, a significant amount of corn residue is projected to be harvested in the Corn Belt for producing cellulosic ethanol to meet the RFS2 advanced biofuel mandates (approximately 6.2 billion gallons). Thus, the additional biogenic feedstock constraint will pull from corn stover resources above and beyond what is used to produce cellulosic ethanol.

### 5.3.1. Marginal Effects for the Corn Belt Corn Stover Case Study

Table L-7 displays marginal *BAF* results for the Corn Belt corn stover case study. Unlike the roundwood scenarios, emissions fluxes are relatively stable over time. For the majority of the simulation horizon, periodic landscape factors are positive (but less than 1), which is driven by increased emissions from *SITETNC* carbon pools on the landscape. After 2015, the proportion of conventional tillage to no-till and conservation till stays relatively constant (thus, there are only minor biogenic soil carbon effects from increased residue harvesting). The majority of *SITETNC* emissions are due to forest harvest fluxes and small levels of deforestation for crop production in response to the additional feedstock demand.

**Table L-7. Corn Belt Corn Stover Landscape Factor Results (Marginal User).**

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | 2015–2030 | 2015–2045 | 2015–2060 |
| *CB Corn Stover Marginal User* | ------ *additional emissions (t $CO_2$) from AEO Reference case baseline level* ------ | | | | |
| | GROW | Per Period | 0 | 0 | 0 |
| | SITETNC | | 183 | 265 | 123 |
| | AVOIDEMIT | | 0.0 | 0.1 | 0.0 |
| | PGE | | 917 | 1,375 | 1,528 |
| | Estimated Landscape Factor | | 0.20 | 0.19 | 0.08 |
| | *Cumulative additional emissions per ton of additional feedstock usage (t $CO_2$) from AEO Reference case baseline level* | | | | |
| | GROW | Cumulative | 0 | 0 | 0 |
| | SITETNC | | 2,645 | 7,838 | 5,435 |
| | AVOIDEMIT | | 0 | 1.7 | 2.0 |

| | | | | | |
|---|---|---|---|---|---|
| | **PGE** | | 13,750 | 41,250 | 68,750 |
| | **Estimated Landscape Factor** | | 0.19 | 0.19 | ***0.08*** |

Cumulative *BAF* values are also relatively stable over time, ending up at 0.08. Thus, 40 years after the initial corn stover demand shock, only a small portion of biogenic $CO_2$ emissions from additional corn stover removals are not balanced by landscape biogenic $CO_2$ sequestration from land management changes.

### 5.3.2. Average Effects for the Corn Belt Corn Stover Case Study

Although the marginal effects in this case study are relatively stable, average effects fluctuate considerably over time in the Corn Belt region (Table L-8). The overall trend is similar to the alternative baseline Corn Belt regional results presented in Appendix K (AEO Reference relative to Zero Biomass) in that biogenic emissions are highly negative (high level of sequestration) in 2015 and then increase over time (see Figure L-3). However, the additional corn stover requirement increases net biogenic $CO_2$ emissions (hence the positive periodic flux values in the marginal case), which essentially shifts the *BAF* trajectory up for the majority of the simulation horizon. Note that the two *BAF* trajectories below converge over the long term, indicating a rise in land use change emissions in the AEO Reference baseline in the long term.

In general, these results show that although biogenic $CO_2$ emissions from corn stover biomass removals in the Corn Belt might be predominately offset by landscape-level $CO_2$ accumulation, additional expansion of corn stover demand could increase the value of agricultural land relative to other uses, which could drive land use change and increase net emissions (especially if the land is converted to agricultural use from forestry). However, even with the resulting emissions effects, biogenic $CO_2$ emissions from corn stover consumption in this scenario are almost fully offset by landscape-level $CO_2$ changes in this case study scenario.

**Table L-8. Corn Belt Corn Stover Landscape Factor Results (Average User).**

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | **2015–2030** | **2015–2045** | **2015–2060** |
| ***CB Corn Stover Average User*** | *------ additional emissions (t $CO_2$) from Zero Biomass baseline levels ------* | | | | |
| | **GROW** | **Per Period** | 0 | 0 | 0 |
| | **SITETNC** | | -3,047 | -2,064 | 433 |
| | **AVOIDEMIT** | | -2.4 | -2.1 | -0.9 |
| | **PGE** | | 1,611 | 2,149 | 2,337 |
| | **Estimated Landscape Factor** | | -1.89 | -0.96 | 0.81 |

| Case Study | Term | Emissions Projection Method | Time Period | | |
|---|---|---|---|---|---|
| | | | 2015-2030 | 2015-2045 | 2015-2060 |
| | *Cumulative additional emissions ($t\ CO_2$) from Zero Biomass baseline level* | | | | |
| | **GROW** | Cumulative | 0 | 0 | 0 |
| | **SITETNC** | | 48,791 | -65,015 | 16,425 |
| | **AVOIDEMIT** | | -37 | -64.2 | -42.4 |
| | **PGE** | | 26,771 | 67,069 | 107,770 |
| | **Estimated Landscape Factor** | | -1.82 | -0.97 | ***0.15*** |

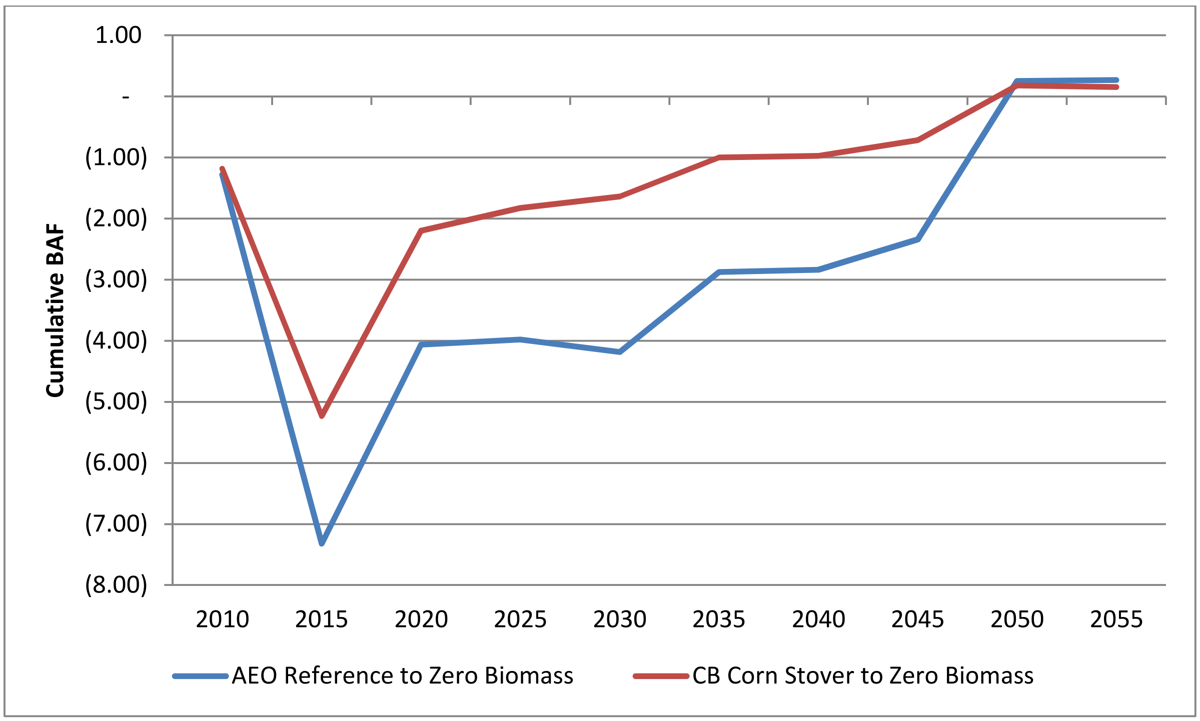


**Figure L-3. Comparison of Average Cumulative Landscape Factors in the Corn Belt Region for the AEO Reference Case Baseline and Corn Stover Case Study Relative to the Zero Biomass Scenario.**

## 6. Summary

The illustrative *BAF* values presented in this appendix do not reflect any specific policies or programs; rather they are estimated outcomes based on the baseline and scenario constructs, as well as the assumptions and parameters in the modeling system. The goal of this exercise is to illustrate the functionality of the future anticipated baseline approach and to provide insights into potential effects of biogenic feedstock production and consumption, the possible directionality of results, investor/market behavior, and magnitude of additionality (per the given specific assumptions and modeling system). There are different temporal and spatial scales that could be used, and choices pertaining to these factors can impact results.

Ultimately, the illustrative case studies and estimated values in this appendix are meant to demonstrate the flexibility of the framework as well as the importance of decisions made in terms of how results are to be calculated. Therefore, decisions about time, space, data aggregation, etc., all should be specific to the policy or program to which the framework is applied.

Appendix M provides an overview and discussion of illustrative case study results as well as sensitivities derived from both the retrospective reference point and future anticipated baseline applications. In that appendix, the future anticipated results reflect the comparison of the 1 million ton increased feedstock scenario to the Zero Biomass scenario to provide an estimate of the average biogenic $CO_2$ emissions effect for all existing and planned biogenic feedstock consumption at national and regional scales and also applies the cumulative calculation method.

# 7. References


Milesi, C., Elvidge, C. D., Nemani, R. R., & Running, S. W. (2003). Assessing the impact of urban land development on net primary productivity in the southeastern United States. *Remote Sensing of Environment 86*(3):401-410.

Wear, D. N. and Greis, J.G. (2002). The Southern Forest Resource Assessment. http://www.srs.fs.usda.gov/sustain/report/pdf/sfra_summary_e.pdf


# 8. Supplemental Data and Information

## 8.1. Details on FASOM-GHG Carbon Accounting

FASOM-GHG incorporates detailed accounting for GHGs emitted from and sequestered by forestry and agricultural activities and land use change in the United States, including the dynamics of carbon sequestration in forests, soils, and wood products. In addition, the model tracks GHG emission reductions in other sectors caused by mitigation actions in the forest and agricultural sectors. In addition to $CO_2$, FASOM-GHG's accounting also includes $CH_4$, and $N_2O$. In this section, we provide additional information on the $CO_2$ accounting functions and parameters used within the model.

To compare landscape-level emissions across baselines, the following $CO_2$ flux categories are aggregated to yield a total net emissions flux. This aggregation is calculated for every time step in the simulation horizon (5-year time steps). Then, annual averages are evaluated for different portions in the simulated horizon to highlight the importance of temporal dynamics.

### 8.1.1. FASOM-GHG Biogenic Feedstock Growth Functions

For FASOM-GHG output, the *GROW* term focuses primarily on forest growth in the context of longer rotation woody biomass (i.e., roundwood). Short rotation woody crops such as hybrid poplar and willow do occur over time frames longer than a year and would typically be produced in plantations (which would achieve a steady state of $CO_2$ flux; thus, growth would be in balance with removals). The agricultural feedstocks complete an entire growth/harvest/combustion cycle entirely within a year (and thus any $CO_2$ sequestration in the feedstock would also be balanced by its removal and use). In FASOM-GHG the net forest carbon growth from period to period equated to *GROW* in the *BAF* equation would be best represented with the change in tree carbon over that same time period less any removals. This could be evaluated both regionally and nationally. Note that this net change in tree carbon would include both growth of trees that did not get harvested as well as a loss associated with the trees that did get harvested (removals).

FASOM-GHG tree carbon calculation is based on two primary sources: timber yields and a set of factors that convert those yields to carbon. With the exception of the Pacific Northwest-West (PNWW) region, the timber yields come from the ATLAS model (Mills and Adams, 2007) as used in the national 2005 RPA Assessment Update (Haynes et al., 2007). In the PNWW the yields are based on FIA plots "grown" using the Forest Vegetation Simulator (FVS) and then averaged over strata. The yields include options for partial harvesting regimes of one or more thinning entries only in the PNWW, Southeast (SE), and South central (SC) regions. The conversion of yields for all management regimes including those that involve partial harvests to carbon are based on Smith et al. (2006).

The growing stock volumes ($V_A$) from the FASOM-GHG yield tables are in thousands of cubic feet per acre and therefore must be converted to cubic meters per hectare ($V_H$) for use in the carbon equations. The volumes per acre are converted using the following equation:[4]

$$\mathbf{V_H = \frac{1000 \cdot V_A}{14.29}} \qquad \textbf{(EQ. L.7)}$$

To convert these volumes to carbon for the regions and forest types in Smith et al. (2007) were mapped to FASOM-GHG regions, and forest types and weighted averages of the parameters were calculated based on acreages from FIA. In addition to the basic Smith et al. (2007) equations (the 1605b tables), the FASOM-GHG parameters include tree carbon and young stand adjustment from an update by Jim Smith in 2007. The *L1*, *L2*, and *L3* parameters[5] displayed in Table L-9 are for the live tree mass equation, the *D2* and *D3* parameters are for the dead tree mass equation, and the *C1* parameter is used to "ramp up" the mass in young stands (because they may have no growing stock volume).

**Table L-9. FASOM-GHG Live and Dead Tree Biomass Equation Parameters.**

| Region and Forest Type | Carbon Equation Parameters | | | | | |
|---|---|---|---|---|---|---|
| | L1 | L2 | L3 | D2 | D3 | C1 |
| **CB** | | | | | | |
| SOFT | 14.434 | 2.937 | 0.804 | 1.754 | 0.397 | 8.74 |
| HARD | 29.651 | 2.493 | 0.861 | 2.996 | 0.266 | 9.89 |
| **LS** | | | | | | |
| SOFT | 14.434 | 2.937 | 0.804 | 1.754 | 0.397 | 8.74 |
| HARD | 29.651 | 2.493 | 0.861 | 2.996 | 0.266 | 9.89 |

[4] Note that this equation is different from the 2008 FASOM-GHG documentation Section 13.2.1.1 where adjustments are made to the growing stock volumes up to total volumes as the model has been updated.
[5] Note that these are the values we use after the weighted average from FIA process. They therefore do not match the 1605b values exactly.

| Region and Forest Type | | Carbon Equation Parameters | | | | | |
|---|---|---|---|---|---|---|---|
| | | L1 | L2 | L3 | D2 | D3 | C1 |
| **NE** | | | | | | | |
| | SOFT | 35.372 | 2.062 | 0.85 | 4.056 | 0.233 | 9.941 |
| | HARD | 31.51 | 2.598 | 0.843 | 3.108 | 0.266 | 8.79 |
| **RM** | | | | | | | |
| | SOFT | 11.082 | 2.836 | 0.776 | 2.543 | 0.402 | 9.749 |
| | HARD | 11.082 | 2.836 | 0.776 | 2.543 | 0.402 | 9.749 |
| **PSW** | | | | | | | |
| | SOFT | 33.524 | 2.022 | 0.852 | 3.099 | 0.12 | 35.277 |
| | HARD | 20.852 | 2.632 | 0.836 | 3.211 | 0.343 | 9.889 |
| **PNWW** | | | | | | | |
| | DOUG_FIR | 31.823 | 1.102 | 0.949 | 5.691 | 0.336 | 6.1 |
| | OTH_SWDS | 17.599 | 1.822 | 0.881 | 1.847 | 0.554 | 7.081 |
| | HARD | 20.852 | 2.632 | 0.836 | 3.211 | 0.343 | 9.889 |
| **PNWE** | | | | | | | |
| | SOFT | 33.524 | 2.022 | 0.852 | 3.099 | 0.12 | 35.277 |
| | HARD | 20.852 | 2.632 | 0.836 | 3.211 | 0.343 | 9.889 |

| Region and Forest Type | | Carbon Equation Parameters | | | | | |
|---|---|---|---|---|---|---|---|
| | | L1 | L2 | L3 | D2 | D3 | C1 |
| SC | | | | | | | |
| | NAT_PINE | 37.244 | 1.553 | 0.846 | 1.203 | 0.271 | 5.743 |
| | OAK_PINE | 30.637 | 2.734 | 0.798 | 1.133 | 0.337 | 5.986 |
| | PLNT_PINE | 30.652 | 1.899 | 0.815 | 1 | 0.138 | 4.107 |
| | SOFT | 37.244 | 1.553 | 0.846 | 1.203 | 0.271 | 5.743 |
| | BOT_HARD | 25.128 | 4.691 | 0.741 | 4.056 | 0.137 | 7.986 |
| | HARD | 25.128 | 4.691 | 0.741 | 4.056 | 0.137 | 7.986 |
| | UP_HARD | 46.794 | 1.964 | 0.876 | 2.396 | 0.186 | 9.381 |
| SE | | | | | | | |
| | NAT_PINE | 34.818 | 1.242 | 0.892 | 1 | 0.324 | 4.91 |
| | OAK_PINE | 21.645 | 2.626 | 0.811 | 1 | 0.351 | 4.351 |
| | PLNT_PINE | 34.148 | 1.157 | 0.908 | 1 | 0.265 | 4.8 |
| | SOFT | 34.818 | 1.242 | 0.892 | 1 | 0.324 | 4.91 |
| | BOT_HARD | 22.811 | 3.978 | 0.756 | 1.747 | 0.337 | 5.498 |
| | HARD | 22.811 | 3.978 | 0.756 | 1.747 | 0.337 | 5.498 |
| | UP_HARD | 28.976 | 3.213 | 0.803 | 2.256 | 0.257 | 6.108 |

Tree carbon is calculated as the sum of live mass ($C_{live}$):

$$\mathbf{C_{live}} = \left(\mathbf{1 - e^{\left(\frac{-age}{C1}\right)}}\right)\left(\mathbf{L1 + L2 \cdot V_H^{L3}}\right) \qquad \textbf{(EQ. L.8)}$$

And dead mass ($C_{dead}$):

$$\mathbf{C_{dead}} = \left(\mathbf{1 - e^{\left(\frac{-age}{C1}\right)}}\right)\left(\mathbf{D2 \cdot V_H^{D3}}\right) \qquad \textbf{(EQ. L.9)}$$

And converted to tree carbon per acre ($C_{tree}$) based on half of the mass being the carbon content:

$$\mathbf{C_{tree}} = \frac{\mathbf{(C_{live} + C_{dead})}}{\mathbf{0.5 \cdot 2.471}} \qquad \textbf{(EQ. L.10)}$$

### 8.1.2. FASOM-GHG Functions Relating to Changes in Site Emissions

The *SITETNC* term represents the feedstock production site-level difference in the net $CO_2$ flux to the atmosphere when biogenic feedstocks are used for bioenergy compared with a previous use/activity considering both emissions and sequestration changes (e.g., in the case of land use change or residue removal). In FASOM-GHG it may be difficult to differentiate between forest organic soil changes and forest litter and understory changes resulting from harvest residual removal. FASOM-GHG has stable soil carbon estimates for each of the major land use classifications (cropland, pasture, afforestation, and forest). Upon land use change there is a linear transition between the prior soil carbon level and that of the new use. The change in these soil carbon accounts resulting from additional biomass utilization can be evaluated by simply taking the difference between scenarios. The litter and understory carbon is based on a forest floor equation along with estimates of understory and coarse woody debris. Unlike tree carbon, these values adapted from Smith et al. (2007) are based solely on forest age, region, and forest type. The parameters for the equation are provided by Table L-10.

**Table L-10. FASOM-GHG Forest Floor Biomass Equation Parameters.**

| Region and Forest Type | | Forest Floor Carbon Parameters | | | | | |
|---|---|---|---|---|---|---|---|
| | | A | B | C | D | und | cwd |
| **CB** | | | | | | | |
| | SOFT | 42 | 57.6 | 23.9 | 13.9 | 2.1 | 13.8 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 2.4 | 10.8 |
| **LS** | | | | | | | |
| | SOFT | 42 | 57.6 | 23.9 | 13.9 | 2.1 | 13.8 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 2.4 | 10.8 |
| **NE** | | | | | | | |
| | SOFT | 42 | 57.6 | 23.9 | 13.9 | 2.6 | 12.2 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 2.2 | 11.2 |
| **RM** | | | | | | | |
| | SOFT | 42 | 57.6 | 23.9 | 13.9 | 5.7 | 12.6 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 9.2 | 26.7 |
| **PSW** | | | | | | | |
| | SOFT | 42 | 57.6 | 23.9 | 13.9 | 4.9 | 12.8 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 2.8 | 11.5 |
| **PNWW** | | | | | | | |
| | DOUG_FIR | 87.5 | 116.7 | 27.5 | 16 | 2 | 11.9 |
| | OTH_SWDS | 87.5 | 116.7 | 27.5 | 16 | 3.2 | 15.4 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 4.5 | 3.9 |
| **PNWE** | | | | | | | |
| | SOFT | 87.5 | 116.7 | 27.5 | 16 | 3 | 14.8 |
| | HARD | 44.7 | 59.5 | 28.9 | 13.2 | 4.5 | 3.9 |
| **SC** | | | | | | | |
| | NAT_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 5.9 | 18.6 |
| | OAK_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 4.4 | 17.3 |
| | PLNT_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 5.9 | 18.6 |
| | SOFT | 20.4 | 27.1 | 12.2 | 3.8 | 5.9 | 18.6 |
| | BOT_HARD | 15.4 | 40.9 | 8.2 | 3.5 | 2.2 | 15.7 |
| | HARD | 15.4 | 40.9 | 8.2 | 3.5 | 2.2 | 15.7 |
| | UP_HARD | 15.4 | 40.9 | 8.2 | 3.5 | 3.7 | 15 |
| **SE** | | | | | | | |
| | NAT_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 6.8 | 23.9 |
| | OAK_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 4.4 | 17.3 |
| | PLNT_PINE | 20.4 | 27.1 | 12.2 | 3.8 | 6.8 | 23.9 |
| | SOFT | 20.4 | 27.1 | 12.2 | 3.8 | 6.8 | 23.9 |
| | BOT_HARD | 15.4 | 40.9 | 8.2 | 3.5 | 2.2 | 21.8 |
| | HARD | 15.4 | 40.9 | 8.2 | 3.5 | 2.2 | 21.8 |
| | UP_HARD | 15.4 | 40.9 | 8.2 | 3.5 | 4.4 | 24.3 |

The litter, understory, and coarse woody debris (U) is then calculated as:

$$\mathbf{U} = \frac{\left(\frac{\mathbf{A \cdot age}}{\mathbf{B+age}}\right) + \mathbf{C \cdot e}^{-\left(\frac{\mathbf{age}}{\mathbf{D}}\right)}}{\mathbf{2.471}} + \frac{\mathbf{(Und+Cwd) \cdot L \cdot 0.5}}{\mathbf{100 \cdot 2.471}} \qquad \textbf{(EQ. L.11)}$$

If it is the first rotation (afforestation), the C and D terms are dropped, giving:

$$\mathbf{U} = \frac{\left(\frac{\mathbf{A \cdot age}}{\mathbf{B+age}}\right)}{\mathbf{2.471}} + \frac{\mathbf{(Und+Cwd) \cdot L \cdot 0.5}}{\mathbf{100 \cdot 2.471}} \qquad \textbf{(EQ. L.12)}$$

In addition to the litter, understory, and coarse woody debris carbon pools, FASOM-GHG tracks soil carbon. The approach used is adapted from earlier work by Birdsey (1996a), which had fixed forestland carbon values in all regions except the South[6] that varied by region, while Smith et al. (2006) have all carbon in forest soils assumed to be constant over time but varied by region and forest type. Birdsey (1996a) also has soil carbon estimates for land that has been converted from both crop and pasture to forest that rises from an initial value that differs for crop or pasture land to a steady state clos but not equal to, the forestland steady-state soil carbon values. To keep soil carbon values consistent across land use types (crop, pasture, and forest), FASOM-GHG does not use any of the Birdsey (1996a) or Smith et al. (2006) values but rather a loosely based approximation of their values and trends.

To begin, the FASOM-GHG uses century-based crop and pasture soil values that are constant in each region. Table L-11 provides those values.

**Table L-11. FASOM-GHG Agricultural Soil Carbon Constants by Land Use and Region.**

| Region | FASOM-GHG Agricultural Land Use | | | | |
|---|---|---|---|---|---|
| | Cropland | Cropland_Pasture | Rangeland | Forest_Pasture | Pasture |
| CB | 15.373 | 18.751 | 18.751 | 18.751 | 18.751 |
| GP | 6.872 | 11.295 | 11.295 | 11.295 | 11.295 |
| LS | 9.946 | 13.619 | 13.619 | 13.619 | 13.619 |
| NE | 7.242 | 11.649 | 11.649 | 11.649 | 11.649 |
| RM | 5.463 | 7.955 | 7.955 | 7.955 | 7.955 |
| PSW | 10.554 | 15.862 | 15.862 | 15.862 | 15.862 |
| PNWW | 14.832 | 23.029 | | | |
| PNWE | 6.665 | 9.216 | 9.216 | 9.216 | 9.216 |
| SC | 15.415 | 20.12 | 20.12 | 20.12 | 20.12 |
| SE | 2.791 | 4.828 | 4.828 | 4.828 | 4.828 |
| SW | 6.471 | 11.882 | 11.882 | 11.882 | 11.882 |

[6] Birdsey (1996a) had minor variation (<10%) in soil carbon for southern forest over the life of a stand.

A regression analysis of the Appendix 3 afforestation soil carbon values (Birdsey, 1996b) as a quadratic function of forest stand age after pasture reversion was estimated, yielding the following functional form:

$$\mathbf{C_{soil} = int + t \cdot age + t2 \cdot age^2} \qquad \textbf{(EQ. L.13)}$$

The parameter estimates are provided in Table L-12 (values in thousand pounds of carbon per acres, not tons).

**Table L-12. FASOM-GHG Forest Soil Equation Parameters.**

| Region | Soil Carbon Parameters | | |
|---|---|---|---|
| | int | t | t2 |
| SE | 44.964 | 0.626 | −0.00337 |
| SC | 44.017 | 0.61 | −0.00322 |
| NE | 93.884 | 1.159 | −0.00461 |
| LS | 75.803 | 0.938 | −0.00383 |
| CB | 48.509 | 0.586 | −0.0023 |
| GP | 46.655 | 0.6 | −0.00266 |
| RM | 46.655 | 0.6 | −0.00266 |
| PNWW | 56.686 | 0.696 | −0.00287 |
| PNWE | 56.686 | 0.696 | −0.00287 |
| PSW | 56.686 | 0.696 | −0.00287 |
| SW | 56.686 | 0.696 | −0.00287 |

The forest soil constants are determined as the maximum soil carbon value achieved when the FASOM-GHG minimum harvest ages for each region, owner, forest type, site class, and management intensity are used in the $C_{soil}$ equation. The values obtained and used for the regional forest soil constants are provided in Table L-13.

**Table L-13. FASOM-GHG Forest Soil Carbon Constant by Region in Metric Tons of Carbon per Acre.**

| Region | Forest |
|---|---|
| CB | 20.561 |
| LS | 30.853 |
| NE | 40.044 |
| RM | 19.931 |
| PSW | 24.912 |
| PNWW | 22.994 |
| PNWE | 24.031 |
| SC | 14.303 |
| SE | 14.617 |

Upon conversion from an agricultural use to forest (afforestation), there is a period of soil adjustment from the prior land use fixed soil amount to the new land use fixed soil amount. The adjustment is based on the parameters from Table L-14 using time since conversion as the age. This yields the following conversion values.

**Table L-14. FASOM-GHG Soil Carbon Conversion Rates by Years Since Conversion and Region.**

| Years Since Land Conversion | Region | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | NE | CB | SC | SE | LS | PSW | PNWW | PNWE | RM |
| **0** | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| **5** | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 | 0.01 |
| **10** | 0.05 | 0.05 | 0.04 | 0.04 | 0.05 | 0.05 | 0.05 | 0.05 | 0.05 |
| **15** | 0.09 | 0.09 | 0.09 | 0.09 | 0.09 | 0.08 | 0.09 | 0.09 | 0.09 |
| **20** | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 | 0.15 |
| **25** | 0.22 | 0.22 | 0.23 | 0.23 | 0.22 | 0.22 | 0.22 | 0.22 | 0.22 |
| **30** | 0.30 | 0.31 | 0.32 | 0.32 | 0.31 | 0.30 | 0.30 | 0.30 | 0.30 |
| **35** | 0.39 | 0.39 | 0.40 | 0.40 | 0.39 | 0.39 | 0.39 | 0.39 | 0.39 |
| **40** | 0.47 | 0.47 | 0.49 | 0.49 | 0.47 | 0.47 | 0.47 | 0.47 | 0.47 |
| **45** | 0.56 | 0.56 | 0.57 | 0.58 | 0.56 | 0.55 | 0.56 | 0.56 | 0.56 |
| **50** | 0.63 | 0.63 | 0.66 | 0.66 | 0.63 | 0.63 | 0.63 | 0.63 | 0.63 |
| **55** | 0.70 | 0.70 | 0.73 | 0.73 | 0.70 | 0.70 | 0.70 | 0.70 | 0.70 |
| **60** | 0.76 | 0.76 | 0.79 | 0.79 | 0.76 | 0.76 | 0.76 | 0.76 | 0.76 |
| **65** | 0.82 | 0.82 | 0.85 | 0.85 | 0.82 | 0.81 | 0.82 | 0.82 | 0.82 |
| **70** | 0.86 | 0.86 | 0.89 | 0.89 | 0.86 | 0.85 | 0.86 | 0.86 | 0.86 |
| **75** | 0.90 | 0.90 | 0.93 | 0.93 | 0.90 | 0.90 | 0.90 | 0.90 | 0.90 |
| **80** | 0.92 | 0.92 | 0.96 | 0.96 | 0.92 | 0.92 | 0.92 | 0.92 | 0.92 |
| **85** | 0.95 | 0.95 | 0.98 | 0.98 | 0.95 | 0.94 | 0.95 | 0.95 | 0.95 |
| **90** | 0.96 | 0.96 | 1.00 | 1.00 | 0.96 | 0.96 | 0.96 | 0.96 | 0.96 |
| **95** | 0.97 | 0.97 | 1.00 | 1.00 | 0.97 | 0.97 | 0.98 | 0.97 | 0.97 |
| **100** | 0.98 | 0.98 | 1.00 | 1.00 | 0.98 | 0.98 | 0.98 | 0.98 | 0.98 |

Using the conversion rates from Table L-14 and the fixed soil carbon amounts from Tables L-12 and L-13 the carbon flux ($\Delta C_{soil}$) associated with a land movement from pasture ($C^{past}_{soil}$) to forest ($C^{for}_{soil}$) using the soil carbon conversion rate, $S_t$, in year $t$ would be calculated using the following equation:

$$\Delta \boldsymbol{C_{soil}} = (\boldsymbol{S_{t-1}} - \boldsymbol{S_t})\left(\boldsymbol{C^{for}_{soil}} - \boldsymbol{C^{past}_{soil}}\right) \qquad \textbf{(EQ. L.14)}$$

### 8.1.3. FASOM-GHG Functions Relating to Changes in Avoided Emissions

In addition to the litter, understory, and coarse woody debris discussed above, FASOM-GHG also accounts for unused fuelwood and logging residues. These are assumed to be different from the coarse wood debris in that unused fuelwood and logging residues can be either used or left to decompose onsite based on region and forest type. Specific decomposition rates from Turner et al.

(1993) and Turner et al. (1995) are applied. Table L-15 gives the FASOM-GHG coarse woody debris decomposition rates.

**Table L-15. FASOM-GHG Annual Coarse Woody Debris Decomposition Rates.**

| Forest Type | FASOM-GHG Region | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | CB | LS | NE | RM | PSW | PNWW | PNWE | SC | SE |
| **Softwood** | 0.048 | 0.048 | 0.053 | 0.02 | 0.023 | 0.027 | 0.027 | 0.057 | 0.057 |
| **Hardwood** | 0.084 | 0.084 | 0.069 | 0.082 | 0.082 | 0.082 | 0.082 | 0.082 | 0.082 |

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# Appendix M: Summary of Illustrative Forestry and Agriculture Results

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## List of Tables



# 1. Introduction

This appendix presents the application of the retrospective reference point and future anticipated baseline approaches to estimate illustrative biogenic assessment factors (*BAF)* for specific feedstocks in specific regions. Although both baseline methodologies produce illustrative assessment factors for the same feedstock-region combinations, the methods differ in structure, and the assumptions are not harmonized between the two methods.

Three case studies are presented below: Southeast roundwood, Pacific Northwest logging residues, and Corn Belt corn stover. Both baselines are applied within specific case study constructs to generate illustrative values for the framework equation terms and assessment factors. Sensitivities to regional scale, feedstock demand, equation term impacts, and time frame were also estimated, and those results are included below.

This appendix uses examples from each baseline approach to produce illustrative equation term values for three of the landscape biogenic attribute terms (*GROW, AVOIDEMIT*, and *SITETNC*) from the biogenic assessment factor equation as presented in the main report (Part 2).

$$\boldsymbol{NBE} = (\boldsymbol{PGE})(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P}) \qquad \textbf{(EQ. M.1)}$$

For simplicity, feedstock carbon losses during storage, transport, and processing ($L$) are held constant at 1.1, feedstock carbon embodied in products ($P$) is also constant at 1, and leakage associated with feedstock production (*LEAK*) is not calculated in the illustrative term calculations here. The *BAF* is calculated by dividing *NBE* by *PGE*:

$$\boldsymbol{BAF = NBE/PGE} \qquad \textbf{(EQ. M.2)}$$

For both the retrospective reference point and future anticipated baseline approaches, the interpretation of the assessment factor is the same. The assessment factor represents the ratio of net biogenic emissions to potential gross emissions. In other words, the assessment factor reflects the extent to which biogenic $CO_2$ emissions from stationary source consumption are counter-balanced by landscape-level biological carbon cycle processes.

For example, an assessment factor of 0.2 indicates that 80% of biogenic $CO_2$ emissions are counter-balanced by landscape-level carbon sequestration, and 20% contributes to atmospheric $CO_2$ concentrations. Similarly, a negative assessment factor suggests a carbon sink (e.g., from improved carbon management on the landscape). That is, where the assessment factor < 0, additional biogenic feedstock consumption leads to a net increase in landscape-level carbon sequestration.

# 2. Retrospective Reference Point Baseline: Southeast Roundwood

In this section, results are generated using a retrospective reference point baseline in which the net change in various carbon pools on the feedstock production landscape between two points of time in the past to how these pools have changed over that period. The values for this case study presented in Table M-1 represent the net biogenic $CO_2$ emissions from a hypothetical electricity facility with an electricity generating unit (EGU) that uses roundwood from the Southeast region as a biogenic feedstock. This case study also examines alternative scenarios as sensitivities evaluating the regional aggregation, roundwood removals level, *BAF* equation term inclusion, land base, and temporal scale.

**Table M-1. Biogenic Assessment Factors Derived from a Reference Point Baseline for the Southeast Roundwood Case Study.**

| Scenario | Time Scale | Growth (billion cu ft) | Removals (billion cu ft) | Growth to Removals Ratio (GROW) (removals–growth)/removals) | Avoided Emissions (AVOIDEMIT) (avoided long term sequestration)/ton removals | Net Landscape Emissions (SITETNC) (other site emissions)/ton removals | Potential Gross Emission (PGE) (million $tCO_2e$) | Assessment Factor (BAF)[2] |
|---|---|---|---|---|---|---|---|---|
| **Southeast** | 2006–2010 | 7.60 | 4.38 | −0.74 | 0 | −0.024 | 0.42 | −0.84 |
| **South Central** | 2006–2010 | 9.58 | 5.38 | −0.78 | 0 | −0.020 | 0.42 | −0.88 |
| **Combined SE/SC** | 2006–2010 | 17.16 | 9.76 | −0.76 | 0 | −0.022 | 0.42 | −0.86 |
| **SE x2 Increased Removals** | 2006–2010 | 7.60 | 5.38 | −0.41 | 0 | −0.020 | 0.42 | −0.48 |
| **SE x3 Increased Removals** | 2006–2010 | 7.60 | 6.38 | −0.19 | 0 | −0.017 | 0.42 | −0.23 |
| **SE x5 Increased Removals** | 2006–2010 | 7.60 | 9.38 | 0.19 | 0 | −0.010 | 0.42 | 0.20 |
| **SE x10 Increased Removals** | 2006–2010 | 7.60 | 14.38 | 0.47 | 0 | −0.007 | 0.42 | 0.51 |
| **Without Net Landscape Emissions** | 2006–2010 | 7.60 | 4.38 | −0.74 | 0 | NA | 0.42 | −0.81 |
| **Change Time Frame[1]** | 1966–1976 | 5.99 | 3.03 | −0.98 | 0 | −0.024 | 0.42 | −1.10 |
| **Change Time Frame** | 1977–1986 | 5.59 | 3.67 | −0.52 | 0 | −0.024 | 0.42 | −0.60 |
| **Change Time Frame** | 1987–1996 | 5.96 | 4.46 | −0.34 | 0 | −0.024 | 0.42 | −0.40 |
| **Change Time Frame** | 1997–2006 | 7.31 | 4.31 | −0.70 | 0 | −0.024 | 0.42 | −0.79 |

[1] The change in time frame sensitivities could only be conducted on timberlands, rather than all working lands (which is the land base used for all other assessment factor calculations above) because the FIA database only had information available this far into the past for timberlands.

## 2.1. Key Insights from the Retrospective Reference Point Baseline Application to Southeast Roundwood

- The current estimated assessment factor for Southeast roundwood is less than 0 at −0.84. Except in the increased removals sensitivities where timber removals are increased and eventually exceed growth (which is held constant), these assessment factors remain negative, indicating a net increase in landscape-level carbon sequestration.
- Aggregating to a larger region (Southeast and South Central) or removing site land use and management biogenic $CO_2$ change from the equation (*SITETNC*) has little impact on the assessment factor in this instance.

- Although growth and removals have varied over the past half-century, the assessment factor would remain negative if calculated over different historical time periods.

## 3. Future Anticipated Baseline: Southeast Roundwood

The case studies presented in Table M-2 in this section begin with regional biomass consumption trajectories from the AEO Reference case and then require an additional 1 million short dry tons of roundwood feedstock consumption in the Southeast. This additional biomass requirement is phased in linearly, beginning with 250,000 short dry tons in the 2015 simulation period, reaching 1 million tons in 2030. The feedstock requirement was phased in over time under the conservative assumption that it could take time for a new facility or demand point to build up a steady supply source of one particular feedstock given regional market dynamics. The additional biomass requirement is then held constant for the remainder of the simulation horizon[1] and must be met by roundwood only.

Instead of calculating the net change in carbon pools on the landscape between two points in time, the future anticipated baseline approach calculates the cumulative net change between two alternative scenarios. The first row in Table M-2 labeled "Incremental Demand vs. AEO Ref" presents the estimated "marginal" *BAF* as discussed in Appendix L.[2] The "AEO Reference case vs. Zero Biomass" results in the tables below compare the AEO Reference case with a Zero Biomass scenario; thus, the values presented correspond to the "average" effects explained in Appendix L but include an *L* factor of 1.1. All other scenarios use the "average user" approach described in Appendix L, which takes the relative difference between the Incremental Demand and Zero Biomass scenarios. Like the reference point section, this case study also examines alternative scenarios as sensitivities evaluating regional aggregation, roundwood demand level, *BAF* equation term inclusion, land base, and temporal scale.

There are two primary differences in the presentation of biogenic assessment values in this appendix. The first difference is the use of the term "relative" to describe the fact that future anticipated baseline biogenic attribute term values are the difference between two alternative cases over a set time period. *Relative growth*, for example, is the difference between the Zero Biomass case and the alternate case in the sum of all tree carbon growth fluxes (in $CO_2$) for the 50-year period between 2010 and 2060. The second is that the sensitivities related to increased roundwood use are based on incremental demand levels for the feedstock rather than the increased removals evaluated in the reference point sensitivities.

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[1] The 2012 Annual Energy Outlook projections do not extend past 2030; thus, biomass consumption shock is held constant after this simulation period.

[2] Comparison of the 1 million ton increased feedstock consumption scenario to the AEO Reference baseline scenario can be interpreted as the *marginal* effect of a new source of consumption that is fueled by a single feedstock, relative to the AEO Reference anticipated baseline.

**Table M-2. Biogenic Assessment Factors Derived from a Future Anticipated Baseline Approach for the Southeast Roundwood Case Study.**

| Scenario | Time Scale | *Relative* Growth & Removals[3] | | *Relative* Carbon Fluxes | | | *Relative* Annual Total Carbon Flux & Biogenic Emissions | | Assessment Factor (BAF) (ratio of net biogenic emissions to potential gross emissions) |
|---|---|---|---|---|---|---|---|---|---|
| | | *Relative* Growth (million $tCO_2e$) | *Relative* Removals (million $tCO_2e$) | *Relative* Net Growth (GROW) (relative growth–relative removals) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Avoided Emissions (AVOIDEMIT) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Net Landscape Emissions (SITETNC) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Potential Gross Emissions (PGE) (sum of all relative carbon fluxes/50 years) (million $tCO_2e$/year) | *Relative* Net Biogenic Emissions (NBE) (emissions from harvest & use of feedstock per year) (million $tCO_2e$ /year) | |
| **Incremental Demand vs. AEO Reference** | 2015–2060 | -15 | -10 | -0.36 | 0.00 | -0.04 | 1.4 | -1 | -0.43 |
| **AEO Ref vs. Zero Biomass** | 2015–2060 | −22 | 27 | 0.01 | 0.00 | 0.00 | 12.1 | 0 | 0.01 |
| **Incremental Demand vs. Zero Biomass** | 2015–2060 | −37 | 17 | −0.03 | 0.00 | 0.00 | 13.4 | 0 | −0.04 |
| **South Central** | 2015–2060 | −75 | 18 | −0.19 | 0.00 | −0.05 | 6.2 | -1 | −0.26 |
| **Combined SE/SC** | 2015–2060 | −112 | 35 | −0.08 | 0.00 | −0.02 | 19.6 | -2 | −0.11 |
| **SE x2 Incremental Demand** | 2015–2060 | −51 | 94 | 0.06 | 0.00 | 0.03 | 14.8 | 1 | 0.10 |
| **SE x3 Incremental Demand** | 2015–2060 | −51 | 26 | −0.03 | 0.00 | 0.15 | 16.2 | 2 | 0.13 |
| **SE x5 Incremental Demand** | 2015–2060 | 33 | 27 | 0.06 | 0.00 | −0.02 | 18.9 | 1 | 0.05 |
| **SE x10 Incremental Demand** | 2015–2060 | −15 | 141 | 0.10 | 0.00 | 0.15 | 25.8 | 6 | 0.27 |
| **Without Onsite Emissions Change** | 2015–2060 | −37 | 17 | −0.03 | 0.00 | 0.00 | 13.4 | 0 | −0.03 |
| **Change Time Frame** | 2015–2030 | −7 | 56 | 0.22 | 0.00 | 0.06 | 4.5 | 1 | 0.31 |

While the *BAF* value as calculated from the equations is technically equal to the sum of *GROW, AVOIDEMIT*, and *SITETNC* in the absence of losses (*L*), the *BAFs* shown above may be slightly as it is assumed that *L= 1.1*. Furthermore, the values provided in the table are rounded to the nearest integer or hundredth particularly in the *AVOIDEMIT* term, which was projected to have a very small magnitude for most case studies (0.003 or less).

[3] Note that $CO_2$ accounting is atmospheric so sequestration is negative and emission is positive.

## 3.1. Key Insights from the Future Anticipated Baseline Application to Southeast Roundwood

- Because the future anticipated baseline compares alternative scenarios instead of comparing two points in time, many of the variables have different interpretations than they do under the reference point baseline.

    - Instead of representing forest growth between two points in time, *Growth* represents the relative difference in cumulative tree carbon growth fluxes in $CO_2$ between the two scenarios (in this case, the case study incremental demand scenario and the Zero Biomass utilization case). Positive values represent net emissions from the landscape for the case study scenario relative to the Zero Biomass utilization scenario. Negative values represent a net increase in sequestration on the landscape for the case study scenario.
    - *Removals* similarly represent the relative difference in cumulative tree carbon harvest in the two scenarios, with positive values representing an increase in harvest emissions (in $CO_2e$) for the case study relative to the Zero Biomass utilization case.
    - Instead of simply representing the ratio of growth to removals, the anticipated future baseline treats *GROW* as difference between growth and removals in order to represent relative net growth.

- The "Incremental Demand vs. AEO Ref" case represents the marginal effect of additional roundwood consumption relative to the AEO reference case. This anticipated additional demand leads to investments in new and replanted tree stands early in the simulation horizon, which increases carbon sequestration overall and reduces total emissions relative to the AEO Reference Case, resulting in a negative BAF (-0.43).
- The increased demand scenarios represent increases in incremental demand relative to the 1 million tons of incremental demand in the Southeast case study scenario (i.e., because the case study has a 1 million ton increase in demand relative to the AEO Reference, the "SE x2 Incremental Demand" case reflects a 2 million ton increase). These sensitivities have little impact on the assessment factor, but it should be noted that they are not directly equivalent to the increased removal scenarios in the reference point baseline table (which increases total biomass removed from the regional landscape proportionally to the increase biogenic feedstock demand).

    - In the initial time periods, the net emissions fluxes related to increased biogenic feedstock demand oscillate between positive and negative net emissions (note only cumulative values are included here, so this time path effect is not shown). These fluctuations occur as the market and related land use activities adjust to increased demand levels (e.g., large volumes of new plantings in the initial periods).

- However, as markets and related land uses adjust to new demand levels over time, equilibrium is reached and the assessment factor trends down to hover around or at 0.
- Thus, increasing the demand for Southeast roundwood for bioenergy use in this specific case study, and almost all of the related sensitivities, results in a small or 0 assessment factor.[4]

- In the sensitivity that shortens the analysis time frame from 50 years (2015–2060) to 20 years (2015–2030), the assessment factor for Southeast roundwood is 0.3, whereas the 50-year case study base case was 0.01. This shows that this baseline approach is quite sensitive to the analysis time frame chosen.

# 4. Retrospective Reference Point Baseline: Pacific Northwest Logging Residues

The values for this case study presented in Table M-3 represent the reference point-derived net biogenic $CO_2$ emissions from a hypothetical electricity facility with an EGU that uses logging residues from the Pacific Northwest region as a biogenic feedstock. This case study also examines an alternative scenario as sensitivity evaluating the equation term inclusion.

**Table M-3. Biogenic Assessment Factors Derived from a Reference Point Baseline for the Pacific Northwest Logging Residues Case Study.**

| Scenario | Time Scale | Growth (billion cu. ft.) | Removals (billion cu. ft.) | Growth to Removals Ratio (GROW) (removals–growth)/ removals) | Avoided Emissions (AVOIDEMIT) (avoided long term sequestration)/ ton removals | Net Landscape Emissions (SITETNC) (other site emissions)/ton removals | Potential Gross Emission (PGE) (million $tCO_2e$) | Assessment Factor (BAF) |
|---|---|---|---|---|---|---|---|---|
| **PNW** | 2006–2010 | N/A | N/A | 0 | −0.98 | 1.0 | 0.42 | 0.02 |
| **Without Net Landscape Emissions** | 2006–2010 | N/A | N/A | 0 | −0.98 | NA | 0.42 | −0.98 |

[4] Emissions from land management (*SITETNC*) have minimal influence on the assessment factor result. Evidence for this can be seen where the value of *SITETNC* is not included in the calculation; the assessment factor for Southeast Roundwood remains the same.

### 4.1. Key Insights from the Retrospective Reference Point Baseline Application to Pacific Northwest Logging Residues

- In this case study, it is assumed that logging residues have an alternative fate of decay on the forest floor, which would result primarily in emissions to the atmosphere. Because residues are not specifically cultivated but rather removed for bioenergy consumption, the assessment factor in this instance depends on the value of avoided emissions.
- Logging residues in this instance receive a slightly positive assessment factor because when logging residues are left on the landscape, a small portion of carbon is retained in soil carbon. Removal of the logging residues, therefore, has a small negative impact on soil carbon levels.
- Very little data are available on logging residues removal volumes and related impacts on landscape emissions. Thus, emissions fluxes associated with land management changes (*SITETNC*) are considered 0 for logging residues under this baseline approach. Therefore, omitting this term in calculating an assessment factor has no impact.

## 5. Future Anticipated Baseline: Pacific Northwest Logging Residues

The case study results from Table M-4 in this section begin with regional Biomass Consumption trajectories from the AEO Reference case and then require an additional 1 million short dry tons of logging residue feedstock consumption in the Pacific Northwest. The additional biomass requirement is phased in using the same method described in the roundwood case study section and likewise must be met by logging residues only.

The first row in Table M-2 labeled "Incremental Demand vs. AEO Ref" present "marginal" *BAF* as discussed in Appendix L. The remaining results are cumulative relative to Zero Biomass baseline. The incremental demand scenario includes 1 million tons more logging residues from the Pacific Northwest demanded by 2030 than AEO Reference case.

**Table M-4. Biogenic Assessment Factors Derived from a Future Anticipated Baseline Approach for the Pacific Northwest Logging Residues Case Study.**

| Scenario | Time Scale | *Relative* Growth & Removals | | *Relative* Carbon Fluxes | | | *Relative* Total Carbon Flux & Biogenic Emissions | | Assessment Factor (BAF) (ratio of net biogenic emissions to potential gross emissions) |
|---|---|---|---|---|---|---|---|---|---|
| | | *Relative* Growth (million $tCO_2e$) | *Relative* Removals (million $tCO_2e$) | *Relative* Net Growth (GROW) (relative growth–relative removals) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Avoided Emissions (AVOIDEMIT) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Net Landscape Emissions (SITETNC) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Potential Gross Emissions (PGE) (sum of all relative carbon fluxes/50 years) (million $tCO_2e$/year) | *Relative* Net Biogenic Emissions (NBE) (emissions from harvest & use of feedstock per year) (million $tCO_2e$ /year) | |
| **Incremental Demand vs. AEO Ref** | 2015-2060 | 15 | -1 | 0.21 | 0.00 | 0.09 | 1.4 | 0.4 | 0.33 |
| **AEO Ref vs. Zero Biomass** | 2015–2060 | −13 | 1 | −0.14 | 0.00 | −0.02 | 1.7 | -0.3 | −0.18 |
| **Incremental Demand vs. Zero Biomass** | 2015–2060 | −14 | 16 | 0.02 | 0.00 | 0.03 | 3.1 | 0.1 | 0.04 |
| **Without Net Landscape Emissions** | 2015–2060 | −14 | 16 | 0.02 | 0.00 | 0.00 | 3.1 | 0.0 | 0.02 |

While the *BAF* value as calculated from the equations is technically equal to the sum of *GROW, AVOIDEMIT,* and *SITETNC* in the absence of losses (*L*), the *BAFs* shown above may be slightly as it is assumed that *L= 1.1*. Furthermore, the values provided in the table are rounded to the nearest integer or hundredth particularly in the *AVOIDEMIT* term, which was projected to have a very small magnitude for most case studies (0.003 or less).

## 5.1. Key Insights from the Future Anticipate Baseline Application to Pacific Northwest Logging Residues

- In the "Incremental Demand vs. AEO Ref" case, there is an increase in forest harvests to respond to the additional demand for forest residues. This increase in harvest leads to a slight increase in net emissions and a resulting BAF of 0.33.
- In the "AEO Ref vs. Zero Biomass" case, a change in silviculture causes a response in *Growth* (probably for long-term stability of the market) but very little change in harvest because residues from existing harvest can be used to meet nearly all of the biogenic demand. This leads to negative values for both "relative net growth" and "relative net landscape emissions" terms and thus a negative assessment factor.
- In the incremental demand case, another million tons of biogenic feedstock are used (nearly doubling AEO Reference demand levels), which leads to both an increase in harvest and a stronger silvicultural response (although muted by the higher harvest). Given the aggregate harvest of approximately 1 billion tons $CO_2e$ over the 2015 through 2060 time

period, the harvest increase of 16 million tons is relatively minor. This leads to positive yet small "relative net growth," "relative net landscape emissions," and assessment factors.

# 6. Retrospective Reference Point Baseline: Corn Belt Corn Stover

The values for this case study presented in Table M-5 represent the reference point-derived net biogenic $CO_2$ emissions from a hypothetical electricity facility with an EGU that uses corn stover from the Corn Belt region as a biogenic feedstock. This case study also examines alternative scenarios as sensitivities evaluating $N_2O$ as well as equation term inclusion.

**Table M-5. Biogenic Assessment Factors Derived from a Reference Point Baseline for the Corn Belt Corn Stover Case Study.**

| Scenario | Time Scale | Growth (billion cu. ft.) | Removals (billion cu. ft.) | Growth to Removals Ratio (GROW) (removals–growth)/ removals) | Avoided Emissions (AVOIDEMIT) (avoided long-term sequestration)/ ton removals | Net Landscape Emissions (SITETNC) (other site emissions)/ton removals | Potential Gross Emission (PGE) (million $tCO_2e$) | Assessment Factor (*BAF*)[1] |
|---|---|---|---|---|---|---|---|---|
| **Base Case** | 2006–2010 | N/A | N/A | 0 | 0 | 0.0026 | 0.44 | 0.0029 |
| **With $N_2O$** | 2006–2010 | N/A | N/A | 0 | 0 | 0.0123 | 0.44 | 0.0135 |

## 6.1. Key Insights from the Retrospective Reference Point Baseline Application to Corn Belt Corn Stover

- In this case study, corn stover production for energy is not considered the motivation for crop production, and the "growth to removals ratio" is assumed to be 0. The assessment factor in this instance depend on the value of "avoided emissions," because the assumed alternate fate of these residues is to decompose or be burned onsite (results here are the former). Therefore, "avoided emissions" are equal to 0.
- When $N_2O$ is included in the "net landscape emissions" calculation, the assessment factor is larger than when $N_2O$ is not included (meaning that only 90% of biogenic $CO_2$ emissions out the stack are counterbalanced by feedstock growth). This suggests that there are increases in the nitrogen fertilizer application to replenish soil nutrients that were lost by removing corn stover that would have otherwise decomposed onsite.

# 7. Future Anticipated Baseline: Corn Belt Corn Stover

The first row in Table M-2 labeled "Incremental Demand vs. AEO Ref" present "marginal" *BAF* as discussed in Appendix L. Remaining sensitivity results are cumulative relative to Zero Biomass baseline. The incremental demand scenario includes 1 million tons more corn stover from the Corn Belt demanded by 2030 than the AEO Reference case.

**Table M-6. Biogenic Assessment Factors Derived from a Future Anticipated Baseline Approach for the Corn Belt Corn Stover Case Study.**

| Scenario | Time Scale | *Relative* Growth & Removals | | *Relative* Carbon Fluxes | | | *Relative* Total Carbon Flux & Biogenic Emissions | | Assessment Factor (BAF) (ratio of net biogenic emissions to potential gross emissions) |
|---|---|---|---|---|---|---|---|---|---|
| | | *Relative* Growth (million $tCO_2e$) | *Relative* Removals (million $tCO_2e$) | *Relative* Net Growth (GROW) (relative growth– relative removals) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Avoided Emissions (AVOIDEMIT) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Net Landscape Emissions (SITETNC) (million $tCO_2e$/ton biogenic feedstock use) | *Relative* Potential Gross Emissions (PGE) (sum of all relative carbon fluxes/50 years) (million $tCO_2e$/year) | *Relative* Net Biogenic Emissions (NBE) (emissions from harvest & use of feedstock per year) (million $tCO_2e$/year) | |
| **Incremental Demand vs. AEO Ref** | 2015-2060 | NA | NA | 0 | 0 | 0.08 | 1.4 | 0.0\ | 0,08 |
| **AEO Ref vs. Zero Biomass** | 2015–2060 | NA | NA | 0 | 0 | 0.27 | 0.8 | 0.2 | 0.27 |
| **Incremental Demand vs. Zero Biomass** | 2015–2060 | NA | NA | 0 | 0 | 0.15 | 2.2 | 0.3 | 0.17 |
| **With $N_2O$** | 2015–2060 | NA | NA | 0 | 0 | 0.15 | 2.2 | 0.3 | 0.17 |
| **Without Net Landscape Emissions** | 2015–2060 | NA | NA | 0 | 0 | 0.00 | 2.2 | 0.0 | 0.00 |

While the *BAF* value as calculated from the equations is technically equal to the sum of *GROW, AVOIDEMIT,* and *SITETNC* in the absence of losses (*L*), the *BAFs* shown above may be slightly as it is assumed that *L= 1.1*. Furthermore, the values provided in the table are rounded to the nearest integer or hundredth particularly in the *AVOIDEMIT* term, which was projected to have a very small magnitude for most case studies (0.003 or less).

## 7.1. Key Insights from the Future Anticipate Baseline Application to Corn Belt Corn Stover

In the “AEO Ref vs. Zero Biomass” scenario (with 1 million ton demand increase over the AEO Reference level), the cumulative assessment factor for corn stover is 0.17. This means that approximately 83% of additional biogenic feedstock consumption is replaced by carbon sequestration on the landscape.

- As “relative net growth” defaults to 0 for corn stover, the relatively large assessment factor is driven by “relative net landscape emissions.” This flux represents changes in agricultural and forestry land management in response to the long-term increase in the demand for corn stover biomass.
- The estimated *BAF* under the “Incremental Demand vs. AEO Ref” case is smaller than the “Incremental Demand vs. Zero Biomass” case. This implies that the marginal landscape emissions effect of increasing corn stover removals in isolation could be smaller than a total shift in biomass consumption in the Corn Belt (with multiple feedstocks used to meet the additional demand).
- Note that when the $N_2O$ flux is included, calculated “relative net landscape emissions” values increase slightly because of the additional corn stover demand, which increases corn production and nitrogen fertilizer use, thus increasing $N_2O$ emissions relative to the Zero Biomass baseline.
- When “relative net landscape emissions” is not included in the assessment factor, the resulting assessment factor is effectively 0, because the primary terms, “relative net growth” and “relative net landscape emissions,” are eliminated.

# Appendix N. Assessing Emissions from Waste-Derived Biogenic Feedstocks

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# 1. Introduction

This appendix describes various emissions pathways that result in biogenic $CO_2$ and $CH_4$ emissions from stationary sources that use waste-derived biogenic feedstocks, and illustrates how the framework could be adapted to derive assessment factors for these biogenic feedstocks. For the purposes of this appendix, the waste-derived biogenic feedstock can be defined as the portion of the biogenic waste material whose management results in point source emissions (i.e., stack emissions). For example, for MSW sent to a combustor, the biogenic feedstock is the entire biogenic fraction of the MSW sent to the combustor. For MSW sent to a landfill, the biogenic feedstock is the collected landfill gas—an amount representing less than the entire biogenic fraction of the landfilled MSW.

As discussed in this appendix, waste-derived biogenic feedstocks include the following:

- Landfill gas[1] generated through the decomposition of municipal solid waste (MSW) in a landfill;
- The biogenic fraction of MSW;
- Biogas generated from the decomposition of livestock waste,[2] biogenic MSW, and/or other food waste in an anaerobic digester;
- Livestock waste; and
- Biogas generated through the treatment of waste water, due to the anaerobic decomposition of biological materials.

The following emission pathways that result in biogenic $CO_2$ emissions from stationary sources using waste-derived biogenic feedstocks are evaluated:

- Combustion of landfill gas, through either a flare or combustion in an electric generating unit (EGU);
- Combustion of MSW;
- Combustion of biogas from an anaerobic digester used to manage livestock waste and/or food waste, through either a flare or combustion in an EGU;
- Combustion of livestock waste; and
- Combustion of biogas from an anaerobic digester used to manage wastewater and associated sludges.

This appendix is organized by the aforementioned emissions pathways. These selected pathways are not meant to represent an exhaustive list of all possible alternate fate pathways. Included in this appendix are illustrative methods for how the framework can be applied to waste-derived biogenic feedstocks used at stationary sources to assess net biogenic carbon-based contributions to the atmosphere using the example pathways described above. These illustrative methods are complemented with illustrations of biogenic assessment factor (*BAF*) values derived through application of the framework to the selected emission pathways.

In the context of stationary sources, greenhouse gas (GHG) emissions from waste-management options can be categorized into direct emissions and indirect emissions.[3] The illustrative framework applications in this appendix address point source biogenic $CO_2$ and $CH_4$ emissions from stationary sources using waste-derived feedstocks. Point source emissions of biogenic $CO_2$ occur as a result of combustion of landfill gas, biogas, MSW, or livestock waste. Combustion typically occurs

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[1] Landfill gas and biogas consists of approximately 50% methane ($CH_4$) and 50% $CO_2$, with small percentages of other gases, such as volatile organic compounds (VOCs).
[2] In this appendix, "livestock waste" refers to eliminated products (e.g., manure, litter, urine) resulting from the digestive process by farm animals (e.g., cattle, sheep, goat, swine, poultry, equine animals, etc.) and associated biogenic materials managed as waste materials (e.g., bedding materials and uneaten animal feed).
[3] Indirect emissions refer to emissions released directly to the atmosphere, rather than through a stack or vent. Indirect emissions include uncollected GHGs (e.g., biogas) that are released to the atmosphere and collected GHGs (e.g., biogas) that are subsequently leaked to the atmosphere.

in a flare or in an EGU. In applications where landfill gas and biogas are combusted, combustion results in the destruction of $CH_4$ and the emission of $CO_2$. However, since combustion efficiency is less than 100%, not all $CH_4$ is destroyed, such that some uncombusted $CH_4$ is released as a point source emission. $CH_4$ has a significantly higher global warming potential (GWP) than $CO_2$.[4] As a result, destruction of $CH_4$ that would have been released to the atmosphere as an indirect emission in the absence of combustion results in a reduction of $CO_2$-equivalent ($CO_2$e) contribution to the atmosphere.

Indirect emissions of $CH_4$ and $CO_2$ occur at landfills, wastewater treatment facilities, and in livestock settings including housing, conveyances, uncovered lagoons storing livestock waste and/or food waste, and land application areas. Indirect emissions of $CH_4$ and $CO_2$ also have the potential to occur via other waste management techniques, including anaerobic digesters. Note that some waste management strategies may result in both direct and indirect GHG emissions (e.g., landfills, wastewater treatment facilities, and anaerobic digesters). Table N-1 summarizes different GHG emissions pathways related to the management of waste.

**Table N-1. Waste Management GHG Emissions Pathways Considered.**

| Type of Waste | Waste Management Option | Biogenic Feedstock | Direct Emissions[1] | Indirect Emissions |
|---|---|---|---|---|
| **MSW** | Landfill | Landfill gas | $CH_4$ and $CO_2$ emissions from combustion of collected landfill gas (flare or EGU) | $CH_4$ and $CO_2$ emissions at the landfill cap, leaks in landfill gas header piping and wells, leachate collection sumps, and cracks or penetrations in the landfill surface or side slopes |
| **Food waste** | Aerobic digestion (composting) | Food waste | N/A | $CO_2$ emissions (oxidation from decomposition)[2] |
| **MSW** | MSW combustor | MSW | $CO_2$ emissions from combustion, typically in an EGU | $CH_4$ and $CO_2$ emissions from pretreatment handling practices |

[4] Methane is a potent GHG, with a 100-year global warming potential (GWP) of 21 (IPCC, 1996). It should be noted that in the IPCC Fourth Assessment Report, the 100-year GWP of $CH_4$ was revised to 25 (IPCC, 2007). To comply with international reporting standards under the UNFCCC, official emission estimates reported by the United States use the IPCC Second Assessment Report GWP values (IPCC 1996). The United States will transition to using the revised GWPs beginning in 2015. In this framework, the GWP of 25 is used for the central examples within each section. The GWPs of 21 and 28 are used in the sensitivity analyses for each section.

| Type of Waste | Waste Management Option | Biogenic Feedstock | Direct Emissions[1] | Indirect Emissions |
|---|---|---|---|---|
| **Livestock waste** | Housing, conveyances, storage in an open lagoon, pond, pit, or pile[3] | Biogas | N/A | $CH_4$ and $CO_2$ emissions from uncovered lagoon, pond, or pit |
| **Livestock waste and/or food waste** | Anaerobic digester | Biogas | $CO_2$ emissions from combustion of collected biogas (flare or EGU) | Potential for indirect $CH_4$ emissions from digester if not all $CH_4$ produced is captured; $CH_4$ emissions from digester effluent |
| **Livestock waste** | Aerobic digestion treated waste (e.g., handled as a solid or sprayed on a field) | Manure and litter | N/A | $CO_2$ emissions (oxidation from decomposition) |
| **Livestock waste** | Livestock waste combustor | Manure and litter | $CO_2$ emissions from combustion, often in an EGU; $CH_4$ emissions from incomplete combustion | $CH_4$ and $CO_2$ emissions from pretreatment handling practices |
| **Wastewater** | Aerobic wastewater treatment process | Wastewater | N/A | $CO_2$ and $CH_4$ emissions from uncovered treatment ponds ($CH_4$ emissions from instances where partial anaerobic conditions are present) |
| **Wastewater** | Anaerobic wastewater treatment process | Biogas | $CO_2$ emissions from combustion of collected biogas (flare or EGU) | Potential for indirect $CH_4$ emissions from digester if not all $CH_4$ produced is captured; $CH_4$ emissions from digester effluent |

[1] Point source emissions consist primarily of combustion emissions (i.e., $CO_2$) and secondarily of uncombusted $CH_4$ emissions via incomplete destruction of biogas during combustion (EPA, 2008b).
[2] If compost piles become anaerobic, $CH_4$ and $N_2O$ may also be generated and emitted.
[3]The term conveyances refers to indirect emissions from the piping when transferring waste to and from units. The term pile refers to poultry litter storage piles.

There are critical differences between the waste-derived biogenic feedstocks addressed in this appendix and the other forest- and agricultural-derived biogenic feedstocks addressed by the framework. The biologically based material in waste-derived feedstocks was removed from the land base for economic and production purposes outside of generating materials for the waste stream (e.g., for manufacture of consumer and industrial products, such as newspaper, food, and construction materials). Materials in the waste stream represents material that has been discarded, where final disposition of the material must be managed in some fashion (EPA, 2011b). As a result, if waste-derived feedstocks had not been processed or used by a stationary source, the material would have been managed through an alternative strategy with an alternative emissions pathway. Whatever the waste management strategy, it would result in biogenic $CO_2$ emissions and likely some amount of $CO_2e$ GHG emissions (e.g., $CH_4$ emissions as a result of anaerobic decomposition). Evaluating the carbon cycle effects of waste management at a stationary source involves a comparison of the biogenic $CO_2$ and $CH_4$ emissions at the stationary source against an alternative emissions pathway that would have resulted under an alternate management strategy.

Evaluating these alternate waste management GHG emissions pathways does not require an analysis of the carbon cycle effects that transpired during the growth and harvest of the primary biogenic materials on the landscape. As a result, many of the biogenic attributes related to the carbon cycle effects of the growth, harvest, and use of other biogenic feedstocks are not relevant for waste-derived biogenic feedstocks. In many cases, as demonstrated in this appendix, a number of the terms in the assessment factor equation drop out when evaluating emission pathways related to waste-derived biogenic feedstocks.

## 1.1. A Simplified Biogenic Assessment Factor Equation for Waste-Derived Biogenic Feedstocks

The *BAF* equation presented in the framework for the non-waste-derived feedstocks (i.e., forestry-derived, agriculture-derived) can be simplified for application to waste-derived biogenic feedstocks. This section provides the simplified general assessment factor equation which is then modified to calculate illustrative *BAF* values for waste-derived feedstocks under different waste management strategies, shown in later sections.

In the assessment factor equation presented in the main body of the framework, the Avoided Emissions (*AVOIDEMIT*) term accounts for the avoidance of estimated biogenic emissions that could have occurred on the feedstock landscape without biogenic feedstock removal (e.g., avoided decomposition or burning), or per an alternative management strategy. The *AVOIDEMIT* term can be adjusted by the emission pathways specific to the type of waste-derived feedstock and waste management strategy. For example, for certain biogas waste feedstocks, *AVOIDEMIT* can be adjusted by the biogas collection efficiency, biogas combustion efficiency, or other factors affecting emission pathways. As a result, some of the terms in the equation as presented in Equation 2 in the main document of the framework are not relevant to the waste-derived feedstocks discussed in this appendix as illustrated below:

$$\boldsymbol{BAF} = \frac{\boldsymbol{NBE}}{\boldsymbol{PGE}} = \frac{(\boldsymbol{PGE})(\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P})}{\boldsymbol{PGE}}$$

The *BAF* is then simplified to (Equation 3 in the main document):

$$\boldsymbol{BAF} = (\boldsymbol{GROW} + \boldsymbol{AVOIDEMIT} + \boldsymbol{SITETNC} + \boldsymbol{LEAK})(\boldsymbol{L})(\boldsymbol{P})$$

When only the waste-derived biogenic feedstocks are considered, the following terms can be dropped: Net Growth on the Production Landscape (*GROW*), Total Net Change in Production Site Non-feedstock Carbon Pools (*SITETNC*), Leakage Associated with Feedstock Production (*LEAK*), the Feedstock Carbon Losses during Storage, Transport and Processing ($L$), and the Feedstock Carbon Embodied in Products ($P$).

As a result, the full assessment framework equation as applied to biogenic $CO_2$ emissions from waste-derived feedstocks can be simplified to Equation N.1.

$$\boldsymbol{BAF = AVOIDEMIT} \qquad \textbf{(EQ. N.1)}$$

*AVOIDEMIT* represents the avoided biogenic emissions that could have occurred per an alternative management strategy instead of the waste-derived feedstock's use in bioenergy production, relative to biogenic feedstock consumption. As discussed in the main document, negative, positive and zero *BAF*s (which is the same as *AVOIDEMIT* in this appendix), have different implications. A positive value implies that use of the feedstock for bioenergy production contributes more emissions to the atmosphere than would have occurred under the alternative management strategy. A zero value implies that both practices are equivalent in terms of how much emissions they contribute to the atmosphere. A negative value implies that using the feedstock for bioenergy production contributes less emissions to the atmosphere than the alternative management practice. In practice, as applied here, the *AVOIDEMIT* term is a proportion expressed as $tCO_2e$ avoided (i.e., the emissions reduced, in $CO_2e$, resulting from an alternate waste management strategy to the combustion method) per $tCO_2e$ emitted using the combustion method (i.e., the emissions, in $CO_2e$, resulting from the combustion waste management strategy). The *AVOIDEMIT* term is applied because the waste management strategy (e.g., collection and combustion of landfill gas) typically results in avoided $CO_2e$ emissions that would have occurred in the absence of that management strategy (e.g., had the landfill gas not been collected and combusted, it may have been released as an indirect emission).[5] The *AVOIDEMIT* term, as applied to the waste-derived biogenic feedstocks described in this appendix, can be conceptually expressed as Equation N.2:

$$\boldsymbol{AVOIDEMIT = 1 - \frac{CO_2e\ emissions\ from\ treatment\ alternative\ to\ combustion}{CO_2e\ emissions\ from\ combustion\ treatment}} \qquad \textbf{(EQ.N.2)}$$

The *AVOIDEMIT* term is calculated for the specific waste-derived feedstock being managed relative to a specific, alternative practice. The following sections of this appendix go into detailed discussion about illustrative methodologies for the calculation of a *BAF* for waste-derived biogenic feedstocks.

Table N-2 presents a summary of illustrative *BAF* values calculated from example inputs using the methodology presented in subsequent sections of this appendix for the waste-derived biogenic feedstocks. These illustrative *BAF* values are dependent on the assumptions applied to the actual waste feedstock and to the alternate fate of the waste feedstock.

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[5] This treatment is conceptually comparable to how the *AVOIDEMIT* term is applied to biogenic feedstocks that are harvested from the landscape.

**Table N-2. Illustrative Example *BAF* Values Associated with the Treatment Methods of the Waste Feedstocks Discussed in this Appendix.**

| Waste Treatment Option | Biogenic Feedstock | Actual Treatment Fate | Alternate Treatment Fate | Illustrative *BAF* | Section Number |
|---|---|---|---|---|---|
| **MSW, landfill** | Landfill gas | Treatment with flares (higher DE) | No gas treatment | −1.48 | 2.2.1 |
| | Landfill gas | Treatment with an EGU (lower DE) | No gas treatment | −1.38 | 2.2.2 |
| | Landfill gas | Treatment with an EGU installed partway through the year | No gas treatment | −0.64 | 2.2.3 |
| **MSW, combustion** | Biogenic fraction of MSW | Incineration | Landfill gas treatment with flaring or EGU | −0.02 | 3.2.1 |
| | | | Landfill with no gas treatment | −1.52 | 3.2.2 |
| **Livestock waste, anaerobic digester** | Manure, litter, and biogas | Treatment with flares, when anaerobic digester measurement data are available | Uncovered anaerobic lagoon | -2.56 | 4.2.1 |
| | | Treatment with flares, prior to the installation of an anaerobic digester | Uncovered anaerobic lagoon | −1.95 | 4.2.2 |
| **Livestock waste, combustion** | Manure, litter, and biogas | Incineration | 1-year litter storage prior to field spreading | 0.06 | 5.2.1 |
| | | | Uncovered anaerobic lagoon | −2.67 | 5.2.2 |
| **Wastewater and wastewater sludge, anaerobic digester** | Biogas | Treatment with flares | Lagoon (with aerobic and anaerobic zones) | −0.88 | 6.2 |

DE = destruction efficiency

Note: Assumptions and scenario details for each waste treatment option and the associated *BAF* calculations are explained in the text. The parameterization of variables used in the calculations presented here are illustrative only; parameter values used in these calculations may not apply to all applications of the framework vis-à-vis the use of waste-derived biogenic feedstocks used at stationary sources.

## 1.2. Biogenic Municipal Solid Waste Management

Biogenic MSW refers to the biogenic (organic) fraction of MSW. In 2012, approximately 250.9 million tons of MSW were generated in the United States (EPA, 2014a). Biogenic materials were the largest component of MSW before recycling (see Table N-3). Of the total MSW generated, 135 million tons (53.8%) went to landfills, 86.6 million tons (34.5%) were recovered (e.g., recycled or composted), and 29.3 million tons (11.7%) were combusted with energy recovery (this includes biogenic as well as fossil fuel-based materials, such as plastics). The proportions of waste recycled, composted, incinerated, or landfilled differ regionally due to multiple factors, including local economics, regulatory differences at the state and local levels, public perceptions, and infrastructure requirements (Bogner et al., 2007; EPA, 2010c). However, there is a lack of literature describing the degree to which composition of MSW can vary from region to region. Therefore, for the purposes of the framework, we will use a national average composition based on EPA data through 2012 (EPA, 2014a).

Although composition of MSW may vary from region to region, this mainly contributes to potential generation amount of $CO_2$ and $CH_4$ in a given landfill, whereas the goal of the framework methodology for waste-derived feedstocks is ultimately concerned with how the $CO_2$ and $CH_4$ from MSW is treated used in one activity versus another. From this perspective, $CO_2$ and $CH_4$ from MSW can be treated similarly across the U.S.

**Table N-3. Percent of MSW Generated and Recovered by MSW Class in 2012 (EPA, 2014a).**

| MSW Class | Biogenic? | Percent Generated | Percent Recovered (as percent of generation) |
|---|---|---|---|
| **Paper and paperboard** | Yes | 27.4 | 64.6 |
| **Yard trimmings** | Yes | 13.5 | 57.7 |
| **Food scraps** | Yes | 14.5 | 4.8 |
| **Plastics** | No | 12.7 | 8.8 |
| **Metals** | No | 8.9 | 34.0 |
| **Rubber and leather** | Partial | 3.0 | 17.9 |
| **Textiles** | Partial | 5.7 | 15.7 |
| **Wood** | Yes | 6.3 | 15.2 |
| **Glass** | No | 4.6 | 27.7 |
| **Miscellaneous** | Uncertain | 1.6 | negligible |

In the United States, MSW typically has one of four fates (Bogner et al., 2007):

- Landfilling;
- Combustion;
- Processing in an anaerobic digester; or
- Composting.

Sections 2 and 3 of this appendix discuss in detail the GHG emissions pathways for MSW landfills and MSW combustion, respectively. Food waste can be treated through anaerobic digestion systems and is relatively common at wastewater treatment plants. The excess capacity of the digestion system can be supplemented from food waste (e.g., EMBUD plant). Waste treatment through anaerobic digestion is discussed for both livestock waste management and wastewater treatment. Composting is not a stationary source activity, and is therefore not discussed in this appendix.

# 2. Biogenic MSW Disposal in MSW Landfills and Associated GHG Emissions Pathways

GHG emissions pathways at MSW landfills result in $CH_4$ and $CO_2$ emissions. In general, landfill-related $CH_4$ and $CO_2$ emissions are of biogenic origin and primarily result from the decomposition, under anaerobic or aerobic conditions, of organic matter such as food, yard wastes, and paper. The decomposition of organic matter in a landfill occurs through a series of microbial reactions, primarily under anaerobic conditions (Bogner, 1992). Methane and $CO_2$ are produced through the action of methanogenic bacteria as they consume the organic matter and convert it into stabilized organic materials and biogas. By volume, the composition of landfill gas ranges from 45% to 55% $CH_4$ and $CO_2$, but is generally assumed to be half $CH_4$ and half $CO_2$ (EPA, 2010a). Landfill gas also contains small amounts of nitrogen, oxygen, and hydrogen; less than 1% non-methane organic compounds (NMOCs); and trace amounts of inorganic compounds (EPA, 2014b). Landfill gas will continue to generate for many years, even decades, after an initial mass of waste is placed in a landfill due to the slow degradation process and compaction of the waste.

There are two general pathways for $CH_4$ and $CO_2$ emissions from landfills—indirect emissions and direct emissions (i.e., point source combustion emissions). These two emissions pathways are affected by the presence of an active gas collection and control system (i.e., flare or EGU). The remainder of this section first discusses the emission pathways as they relate to controlled and uncontrolled landfills, and then goes into further detail about key parameters affecting the amount and type of emissions from these pathways.

- <u>*Uncontrolled Landfills*</u>—An "uncontrolled" landfill refers to a landfill that has no active system, such as a gas collection and control system, in place to minimize indirect landfill gas emissions to the atmosphere. Though the landfill biogas is not collected from uncontrolled landfills, the biogas may be managed through the use of a topsoil cover to passively treat the uncollected biogas via $CH_4$ oxidation. Indirect emissions are the primary emissions pathway from uncontrolled landfills (see Figure N-1).
  - *Direct emissions*: None.
  - *Indirect emissions:* The primary GHG emissions pathway at uncontrolled landfills is indirect emissions of $CH_4$ and $CO_2$ through the landfill soil cover. A fraction of the $CH_4$ in the biogas (ranging from 10% to 35% (IPCC, 2006; EPA, 2013c; SWICS, 2009) will be oxidized by bacteria in the cover soil as the gas migrates vertically through the landfill cover soils.
- <u>*Controlled Landfills*</u>—A "controlled" landfill refers to a landfill that has an active landfill gas collection and control system in place. The collection system consists of network of pipes and collection wells strategically placed throughout the disposal areas to collect and transport the biogas to a central control system. A control system typically involves a combustion device such as a flare, turbine, or boiler for combustion of the collected landfill gas. Controlled landfills also include a topsoil cover to passively treat the remaining landfill gas that is not collected via $CH_4$ oxidation. In the United States, there are approximately 594 operational landfill gas-to-energy projects, at which landfill gas is used as fuel for generation of electricity or process heat in industrial applications (EPA, 2013a).

Approximately 25% of the roughly 2,400 currently operating or recently closed MSW landfills in the United States include landfill gas collection and control systems (flaring or energy generation) (EPA, 2013a). An estimated 540 additional, existing domestic MSW landfills have the potential to capture landfill gas for energy use (EPA, 2013a). The primary GHG emissions pathway at controlled landfills is direct emissions of $CO_2$ (see Figure N-2).

- *Direct emissions:* The primary GHG emissions pathway at controlled landfills is point source emissions of $CO_2$. The $CH_4$ in the landfill gas that is collected and combusted will be converted to $CO_2$; the $CO_2$ in the collected landfill gas will be directly emitted as $CO_2$; and the $CH_4$ in the collected landfill gas that is not combusted will be directly emitted as $CH_4$.
- *Indirect emissions:* Both $CH_4$ and $CO_2$ will be emitted through the landfill soil cover. A fraction of the $CH_4$ in the biogas (ranging from 10% to 35% (IPCC, 2006; EPA, 2013c; SWICS, 2009) will be oxidized by bacteria in the cover soil as the gas migrates vertically through the landfill cover soils.

When organic materials are landfilled, a portion of the carbon in the materials will not readily degrade due to several factors, including environmental conditions (e.g., moisture, pH, temperature), and the creation of anaerobic environments through waste disposal and compaction. When the environment in which wastes are placed becomes anaerobic, the organisms that normally break down the waste cannot survive to decompose a portion of the organic materials, thus this portion will remain in the landfill. This process is referred to as carbon storage because this carbon is permanently removed from the global carbon cycle.

Cellulose and hemicellulose are the major biodegradable components of MSW (Barlaz, 1998; Barlaz, 2006). Additionally, lignin will not degrade at all when placed in a modern landfill (Barlaz, 1998). On a dry weight basis, MSW contains between 30% and 50% cellulose, 7% to 12% hemicellulose, and 15% to 28% lignin (Hilger and Barlaz, 2001). The amount of cellulose and hemicellulose in the organic materials that will degrade depends on the type of material. Laboratory bench scale research has been conducted to quantify carbon storage factors for several materials of the MSW stream (Barlaz, 1998; ICF, 2008), including yard waste, food, and various paper products. These carbon storage factors represent the mass of carbon stored in a landfill per initial mass of the component and range from 0.05 to 0.47 kg of carbon sequestered per dry kg of waste component (Barlaz, 1998; ICF, 2008). The 2006 IPCC Guidelines (IPCC, 2006a) recommends a default factor of 0.5 for the fraction of degradable organic carbon that is anaerobically decomposed in the landfill, suggesting that 50% of the biogenic carbon placed in a landfill becomes stored carbon.

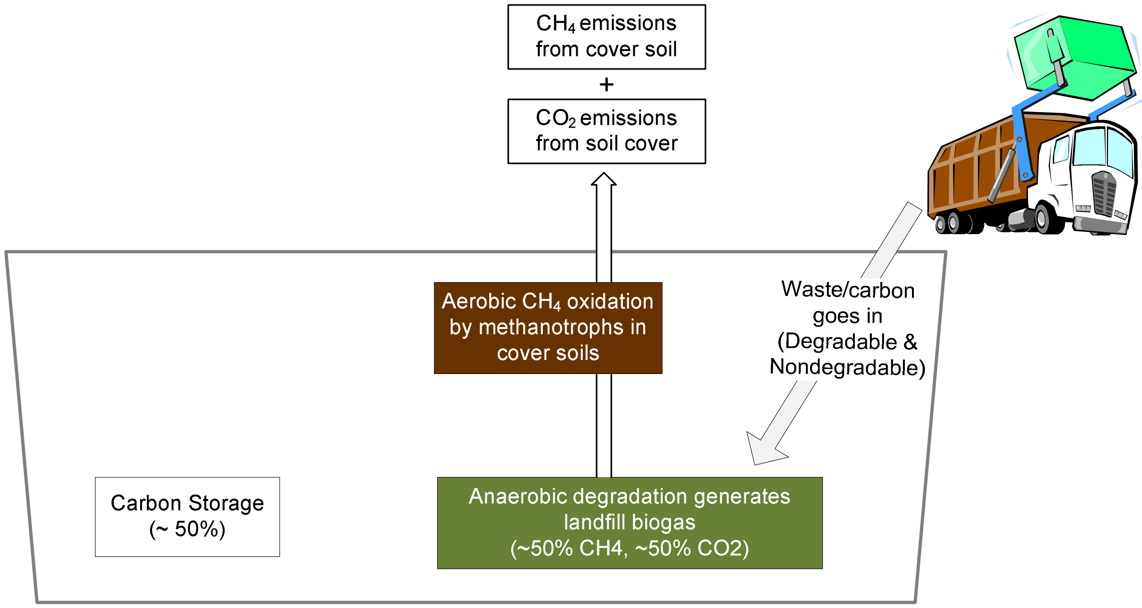


**Figure N-1. Carbon Balance for an Uncontrolled Landfill.**

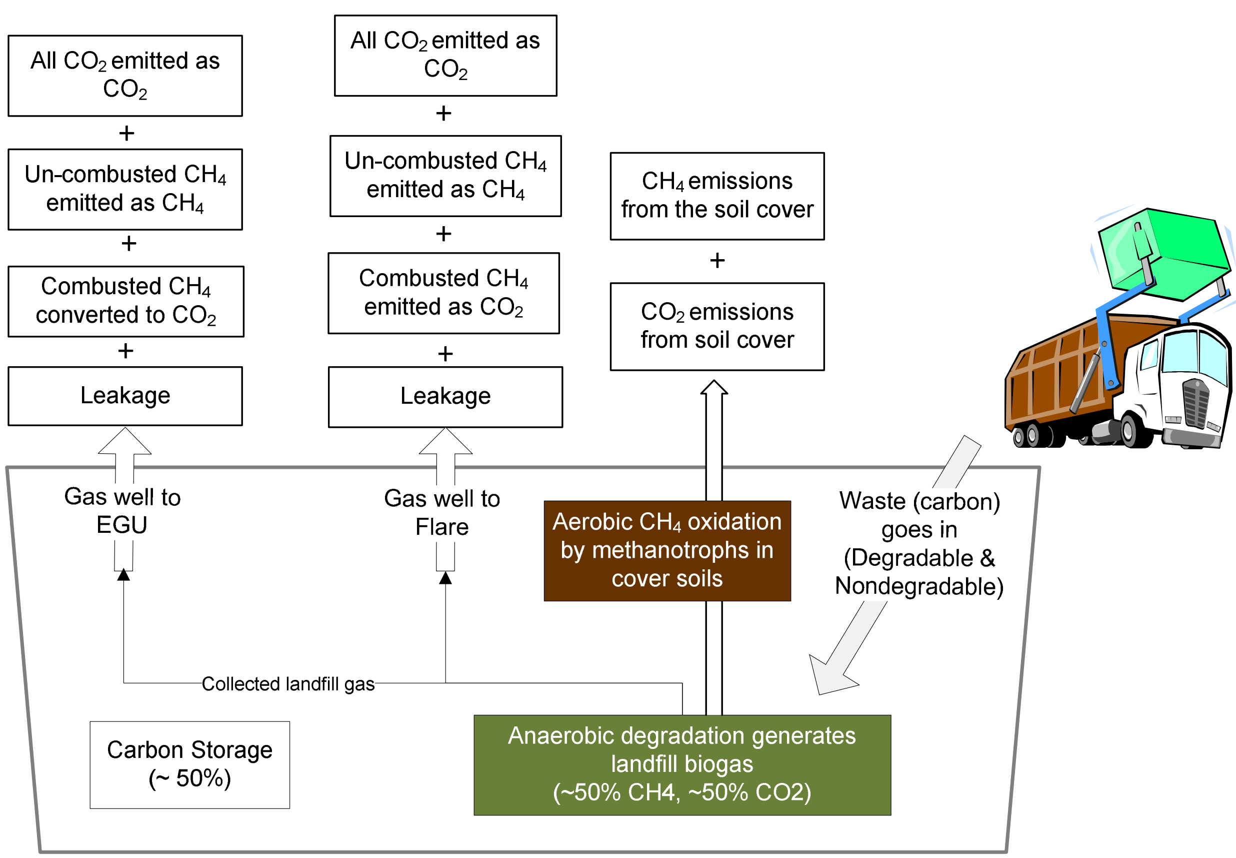


**Figure N-2. Carbon Balance for a Controlled Landfill.**

## 2.1. Indirect Emissions from MSW Landfills

The first pathway for GHG emissions from landfills assessed in this framework is indirect emissions ($CH_4$ and $CO_2$) released directly from the landfill cover to the atmosphere. The amount and rate of total $CH_4$ generation in landfills, as well as the amount of indirect emissions of $CH_4$ and $CO_2$, depends upon the quantity and composition of the landfilled material, as well as the landfill design and surrounding environmental conditions. If not collected and combusted, a portion of the $CH_4$ generated in a landfill oxidizes to $CO_2$ as it travels through the top layer of the landfill cover; the remaining unoxidized portion of the landfill gas is emitted through the landfill cover. This process results in indirect emissions of both $CH_4$ and $CO_2$ through the landfill cover.

Methane oxidation efficiency is directly affected by the thickness, physical properties, moisture content, and temperature of landfill cover soils. The rate of $CH_4$ oxidation in landfill cover soils is linear to a point, after which the methanotrophs in the cover soils reach an upper limit in their ability to oxidize the $CH_4$ and the remaining $CH_4$ passes through the cover soil without being oxidized (Chanton et al., 2011b). Methane oxidation efficiency can vary substantially between and within landfills. Cover soil properties (i.e., temperature and soil moisture) van vary as a function of climate, such that the efficiency of $CH_4$ oxidation may vary regionally and seasonally (Spokas and Bogner, 2011). In hot, arid climates, $CH_4$ oxidation in landfill cover soils can be limited, resulting in higher indirect $CH_4$ emissions than in cooler, wetter climates. In hot, arid climates soil moisture is limited for much of the year, thus reducing $CH_4$ oxidation rates, and seasonally high soil temperatures prevent methanotrophic activity (Chanton et al., 2011a, Spokas and Bogner, 2011).

The IPCC (2006a) and EPA (2009a) default value for the $CH_4$ oxidation fraction in cover soils of modern, managed landfills such as those found in the United States is 10% of generated $CH_4$. EPA considers 10% to be a conservative (lower end) default oxidation fraction. Some studies point to an average $CH_4$ oxidation rate of approximately 40% (ranging from negligible to 100%) of the total $CH_4$ arriving at the base of the landfill cover soils (Bogner et al., 2007; Chanton et al., 2009; Chanton et al., 2011a; Spokas and Bogner, 2011). Because field and laboratory studies have shown large variations in oxidation rates, particularly for landfills with active gas collection and control systems, EPA expanded the default oxidation fraction value to include those based on the calculated $CH_4$ flux[6] rate in grams per square meter per day ($g/m^2/day$) to the bottom of a landfill's cover soil prior to any oxidation (EPA, 2013c, 40 CFR § 98).

- For high rates of $CH_4$ flux (greater than 70 $g/m^2/day$) the default oxidation fraction is 10%;
- For moderate rates of $CH_4$ flux (10 to 70 $g/m^2/day$) the default oxidation fraction is 25%;
- For low rates of $CH_4$ flux (less than 10 $g/m^2/day$) the default oxidation fraction is 35%

[6] The methane flux rate is referred to as the continuous flow of methane from an area within a landfill where methane is produced to the atmosphere over a specified period of time.

## 2.2. Direct Emissions from MSW Landfills

The second pathway for GHG emissions from landfills is direct, point source emissions of $CO_2$ and, depending on the destruction efficiency (DE) of the combustion device, $CH_4$ emissions from landfill gas collection and combustion through flaring or use as a fuel in an EGU (often referred to as landfill gas-to-energy projects).[7]

Landfill gas collection systems vary in landfill gas collection efficiency (CE). Recovery ranges from 35% to 90% of the gas generated in a particular landfill cell, depending on the placement of the piping network and collection wells (Spokas et al., 2006; EPA, 2012a). The default collection efficiency recommended by EPA is 75% (the average from a range of 50% to 95%), meaning that 75% of the landfill gas generated is collected and routed to a control device (EPA, 2008a; EPA, 2010b; EPA, 2013c). However, actual collection efficiencies may vary substantially, and due to the cost of determining the amount of landfill gas generated in a landfill, are not cost-effective and therefore difficult to quantify. Very few published studies documenting measured CEs exist, and of those, the results are highly variable and appear to be correlated with the type of landfill cover system. For example, Spokas et al. (2006) conducted field studies of the methane mass balance at three landfills in France and quantified collection efficiencies ranging from 54% to 100%, depending on cover type and presence of a gas collection system.

Collected landfill gas may be combusted using a flare, as a fuel for an EGU, or directly in boilers and other applications. Landfill gas may be purified to create compressed natural gas or liquefied natural gas, or for injection into natural gas pipelines. Of the total estimated $CH_4$ generated at MSW and industrial landfills in 2012, 30.3% was flared, 34.5% was used to generate electricity, and 3.5% was oxidized at the landfill cap (EPA, 2014b).

The combustion process destroys, or oxidizes, the $CH_4$ in the landfill gas to $CO_2$, resulting in $CO_2$ emissions.[8] A portion of the collected gas will not be combusted due to inefficiencies in the combustion device, thus this fraction of $CH_4$ will be emitted to the atmosphere as $CH_4$. Destruction efficiencies for $CH_4$ in landfill gas range from 90% to 99.9% (IPCC, 2006; SWICS, 2009; New Zealand Ministry for the Environment, 2010; EPA, 2013c). Additionally, the $CO_2$ in the collected gas will not be combusted and will be directly emitted as $CO_2$.

MSW landfills with a design capacity of 2.5 million megagrams (Mg) and 2.5 million cubic meters of waste are required to calculate their annual emissions of non-methane organic compounds (NMOC). Landfills that emit 50 Mg or more of NMOC per year are required by EPA regulations to install a landfill gas capture and control system in order to control NMOC emissions (EPA, 1996). To

---

[7] Typically, both GHG emissions pathways will be present at a landfill, as landfills with landfill gas collection and destruction systems generally do not capture all $CH_4$ generated in the landfill. Uncaptured $CH_4$ will either be oxidized via methanotrophic activity in the cover soils, or will be released directly to the atmosphere as indirect emissions.

[8] Note that as the combustion process is never complete, some $CH_4$ in landfill gas is not destroyed and therefore stack gas emissions contain a small percentage of $CH_4$.

comply with EPA regulations, landfill operators must, at a minimum, collect and combust their biogas. A co-benefit of NMOC emissions control is the destruction of $CH_4$ present in the landfill gas.

Both uncontrolled and controlled landfills include liners and leachate collection systems to prevent pollutants from migrating beyond the landfill, which can result in ground and/or surface water pollution.[9] Both uncontrolled and controlled landfills also store carbon. A portion of the carbon of the landfilled biomass materials will not decompose due to the anaerobic environment created through modern landfilling. The carbon that does not decompose is therefore removed from the global carbon cycle and stored in the landfill. The fraction of the amount of carbon stored in a landfill is typically assumed to be approximately 50%.

Once a landfill has reached its design capacity, it is closed.[10] A final cover is installed and the site owner is required to monitor and maintain a closed landfill throughout a post-closure monitoring period (EPA, 1996). Post-closure monitoring includes leachate collection and treatment, groundwater monitoring, inspection of the final cover and maintenance as required, and monitoring to ensure that $CH_4$ is not migrating off-site. Collection and combustion of landfill gas may continue throughout this period. EPA regulations (40 CFR § 258.61) specify a 30-year post-closure monitoring period unless this period is extended by a regulatory agency on a site-specific basis (EPA, 1996).

## 2.3. Method for Calculating an Illustrative *BAF* Value for Biogenic Emissions from MSW Landfilling

The assessment factor equation can be applied to direct biogenic emissions resulting from the collection and combustion of landfill gas. The biogenic feedstock from an MSW landfill is the biogas generated from MSW decomposition and the biogas collected from the landfill. This section provides a method for calculating an illustrative *BAF* value to be applied to direct biogenic emissions from MSW landfills. The *BAF* methodology for MSW landfilling neither includes benefits from off-setting fossil fuels through landfill gas-to-energy projects, nor any carbon storage.

In Equation N.2, the *AVOIDEMIT* term is used to represent the net GHG emissions reductions achieved through capture and combustion of landfill gas, compared to an alternate, Reference case emissions pathway of indirect $CH_4$ and $CO_2$ emissions through the landfill cover (had the landfill gas not been collected and combusted). In other words, biogenic emissions from a controlled landfill are being compared to the biogenic emissions from an uncontrolled landfill.

In practice, as applied here, the *AVOIDEMIT* term is expressed as 1 minus the ratio of metric tons of $CO_2e$ ($tCO_2e$) avoided (i.e., the point and indirect emissions, in $CO_2e$, of the collected biogas had that biogas not been collected and combusted, after accounting for indirect emissions of $CO_2$ and $CH_4$, and the $CH_4$ oxidation that would have occurred in the landfill cover soil) per $tCO_2e$ removed via

---

[9] Synthetic liners and compacted clay soil typically line the sides and bottom of a landfill to protect groundwater and the underlying soil from leachate releases. Leachate collection and removal systems sit on top of the liners to remove leachate from the landfill for collection and disposal.

[10] A closed landfill as referred to in this context means a landfill that no longer accepts waste for disposal.

combustion (i.e., the emissions, in $CO_2$e, of the collected and combusted biogas, after accounting for the CE of the gas collection system and the DE of the biogas destruction device). For the biogas feedstock that is generated and collected from landfills, the *AVOIDEMIT* term can be conceptually expressed by 1 minus a simplified ratio of $CO_2$e emissions of the treatment fates:

$$\boldsymbol{AVOIDEMIT} = \mathbf{1} - \frac{\mathbf{CO_2e\ emissions\ from\ treatment\ alternative\ to\ combustion}}{\mathbf{CO_2e\ emissions\ from\ combustion\ treatment}} \quad \textbf{(EQ.N.2)}$$

Note that the same amount of biogas and constituents of the biogas are considered for both the actual and alternate treatment fates.

### 2.3.1. Boundaries and Assumptions for MSW Landfilling Methodology

The methodology presented in this appendix for treatment of MSW through landfilling does not consider offsets from electricity generation, carbon storage, or losses from the gas collection system. Assumptions regarding the operation of the gas collection system are also made. The rationale for the boundary considered within the scope and major assumptions made are provided as follows:

- Offsets—Landfill gas-to-energy projects reduce fossil fuel usage whereas flaring landfill gas does not. However, because this is not a lifecycle analysis, the effects of reduced fossil fuel usage is not included in the calculations presented here. The EPA's Landfill Gas Energy Benefits Calculator can be used to estimate direct, avoided, and total greenhouse gas reductions, as well as environmental and energy benefits, for the current year of a landfill gas energy project if desired (EPA, 2012b).
- Carbon storage—Carbon storage refers to the fraction of carbon remaining in the biogenic materials after accounting for the carbon exiting the system as landfill gas or that is dissolved in the leachate. The amount of carbon storage will vary with environmental conditions in the landfill, but can be generally thought to be about half of the carbon in each biomass material that remains in a landfill. Carbon storage is not considered in the treatment of waste-derived biogenic feedstocks by MSW landfilling in this framework because the amount of carbon storage in a given landfill will theoretically be equivalent despite the treatment fate of the landfill biogas.
- Losses—Indirect emissions from equipment leaks (e.g., valves, connectors, and open-ended lines) on or associated with a wellhead, or in the delivery infrastructure from the biogas collection system to the biogas destruction device are possible. However, in the context of landfill gas collection and control, losses are expected to be insignificant, especially for instances where the biogas destruction device is co-located at a landfill.
- Operation of the gas collection system—One important assumption to note is that the methodology for MSW landfilling assumes the landfill gas collection system is operating continuously. It is possible to perform the calculations with a gas collection system that is not continuously operated by applying an additional factor to account for the fraction that the recovery system was operating (fRec) to the equations used to calculate the $CO_2$ and $CH_4$ emissions from landfills with gas collection and control.

### 2.3.2. Explanation of MSW Landfilling Methodology

Both the numerator and denominator of the *AVOIDEMIT* equation can be calculated using Equation N.3. This equation considers the following emissions pathways from an MSW landfill with or without gas collection and control the amount of

- Indirect $CH_4$ emissions from the landfill surface;
- Indirect $CO_2$ emissions from the landfill surface;
- Direct $CH_4$ emissions from the $CH_4$ in the collected landfill gas that is not combusted (as a result of a combustion efficiency less than 100%);
- Direct $CH_4$ emissions in the collected landfill gas that is combusted and converted to $CO_2$; and
- Direct $CO_2$ emissions in the collected biogas that is emitted as $CO_2$.

$$\boldsymbol{CO_2e\ emissions\ from\ MSW\ landfilling}$$

$$= \boldsymbol{GWP_{CH_4}(CH_4R - CH_4D + CH_4U) + \left(CH_4D \times \frac{44}{16}\right) + CO_2R + CO_2U} \qquad \textbf{(EQ. N.3)}$$

Where:

| | |
|---|---|
| $CO_2$e emissions | = metric tons $CO_2$e emissions from MSW landfilling (MT/year). |
| $GWP_{CH_4}$ | = 100-year GWP of $CH_4$, 25 (IPCC, 2007). |
| $CH_4R$ | = the amount of $CH_4$ recovered and sent to the landfill gas destruction device (Equation N.4). |
| $CH_4D$ | = amount of $CH_4$ destroyed via combustion (Equation N.5). |
| $CH_4U$ | = amount of uncollected $CH_4$ emitted through landfill cover surface; separate calculations for landfills with gas collection and landfills without gas collection (Equation N.6 or Equation N.7, depending on the presence of a gas collection system). |
| $CO_2R$ | = the amount of $CO_2$ recovered, sent to the landfill gas destruction device, and emitted to the atmosphere (Equation N.8). |
| $CO_2U$ | = amount of uncollected $CO_2$ emitted through landfill cover surface (Equation N.9 or Equation N.10, depending on the presence of a gas collection system). |
| 44/16 | = molecular weight ratio of $CO_2$ to $CH_4$. |

Equation N.3 can be grouped and explained in three major parts:

- The first part, $GWP_{CH_4}(CH_4R - CH_4D + CH_4U)$, accounts for the amount of $CH_4$ that is collected, but not combusted ($CH_4R$ – $CH_4D$) plus the amount of $CH_4$ in the generated landfill gas that is not collected and emitted as indirect emissions through the landfill cover surface as $CH_4$. Because the terms in Equation N.3 need to be in units of $tCO_2e$, these quantities must be adjusted by the 100-year GWP for $CH_4$.

- The second part, ($CH_4D \times 44/16$), accounts for the quantity of $CH_4$ that is collected and oxidized to $CO_2$ during combustion. The amount of $CH_4$ destroyed needs to be adjusted by the 44/16 conversion factor because the gas is being converted to $CO_2$.
- The third part, $CO_2R + CO_2D + CO_2U$, accounts for all of the $CO_2$ emissions (direct and indirect).
  - $CO_2R$ is the amount of $CO_2$ in the biogas that is collected and sent to the destruction device; this quantity will be directly emitted as $CO_2$;
  - $CO_2D$ is the amount of $CO_2$ that is collected, but not passed through the destruction device;
  - $CO_2U$ is the amount of $CO_2$ in the generated landfill gas that is not collected and emitted as indirect emissions through the landfill cover surface as $CO_2$. This quantity of $CO_2$ is not adjusted for oxidation as the gas passes through the cover.

The annual amount of $CH_4$ that is collected, or recovered, from the landfill gas and sent to the destruction device can be calculated using Equation N.4. The $CH_4$ concentration in the landfill gas is typically monitored, or may be assumed as a percentage between 45% and 55%.

$$\mathbf{CH_4R = V \times \frac{C_{CH_4}}{100\%} \times 0.0423 \times \frac{520^\circ\ R}{T} \times \frac{P}{1\ atm} \times \frac{0.454\ metric\ ton}{1{,}000\ lbs}} \qquad \textbf{(EQ. N.4)}$$

Where:

| | |
|---|---|
| $CH_4R$ | = amount of $CH_4$ recovered from the landfill and sent to the landfill gas destruction device (metric tons $CH_4$/year). |
| V | = annual volumetric flow rate of biogas to the landfill gas destruction device (cubic feet biogas per year), as determined from daily monitoring. |
| $C_{CH_4}$ | = average annual $CH_4$ concentration of biogas (percent, fraction, wet basis). |
| 0.0423 | = density of $CH_4$ pounds per standard cubic foot (at 520ºR or 15.74ºC and 1 atm). |
| T | = annual average temperature (ºR) at which flow is measured. |
| P | = annual average pressure (atm) at which flow is measured. |
| 0.454/1000 | = conversion factor from pounds to metric tons. |

Equation N.5 can be used to calculate the quantity of $CH_4$ destroyed in a landfill gas destruction device. As mentioned previously, achieving 100% destruction efficiency is not feasible, thus the amount of $CH_4$ recovered must be adjusted by the destruction, or combustion, efficiency of the landfill gas destruction device. This adjustment accounts for the proportion of collected $CH_4$ in the biogas that is not destroyed by the destruction device. The collected $CH_4$ in the landfill gas that is not combusted is a direct source of $CH_4$ emissions to the atmosphere.

$$\mathbf{CH_4D = CH_4R \times DE} \qquad \textbf{(EQ. N.5)}$$

Where:

CH$_4$D = $CH_4$ destroyed at a landfill gas destruction device (metric tons $CH_4$/year).

CH$_4$R = amount of $CH_4$ recovered and sent to the landfill gas destruction device (Equation N.4).

DE = $CH_4$ destruction efficiency from flaring or combustion in an EGU, decimal percent. The DE varies with the type of landfill gas destruction device used; it can be estimated as the lesser of the manufacturer's specified destruction efficiency and 0.99 (EPA, 2013c).

The presence of a landfill gas collection system affects the amount of indirect $CH_4$ emissions from the landfill. When calculating the amount of indirect $CH_4$ emitted from a landfill with gas collection, only the uncollected portion of $CH_4$ in the landfill gas is adjusted for oxidation. Alternatively, for a landfill without gas collection, all of the $CH_4$ in the generated landfill gas is adjusted for oxidation. Equations N.6 and N.7 can be used to determine the amount of uncollected, or indirect, $CH_4$ emissions from a landfill with gas collection and a landfill without gas collection, respectively.

Both equations adjust the amount of $CH_4$ recovered by the term, 1/CE, which represents the portion of generated landfill gas that is not collected by the gas collection system. Equation N.6 subtracts the term $CH_4R$ to account for the quantity of $CH_4$ that is collected and sent to the destruction device so that only the uncollected portion is adjusted for oxidation. Note that this term is not included in Equation N.7 because all $CH_4$ generated must be adjusted for oxidation in landfills without gas collection.

$$\mathbf{CH_4U, gas\ collection} = \left(\left(\frac{\mathbf{1}}{\boldsymbol{CE}} \times \boldsymbol{CH_4R}\right) - \boldsymbol{CH_4R}\right) \times (\mathbf{1} - \boldsymbol{OX}) \qquad \textbf{(EQ. N.6)}$$

$$\mathbf{CH_4U, without\ gas\ collection} = \left(\frac{\mathbf{1}}{\boldsymbol{CE}} \times \boldsymbol{CH_4R}\right) \times (\mathbf{1} - \boldsymbol{OX}) \qquad \textbf{(EQ. N.7)}$$

Where:

CE = collection efficiency of the landfill gas collection system, decimal percent

CH$_4$R = the amount of $CH_4$ recovered and sent to the landfill gas device (Equation N.4)

OX = methane oxidation fraction

The amount of $CO_2$ recovered from the landfill gas that is sent to the destruction device can be calculated using Equation N.8.

$$\mathbf{CO_2R = V \times \left(1 - \frac{C_{CH_4}}{100\%}\right) \times 0.1166 \times \frac{520°R}{T} \times \frac{P}{1\ atm} \times \frac{0.454\ metric\ ton}{1{,}000\ lbs}} \qquad \textbf{(EQ. N.8)}$$

Where:

| | |
|---|---|
| $CO_2R$ | = the amount of $CO_2$ recovered and sent to the landfill gas destruction device (metric tons $CO_2$/year). |
| V | = annual volumetric flow rate of biogas to the landfill gas destruction device (cubic feet biogas per year), as determined from daily monitoring. |
| $(1 - C_{CH_4}/100\%)$ | = average annual $CO_2$ concentration of landfill gas, (C = average annual $CH_4$ concentration of biogas, percent, fraction wet basis). |
| 0.1160 | = density of $CO_2$ pounds per standard cubic foot (at 520ºR or 15.74ºC and 1 unit of average annual pressure [atm]). |
| T | = average annual temperature (ºR) at which flow is measured. |
| P | = atm at which flow is measured. |
| 0.454/1000 | = conversion factor from pounds to metric tons. |

Calculating indirect $CO_2$ emissions from the landfill surface is similar to that used to calculate indirect $CH_4$ emissions from the landfill surface (see Equations N.6 and N.7). Equations N.9 and N.10 present two ways to calculate indirect $CO_2$ emissions from either a landfill with a gas collection system, or one without.

Both equations are adjusted by the CE in order to consider only the portion of uncollected $CO_2$ that is emitted as $CO_2$ through the landfill cover surface and the portion of uncollected $CH_4$ that is emitted through the landfill cover surface and oxidized to $CO_2$ by the methanotrophic bacteria. The conversion factor of 44/16 is applied to the portion of $CH_4$ in the uncollected gas that is oxidized to $CO_2$.

$$\mathbf{CO_2U, gas\ collection} = \left(\left(\frac{1}{CE} \times CO_2R\right) - CO_2R\right) + OX\left(\left(\frac{1}{CE} \times CH_4R\right) - CH_4R\right) \times \frac{44}{16} \quad \textbf{(EQ. N.9)}$$

$$\mathbf{CO_2U, without\ gas\ collection} = \frac{1}{CE} \times CO_2R + \left(OX \times \frac{1}{CE} \times CH_4R\right) \times \frac{44}{16} \quad \textbf{(EQ. N.10)}$$

Where:

| | |
|---|---|
| CE | = collection efficiency of the landfill gas collection system, decimal percent. |
| $CO_2R$ | = amount of $CO_2$ recovered and sent to the landfill gas destruction device (Equation N.8). |
| $CH_4R$ | = the amount of $CH_4$ recovered and sent to the landfill gas destruction device (Equation N.4). |
| OX | = methane oxidation fraction. |
| 44/16 | = molecular weight ratio of $CO_2$ to $CH_4$. |

Several parameters are presented and used in the equations in the remainder of this section. Table N-4 presents the parameters used, typical or default values, ranges presented in the literature, and references.

**Table N-4. Summary of Parameters Used When Calculating a *BAF* for MSW Landfilling.**

| Parameter Description | Symbol | Value Used in Examples | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Oxidation fraction** | OX | 0.10 | 0.10 to 0.35 | Fraction | 0.10 is the default used in many accounting methodologies | IPCC, 2006 |
| **Oxidation fraction** | OX | 0.25 | 0.10 to 0.35 | Fraction | Higher oxidation fractions are observed for landfills with gas collection systems and low $CH_4$ flux rates | EPA, 2013c; SWICS, 2009 |
| **Concentration of $CH_4$ in the landfill gas or biogas** | $C_{CH_4}$ | 0.55 | 0.45 to 0.60 | Percent | | IPCC, 2006 |
| **Collection efficiency** | CE | 0.75 | 0.60 to 0.95 | Fraction | Higher CEs are associated with closed landfills and well-designed systems with low permeable covers | EPA, 2010b; EPA, 2013c |
| **Destruction efficiency (of a landfill gas flare)** | DE | 0.99 | 0.90 to 0.9977 | Fraction | 0.99 is considered the default DE of $CH_4$ for a flare | EPA, 2011a; EPA, 2013c |
| **Destruction efficiency (of an EGU)** | DE | 0.97 | 0.96 to 0.99 | Fraction | DE of $CH_4$ in a direct use system (e.g., boilers, heaters) varies by technology | EPA, 2011a; EPA, 2013c |
| **Density of $CH_4$ in landfill gas** | – | 0.0423 | – | lbs/scf | At 520 ºR or 15.74 ºC and 1 atm | EPA, 2011a; EPA, 2013c |

| Parameter Description | Symbol | Value Used in Examples | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Density of $CO_2$ in landfill gas** | – | 0.1160 | – | lbs/scf | At 520 ºR or 15.74 ºC and 1 atm | Calculated value[1] |

CE = collection efficiency; DE = destruction efficiency; EGU = electricity generating unit; F = fraction of $CH_4$ in landfill gas; lbs/scf = pounds per standard cubic foot; OX = oxidation fraction; R = Rankine

[1] This value is calculated using a 60 degree Fahrenheit conversion: 44.01 * (2.20462/836.6) = 0.1160, where 44.01 = the molecular weight of $CO_2$; 2.20462 is a unit conversion factor from kilograms to pounds; and 836.6 scf/kg-mol is the molar volume conversion factor.

## 2.4. Example *AVOIDEMIT* and *BAF* Calculations for Landfill Biogas

Three example scenarios are presented here for calculating a *BAF* value for landfill gas. In order to derive a *BAF* value for landfill gas, the numerator and denominator of the *AVOIDEMIT* must be calculated specific to the treatment and alternate fate of the collected landfill gas feedstock. Scenarios differ by the treatment of the collected gas (the denominator in the *AVOIDEMIT* term) and the alternate fate of the collected gas (the numerator in the *AVOIDEMIT* term).

### 2.4.1. Example Calculations for a Controlled Landfill (Flaring) Compared to an Uncontrolled Landfill

In this example, a *BAF* value is calculated for the treatment of collected gas via flares (denominator) and the alternate fate is to not collect or control any gas generated in the landfill (numerator). Equations N.4 through N.10 can be used to determine the inputs into Equation N.3 as shown below.

In this example, the landfill with gas collection recovered approximately 150 million cubic feet of landfill gas in the past year. The landfill gas monitoring system automatically corrects for temperature and pressure, and computed an annual average $CH_4$ concentration in the gas of 55%.

#### Step 1: Calculate the Amount of $CH_4$ and $CO_2$ Recovered by the Landfill Gas Collection System

The starting point for both treatment fates for treatment through MSW landfilling is the amount of gas recovered. Equations N.4 and N.8 can be used to calculate the amount of $CH_4$ and $CO_2$ recovered by the landfill gas collection system:

$$\mathbf{CH_4R = 150,000,000 \times \frac{55}{100} \times 0.0423 \times \frac{520}{520} \times \frac{1}{1} \times \frac{0.454}{1{,}000} = 1584.35}\ \boldsymbol{MT\ CH_4}$$

$$\mathbf{CO_2R = 150,000,000 \times \left(1 - \frac{55}{100}\right) \times 0.1160 \times \frac{520}{520} \times \frac{1}{1} \times \frac{0.454}{1{,}000} = 3554.82}\ \boldsymbol{MT\ CO_2}$$

#### Step 2: Calculate the $CO_2e$ Emissions for MSW Landfilling without Biogas Collection and Control

The numerator calculates the $CO_2e$ emissions profile of the biogas feedstock had the gas not been collected and combusted. Because there is no gas collection or control for the alternate fate, the $CH_4R$, $CH_4D$, $CO_2R$, and $CO_2D$ terms in Equation N.3 can be dropped, leaving only the $CH_4U$ and $CO_2U$ terms. Equations N.7 and N.10 can be used to calculate the amount of indirect $CH_4$ and $CO_2$ emitted by a landfill without gas collection and control, assuming a representative CE of 75% and 10% oxidation fraction:

$$\mathbf{CH_4U, without\ gas\ collection} = \left(\frac{1}{0.75} \times 1584.35\right) \times (1 - 0.10) = 1901.22\ MT\ CH_4$$

$$\mathbf{CO_2U, without\ gas\ collection} = \frac{1}{0.75} \times 3554.82 + \left(0.10 \times \frac{1}{0.75} \times 1584.35\right) \times \frac{44}{16} = 5320.69\ MT\ CO_2$$

The net $CO_2e$ emissions profile of the gas feedstock had the gas not been collected and combusted is calculated using Equation N.3:

$$CO_2e\ emissions\ from\ MSW\ landfilling\ without\ gas\ collection$$
$$= 25(0 - 0 + 1901.22) + \left(0 \times \frac{44}{16}\right) + 0 + 5320.69 = 52,851.08\ MT\ CO_2e$$

### Step 3: Calculate the $CO_2e$ Emissions with Gas Collection and Control (Flaring)

The denominator calculates the $CO_2e$ emissions profile of the gas feedstock had the gas been collected and combusted using a flare with a destruction efficiency of 99%. Equation N.5 can be used to calculate the $CH_4D$ term in Equation N.3:

$$\mathbf{CH_4D} = 1584.35 \times 0.99 = 1568.50\ \mathrm{MT}\ CH_4$$

Additionally, Equations N.6 and N.9 can be used to calculate the amount of indirect $CH_4$ and $CO_2$ emitted by a landfill with gas collection and control, assuming a representative CE of 75% and 10% oxidation fraction:

$$\mathbf{CH_4U, gas\ collection} = \left(\left(\frac{1}{0.75} \times 1584.35\right) - 1584.35\right) \times (1 - 0.10) = 475.30\ MT\ CH_4$$

$$\mathbf{CO_2U, gas\ collection}$$
$$= \left(\left(\frac{1}{0.75} \times 3554.82\right) - 3554.82\right) + 0.10\left(\left(\frac{1}{0.75} \times 1584.35\right) - 1584.35\right) \times \frac{44}{16} = 1330.17\ MT\ CO_2$$

Equation N.3 can now be used to calculate the $CO_2e$ emissions profile of the feedstock given that the gas was collected and combusted via flaring:

$$CO_2e\ emissions\ from\ MSW\ landfilling\ with\ gas\ collection$$
$$= 25(1584.35 - 1568.30 + 475.30) + \left(1568.50 \times \frac{44}{16}\right) + 3554.82 + 1330.17 = 21,477.06\ MT\ CO_2e$$

### Step 4: Calculate the *BAF* Value

Bringing the numerator and the denominator into the *AVOIDEMIT* term and calculating the assessment factor equation (Equation N.1 and Equation N.2) results in:

***BAF = AVOIDEMIT = 1 – (52,851.08 / 21,477.06)***

$$\boldsymbol{BAF = -1.46}$$

Negative *BAF* values, such as that calculated in Example 1, indicate that combustion of collected landfill gas feedstock by a stationary source results in a net $CO_2$e emissions reduction relative to releasing the collected landfill gas directly to the atmosphere without gas collection and combustion.

### 2.4.2. Example Calculations for a Controlled Landfill (EGU) Compared to an Uncontrolled Landfill

In this example, the same annual volume of landfill gas has been collected as in Section 2.4.1 and a *BAF* value is calculated for the treatment of collected gas via an EGU (denominator). The alternate treatment fate is similar to the numerator calculated in Section 2.4.1, thus the value of the numerator is the same as in Section 2.4.1. The denominator is also similar to that calculated in Section 2.4.1except that the gas DE is 0.97 instead of 0.99 because an EGU typically has a lower DE than a flare. The offsets from electricity generation by the EGU are not included in the framework.

#### Step 1—Calculate the $CO_2$e Emissions for MSW Landfilling without Gas Collection and Control

The numerator will be the same as that calculated in Example 1 when the same CE and OX values are used (Equations N.7 and N.10). Similar to Example 1, the net $CO_2$e emissions profile of the biogas feedstock had the gas not been collected and combusted is calculated using Equation N.3:

$$\boldsymbol{CO_2e\ emissions\ from\ MSW\ landfilling\ without\ gas\ collection}$$

$$\boldsymbol{= 25(0 - 0 + 1901.22) + \left(0 \times \frac{44}{16}\right) + 0 + 5320.69 = 52,851.08\ MT\ CO_2e}$$

#### Step 2—Calculate the $CO_2$e Emissions with Gas Collection and Control (EGU)

The denominator calculates the $CO_2$e emissions profile of the biogas feedstock had the biogas been collected and combusted in an EGU with a DE of 97%. Equation N.5 can be used to calculate the $CH_4$D term in Equation N.3:

$$\boldsymbol{\mathrm{CH_4D} = 1584.35\ \times 0.97 = 1536.82\ \mathrm{MT}\ CH_4}$$

The quantities of indirect $CH_4$ and $CO_2$ (Equations N.6 and N.9) will be the same as those presented in Section 2.4.1:

$$\boldsymbol{\mathrm{CH_4U, gas\ collection} = \left(\left(\frac{1}{0.75} \times\ 1584.35\ \right) -\ 1584.35\ \right) \times\ (1 - 0.10) = 475.31\ MT\ CH_4}$$

$$\boldsymbol{\mathrm{CO_2U, gas\ collection} = \left(\left(\frac{1}{0.75} \times 3554.82\right) - 3554.82\right) +\ 0.10\left(\left(\frac{1}{0.75} \times 1584.35\right) - 1584.35\right) \times \frac{44}{16} = 1330.17\ MT\ CO_2}$$

Equation N.3 can now be used to calculate the $CO_2$e emissions profile of the feedstock given that the biogas was collected and combusted via an EGU:

$$\boldsymbol{CO_2e\ emissions\ from\ MSW\ landfilling\ with\ gas\ collection}$$

$$= \mathbf{25(1584.35 - 1536.82 + 475.30) + \left(1536.82 \times \frac{44}{16}\right) + 3554.82 + 1330.17}$$
$$= \mathbf{22,182.09}\ \boldsymbol{MT\ CO_2e}$$

### Step 4: Calculate the *BAF* Value

Bringing the numerator and the denominator into the *AVOIDEMIT* term and calculating the assessment factor equation (Equations N.1 and N.2) results in:

***BAF = AVOIDEMIT = 1 – (52,851.08 / 22,182.09)***

***BAF = -1.38***

The *BAF* for this example is slightly greater than the *BAF* of −1.46 calculated in Section 2.4.1 as a result of the lower DE of the EGU relative to combustion using a flare.

## 2.4.3. Example Calculations for a Controlled Landfill (EGU) with a Gas Collection System Installed Mid-Way through the Year Compared to an Uncontrolled Landfill

In this example, the same annual volume of landfill gas has been collected as in Section 2.4.1 and 2.4.2. A *BAF* value is calculated for the treatment of collected gas via a gas collection system and EGU (denominator) that was operationalized midway through the year. The alternate treatment fate is similar to the numerator calculated in Section 2.4.1 and 2.4.2, thus the value of the numerator is the same as in Section 2.4.1 and 2.4.2. The method of calculating the denominator is different from that presented in Section 2.4.1 and 2.4.2in that an extra term has been added to account for the fraction of hours the gas collection system and control device operated during the year (fRec).

### Step 1: Calculate the Amount of $CH_4$ and $CO_2$ Recovered by the Landfill Gas Collection System

The starting point for both treatment fates, the amount of gas recovered, is the same as Section 2.4.1 and 2.4.2 (Equations N.4 and N.8):

$$\mathbf{CH_4R = 150,000,000 \times \frac{55}{100} \times 0.0423 \times \frac{520}{520} \times \frac{1}{1} \times \frac{0.454}{1,000} = 1584.35}\ \boldsymbol{MT\ CH_4}$$

$$\mathbf{CO_2R = 150,000,000 \times \left(1 - \frac{55}{100}\right) \times 0.1160 \times \frac{520}{520} \times \frac{1}{1} \times \frac{0.454}{1,000} = 3554.82}\ \boldsymbol{MT\ CO_2}$$

### Step 2: Calculate the Fraction of Hours the Recovery System Operated During the Year

fRec = actual operating hours of the recovery system/number of hours in the year

In this example, the gas collection system was installed and fully operational on May 1st in a non-leap year. There are 244 days between May 1st and December 31st, or 5856 hours. Therefore, fRec = 5856/8760 = 0.66849.

### Step 3: Calculate the $CO_2$e Emissions for MSW Landfilling without Gas Collection and Control

Equations N.7 and N.10 are slightly modified by dividing the amount of recovered $CH_4$ and $CO_2$ by fRec to give Equations N.11 and N.12:

$$\mathbf{CH_4U, without\ gas\ collection} = \left(\frac{1}{CE} \times \frac{CH_4R}{fRec}\right) \times (1 - OX) \qquad \textbf{(EQ. N.21)}$$

$$\mathbf{CH_4U, without\ gas\ collection} = \left(\frac{1}{0.75} \times \frac{1584.35}{0.66849}\right) \times (1 - 0.10) = 2,837.64\ MT\ CH_4$$

$$\mathbf{CO_2U, without\ gas\ collection} = \frac{1}{CE} \times \frac{CO_2R}{fRec} + \left(OX \times \frac{1}{CE} \times \frac{CH_4R}{fRec}\right) \times \frac{44}{16} \qquad \textbf{(EQ. N.12)}$$

$$\mathbf{CO_2U, without\ gas\ collection} = \frac{1}{0.75} \times \frac{3554.82}{0.66849} + \left(0.10 \times \frac{1}{0.75} \times \frac{1584.35}{0.66849}\right) \times \frac{44}{16} = 7,941.32\ MT\ CO_2$$

Similar to Section 2.4.1 and 2.4.2, the net $CO_2$e emissions profile of the gas feedstock had the gas not been collected and combusted is calculated using Equation N.3:

**$CO_2$e emissions from MSW landfilling without gas collection =**

$$25(0 - 0 + 2837.64) + \left(0 \times \frac{44}{16}\right) + 0 + 7941.32 = 78,882.21\ MT\ CO_2e$$

### Step 4: Calculate the $CO_2$e emissions with gas collection and control (EGU)

The denominator calculates the $CO_2$e emissions profile of the gas feedstock had the biogas been collected and combusted in an EGU with a DE of 97%. Equation N.5 can be used to calculate the $CH_4$D term in Equation N.3:

$$\mathbf{CH_4D} = 1584.35 \times 0.97 = 1536.82\ \mathrm{MT}\ CH_4$$

The quantities of indirect $CH_4$ and $CO_2$ (Equations N.6 and N.9) will be the similar to those presented in Examples 1 and 2, except that fRec must now be factored into Equations N.6 and N.9 to give Equations N.13 and N.14:

$$\mathbf{CH_4U, gas\ collection} = \left(\left(\frac{1}{CE} \times \frac{CH_4R}{fRec}\right) - CH_4R\right) \times (1 - OX) \qquad \textbf{(EQ. N.33)}$$

$$\mathbf{CH_4U, gas\ collection} = \left(\left(\frac{1}{0.75} \times \frac{1584.35}{0.66849}\right) - 1584.35\right) \times (1 - 0.10)$$
$$= 1418.12\ MT\ CH_4$$

$$\mathbf{CO_2U, without\ gas\ collection} = \frac{1}{CE} \times \frac{CO_2R}{fRec} + \left(OX \times \frac{1}{CE} \times \frac{CH_4R}{fRec}\right) \times \frac{44}{16} \qquad \textbf{(EQ. N.44)}$$

$$\mathbf{CO_2U, gas\ collection} = \left(\left(\frac{1}{0.75} \times \frac{3554.82}{0.66849}\right) - 3554.82\right) +$$

$$0.10\left(\left(\frac{1}{0.75} \times \frac{1584.35}{0.6649}\right) - 1584.35\right) \times \frac{44}{16} = 3950.81\ MT\ CO_2$$

Equation N.3 can now be used to calculate the $CO_2e$ emissions profile of the feedstock given that the biogas was collected and combusted via an EGU:

$CO_2e$ emissions from MSW landfilling with gas collection

**= 25 (1584.35 – 1536.82 + 1411.72) + (1536.82 x 44/16) + 3554.82 + 3950.81 = 48,213.22 MT $CO_2e$**

### Step 5: Calculate the *BAF* Value

Bringing the numerator and the denominator into the *AVOIDEMIT* term and calculating the assessment factor equation (Equation N.1 and N.2) results in:

***BAF* = *AVOIDEMIT* = 1 – (78,882.21 / 48,213.22)**

***BAF* = −0.64**

The *BAF* for this example is approximately two and a half times greater than the *BAF* of −1.50 calculated in Section 2.4.2 as a result of the fraction of hours the gas collection and control system were operational during the year.

## 2.5. Sensitivity Analysis for MSW Landfill Biogas

A simple sensitivity analysis is presented to better understand the relationship between and impact of certain key parameters in the framework for MSW landfilling. Key parameters specific to MSW landfilling include the oxidation fraction (OX), the collection efficiency (CE) of the landfill gas collection system, the destruction efficiency (DE) of the selected combustion device, and the $CH_4$ GWP used (i.e., 21, 25, or 28). Table N-5 presents the range of *BAF* values after modifying the key parameters and using the inputs from Example 2-1 in Section 2.3 of this appendix. Sources for the parameter values used here can be found in Table N-4 of Section 2.3.2. The actual fate is MSW landfilling with flaring and the alternate fate is MSW landfilling without gas collection and combustion.

Two categories of analyses are presented in Table N-5: the first (1a through 1d) compares the impact of modifying the CE and DE values, while the second (2a through 2d) compares the impact of modifying all 3 key parameters. In the second set of analyses, a value for OX other than the representative value of 0.10 was used in the actual fate (i.e., denominator) calculations. The only difference between the a, b, c, and d analyses is the change in OX factors. For example, when comparing Analyses 1a and 2a, the only difference is that 1a uses an OX of 0.10 for both the actual and alternate fates, while 2a uses different OX values for each fate. Analyses 2a, b, c, and d yield lower *BAF* values than Analyses 1a, b, c, and d. Analysis 2c yields the lowest *BAF* values and Analysis 1b yields the highest *BAF* value. Despite modifying the key parameters, all *BAF* values are negative.

**Table N-5. Sensitivity Analysis for MSW Landfilling.**

| Analysis | Key Parameter and Value | | | BAF | | |
|---|---|---|---|---|---|---|
| | OX | CE | Flare DE | GWP=21 | GWP=25 | GWP=28 |
| **1a** | 0.10 | 0.75 | 0.99 | −1.319 | −1.461 | −1.551 |
| **2a** | Without GCS = 0.10<br>With GCS = 0.25 | 0.75 | 0.99 | −1.504 | −1.681 | −1.795 |
| **1b** | 0.10 | 0.75 | 0.98 | −1.285 | −1.421 | −1.508 |
| **2b** | Without GCS = 0.10<br>With GCS = 0.25 | 0.75 | 0.98 | −1.465 | −1.634 | −1.743 |
| **1c** | 0.10 | 0.95 | 0.99 | −2.577 | −3.031 | −3.352 |
| **2c** | Without GCS = 0.10<br>With GCS = 0.25 | 0.95 | 0.99 | −2.660 | −3.143 | −3.485 |
| **1d** | 0.10 | 0.75 | 0.99 | −1.319 | −1.461 | −1.551 |
| **2d** | Without GCS = 0.10<br>With GCS = 0.25 | 0.75 | 0.99 | −1.719 | −1.940 | −2.086 |

Note: References for the key parameters and values are presented in Table N-4 of Section 2.3.2.
Note: Methane is a potent GHG, with a 100-year GWP of 21 (IPCC, 1996). It should be noted that in the IPCC Fourth Assessment Report, the 100-year GWP of $CH_4$ was revised to 25 (IPCC, 2007). To comply with international reporting standards under the UNFCCC, official emission estimates reported by the United States use the IPCC Second Assessment Report GWP values (IPCC, 1996). The United States will transition to using the revised GWPs beginning in 2015. In this framework, the GWP of 25 for the central examples within each section. The GWPs of 21 and 28 are used in the sensitivity analyses for each section.

# 3. Disposal of Biogenic MSW through Combustion and Associated GHG Emissions Pathways

As an alternative to disposing of MSW in a landfill, it can be directly combusted in waste-to-energy facilities to generate electricity. In the United States, almost all incineration of MSW occurs at waste-to-energy facilities or industrial facilities where energy is recovered (EPA, 2014b). Based on data from EPA's Greenhouse Gas Reporting Program (GHGRP) and EPA's Emissions and Generation Resource Integrated Database (eGRID), there are roughly 142 MSW combustors in the United States that emit approximately 30 million metric tons of biogenic $CO_2e$.[11] Incineration oxidizes almost all of the carbon in the MSW to $CO_2$ (Astrup et al., 2009). Generally less than 0.5% of the carbon remains in the ashes (i.e., it is not emitted to the atmosphere, Astrup et al., 2009).

Although MSW consists mainly of biogenic resources such as food, paper, and wood products, it also includes resources derived from fossil fuels, such as tires[12] and plastics. After the MSW is delivered

[11] Based on GHGRP data for the 2011 reporting year and eGRID data for the 2009 reporting year.
[12] Tires contain a biogenic component in the form of natural rubber. Whole tires (including steel, etc.) from the combined grouping of passenger vehicles and trucks are, on average, composed of 28% natural rubber (Rubber Manufacturers Association, unpublished data 2013). Tire-derived-fuel is used in cement kilns, utility boilers, pulp and paper mills, industrial boilers, and dedicated scrap tire-to-energy facilities (EPA 2009b).

to a stationary source facility, it is incinerated in an EGU either "as is" (mass burn without recovery of recyclables), as refuse-derived fuel (burn after recyclables have been recovered), or combustion with energy recovery of source separated materials in MSW (e.g., wood pallets and tire-derived fuel; EPA, 2009b). Point source stack emissions from combustion of biogenic MSW feedstocks are primarily $CO_2$. For the purposes of this document, biogenic MSW is the feedstock when disposed of in a combustor.

## 3.1. Method for Calculating an Illustrative *BAF* Value for Biogenic Emissions Resulting From MSW Combustion

The assessment factor equation can be applied to direct biogenic $CO_2$ emissions from MSW combustion. This section provides an illustrative method for calculating a *BAF* value that is applied to direct biogenic $CO_2$ emissions from MSW combustors. Here, the biogenic feedstock is MSW that is collected and incinerated, oxidizing the biogenic waste-feedstock to $CO_2$ emissions.

Landfilling the biogenic MSW can be considered the alternate fate of the MSW feedstock had it not been incinerated. The emissions profile resulting from this alternate fate represents the numerator of the emissions ratio term in *AVOIDEMIT*. Were the MSW to have been disposed of in a landfill, it would undergo anaerobic decomposition, resulting in biogas that may or may not be collected and destroyed by combustion. However, a portion of the carbon in the biogenic waste-derived feedstock does not decompose in the landfill; instead that carbon is stored in the landfill. Such storage effectively removes the remaining landfilled carbon from the global carbon cycle by transferring that carbon into long-term storage within a landfill (Staley and Barlaz, 2009).[13] The factors affecting degradation can result in the long-term, potentially permanent, carbon storage of approximately 50% of total landfilled organic carbon (Bogner et al., 2007; Manfredi et al., 2009).

In applying the assessment factor equation, net GHG emissions reductions are accounted for in the *AVOIDEMIT* term. In practice, as applied here, the *AVOIDEMIT* term is a ratio expressed as $tCO_2e$ avoided (i.e., the emissions, in $CO_2e$, of the MSW had it not been combusted) per $tCO_2e$ removed via combustion (i.e., the emissions, in $CO_2e$, of the combusted MSW). For the MSW feedstock incinerated in an MSW combustor, the *AVOIDEMIT* term can be conceptually expressed by the simplified ratio of:

$$\boldsymbol{AVOIDEMIT = 1 - \frac{(\textbf{emissions from non-combustion treatment of MSW})}{(\textbf{emissions from MSW combustion})}} \qquad \textbf{(EQ. N.15)}$$

### 3.1.1. Calculating the Numerator

In computing *AVOIDEMIT*, the numerator (i.e., emissions from MSW treatment alternative to incineration) can be calculated under the assumption that had the MSW not been incinerated, it

[13] While cellulose, hemicellulose, and lignin (present in paper and wood products) can degrade and be converted to $CH_4$ in landfills, the anaerobic conditions in landfills prevent their full degradation (Bogner, 1992; Barlaz, 2006; Wang et al., 2011). Furthermore, the presence of lignin can inhibit cellulose and hemicellulose degradation (Micales and Skog, 1997; Barlaz, 2006). Because lignins effectively prevent degradation, between 84% and 100% of the initial carbon in landfilled wood products is sequestered indefinitely (Micales and Skog, 1997; Wang et al., 2011). The extent of decomposition varies between types of wood (Wang et al., 2011).

would have been landfilled. Given this alternate fate, the value of the numerator must account for the fraction of landfilled MSW that decays anaerobically, thereby producing landfill gas that may or may not be collected. If the landfill gas is collected and combusted (e.g., flared), the collection efficiency and destruction efficiency must be accounted for. In accounting for indirect emissions from the landfill cap, the $CH_4$ oxidation via the landfill cover soils must be accounted for. And finally, the fraction of landfilled MSW that does not decay such that biogenic carbon is stored within the landfill must also be accounted for. The following equation can be used in the numerator of the *AVOIDEMIT* term:

$CO_2e$ emissions avoided by landfilling the MSW (kg $CO_2e$/metric ton MSW wet weight) =

$$\mathbf{GWP_{CH_4} \times (CH_{4soils} + CH_{4combustion}) + CO_{2soils} + CO_{2combustion}} \quad \textbf{(EQ. N.16)}$$

Where:

$GWP_{CH_4}$ = 100-year GWP of $CH_4$, 25 (IPCC, 2007)

$$\mathbf{CH_{4soils} = CH_{4generated} \times (1 - CE) \times (1 - OX)} \quad \textbf{(EQ. N.57)}$$

$$\mathbf{CH_{4combustion} = CH_{4generated} \times (CE) \times (1 - DE)} \quad \textbf{(EQ. N.68)}$$

Where:

$$\mathbf{CH_{4generated} = C \times D_{lfg} \times (\%CH_4) \times (16/12)} \quad \textbf{(EQ. N.79)}$$

C = amount of biogenic C in MSW (kg C/metric ton MSW wet weight).

$D_{lfg}$ = dissimilation coefficient (fraction of biogenic C that leaves the landfill via decomposition of biogenic waste).

$\%CH_4$ = proportion of gas that is $CH_4$.

(16/12) = molecular weight ratio of $CH_4$ to C.

CE = gas collection efficiency.

OX = $CH_4$ oxidation factor associated with the landfill cover soil.

DE = gas destruction efficiency (i.e., combustion efficiency of flare or EGU).

$$\mathbf{CO_{2soils} = (CO_{2generated} + CH_{4generated} \times 44/16 \times OX) \times (1 - CE)} \quad \textbf{(EQ. N.20)}$$

$$\mathbf{CO_{2combustion} = (CO_{2generated} + CH_{4generated} \times 44/16 \times DE) \times CE} \quad \textbf{(EQ. N.21)}$$

Where:

$$\mathbf{CO_{2generated} = C \times D_{lfg} \times (\%CO_2) \times (44/12)} \quad \textbf{(EQ. N.82)}$$

Where:

C = amount of biogenic C in MSW (kg C/metric ton MSW wet weight).

$D_{lfg}$ = dissimilation coefficient (fraction of biogenic C that leaves the landfill via decomposition of biogenic waste).

$(\%CO_2)$ = proportion of gas that is $CO_2$.

(44/12) = molecular weight ratio of $CO_2$ to C.

CE = gas collection efficiency.

OX = $CH_4$ oxidation factor associated with the landfill cover soil.

DE = gas destruction efficiency (i.e., combustion efficiency of flare or EGU).

### 3.1.2. Calculating the Denominator

In the derivation of *AVOIDEMIT* (Equation N.15), the denominator (i.e., emissions from MSW combustion) is based on the carbon content of the point source, stack emissions from the MSW combustion device. The value of the denominator is equal to the $CO_2e$ of the combusted MSW, adjusted by the proportion of combusted biogenic carbon in MSW that is converted from C to $CO_2$. MSW combustion results in near-complete oxidation of C to $CO_2$; generally less than 0.5% of the initial amount of C remains in solid form (ash) post-combustion. The following equation can be used to calculate the total $CO_2e$ emissions from MSW combustion (the denominator of the *AVOIDEMIT* term):

$CO_2e$ emissions from MSW combustion (kg $CO_2e$/metric ton MSW wet weight) =

$$\mathbf{C \times 0.995 \times (44/12)} \qquad \textbf{(EQ. N.93)}$$

Where:

C = amount of biogenic C in MSW (kg C/metric ton MSW wet weight).

0.995 = proportion of C in MSW that is oxidized through combustion (i.e., combustion efficiency).

(44/12) = molecular weight ratio of $CO_2$ to C.

After solving for the numerator and the denominator of the *AVOIDEMIT* term, the *BAF* value can be calculated using Equation N.1. See Section 3.2 for an illustrative example calculation of the numerator and denominator in the *AVOIDEMIT* term and its subsequent application in estimating a *BAF* value.

Several parameters are presented and used in the equations in the remainder of this section. Table N-6 presents the parameters used, typical or default values, ranges presented in the literature, and references.

**Table N-6. Summary of Parameters Used When Calculating an Illustrative *BAF* for MSW Combustion.**

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Oxidation fraction** | OX | 0.10 | 0.10 to 0.35 | Fraction | 0.10 is the default used in many accounting methodologies | IPCC, 2006 |
| **Percent of $CH_4$ or $CO_2$ in the landfill gas** | %$CH_4$, %$CO_2$ | 0.55 | 0.45 to 0.60 | Percent | | IPCC, 2006 |
| **Collection efficiency** | CE | 0.75 | 0.60 to 0.95 | Fraction | Higher CEs are associated with closed landfills and well-designed systems with low permeable covers | EPA, 2010b; EPA, 2013c |
| **Destruction efficiency (of a landfill gas flare or EGU)** | DE | 0.99 | 0.90 to 0.9977 | Fraction | 0.99 is considered the default DE of $CH_4$ for a flare; EGUs may be slightly less | EPA, 2011a; EPA, 2013c |
| **Fraction of biogenic carbon in MSW** | C | 90 | Dependent on the composition of the MSW | Kilograms Carbon per metric ton of MSW, wet weight | | Staley and Barlaz, 2009 |
| **Dissimila-tion coefficient** | Dlfg | 0.50 | – | Percent | 50% of the biogenic carbon in the MSW goes into long-term storage | IPCC, 2006 |
| **Combustion efficiency of an MSW combustor** | – | 0.995 | 0.98 to 0.9999 | Fraction | The proportion of carbon in MSW that is oxidized through combustion | IPCC, 2006; Astrup et al., 2009 |

## 3.2. Example *AVOIDEMIT* and *BAF* Calculations for MSW Combustion

Two example scenarios are presented here for calculating a *BAF* value for MSW combustion compared to an alternate of landfilling with gas collection and an alternate of landfilling without gas collection. A hypothetical example is used to calculate *AVOIDEMIT* and the *BAF* for MSW combustion. Actual *AVOIDEMIT* and the *BAF* values will vary depending on the specific circumstances of MSW combustion and its alternate fate.

### 3.2.1. Example Calculations for MSW Combustion Compared to a Landfill with Gas Collection (EGU)

Equation N.16 can be used to calculate the numerator of the *AVOIDEMIT* term, but Equations N.17 through N.22 must be calculated first in order to solve for Equation N.16. To solve for the numerator, emissions from non-combustion treatment of MSW, it will be assumed that had the MSW not been combusted then it would have been landfilled, thereby producing biogas. Some of the biogas would have been collected and combusted, some would have been oxidized via the landfill cover soils, and the remainder would have been an indirect emission. All of these emission pathways are considered here because the feedstock for combustion of biogenic MSW is the biogenic MSW.

For this hypothetical example, the following conditions apply:

- The amount of biogenic carbon in MSW is estimated at 90 kg C/metric ton MSW wet weight (ww) (i.e., C = 90).
- The amount of biogenic carbon leaving the landfill is estimated at 50% ($D_{lfg}$ = 0.5), such that 50% goes into long-term storage.
- On a mass basis, 55% of the carbon becomes $CH_4$ and 45% of the carbon becomes $CO_2$.
- The fraction of $CH_4$ in the landfill gas that is oxidized via cover soils is the default of 0.10 (OX = 0.1).
- Landfill gas collection efficiency is 75% (CE = 0.75).
- Destruction efficiency of the collected landfill gas is 99% (DE = 0.99).

#### Step 1: Calculate the $CH_4$ and $CO_2$ Generated and Emitted by a Landfill with Gas Collection and Combustion in an EGU

To calculate the numerator, Equations N.19 and N.22 must first be solved:

$$\mathbf{CH_{4generated} = 90\ kg\ C\ per\ metric\ ton\ MSW\ ww \times 0.5 \times (55/100) \times (16/12)}$$

$$\mathbf{= 33.0\ kg\ CH_4\ per\ metric\ ton\ MSW\ ww}$$

$$\mathbf{CO_{2generated} = 90\ kg\ C\ per\ metric\ ton\ MSW\ ww \times 0.5 \times (45/100) \times (44/12)}$$

$$\mathbf{= 74.25\ CO_2\ per\ metric\ ton\ MSW\ ww}$$

Using the computed value from Equation N.19, Equation N.17 can be solved:

$$\mathbf{CH_{4soils} = 33.0 \times (1 - 0.75) \times (1 - 0.1)}$$

$$\mathbf{= 7.425\ kg\ CH_4\ per\ metric\ ton\ MSW\ ww}$$

Using the computed value from Equation N.19, Equation N.18 can be solved:

$$\mathbf{CH_{4combustion} = 33.0 \times (0.75) \times (1 - 0.99)}$$

$$\mathbf{= 0.2475\ kg\ CH_4\ per\ metric\ ton\ MSW\ ww}$$

Using the computed values from Equations N.19 and N.22, Equation N.20 can be solved:

$$\mathbf{CO_{2soils} = (74.25 + (33.0 \times 44/16 \times 0.1)) \times (1 - 0.75)}$$

$$\mathbf{= 20.8312\ kg\ CO_2\ per\ metric\ ton\ MSW\ ww}$$

Using the computed values from Equations N.19 and N.22, Equation N.21 can be solved:

$$\mathbf{CO_{2combustion} = (74.25 + (33.0 \times 44/16 \times 0.99)) \times 0.75}$$

$$\mathbf{= 123.0694\ kg\ CO_2\ per\ metric\ ton\ MSW\ ww}$$

Using the computed values from Equations N.17, N.18, N.20, and N.21, the numerator (i.e., Equation N.16) can be solved:

$CO_2e$ emissions avoided by not having landfilled the MSW (kg $CO_2e$/metric ton MSW ww)

$$\mathbf{= 25 \times (7.425 + 0.2475) + 20.8312 + 123.0694}$$

$$\mathbf{= 335.713\ kg\ CO_2e\ per\ metric\ ton\ MSW\ ww}$$

### Step 2: Calculate the $CO_2$ Emitted Through MSW Combustion

Next, the denominator (total $CO_2e$ emissions from MSW combustion) of the *AVOIDEMIT* term can be solved (kg $CO_2e$/metric ton MSW wet weight) using Equation N.23:

$CO_2e$ emissions from MSW combustion

$$\mathbf{= 90 \times 0.995 \times (44/12)}$$

$$\mathbf{= 328.350}$$

### Step 3: Calculate the *BAF* Value

Next, the *AVOIDEMIT* term can be computed and input into the assessment factor equation (Equations N.1 and N.2):

***BAF*** **= *AVOIDEMIT***

**= 1 – (335.711 / 328.350)**

**= −0.022**

It should be noted that this calculation does not take into account any reduction in fossil fuel usage as a result of any heat, power, or both that may have been generated in the MSW incineration process.

### 3.2.2. Example Calculations for MSW Combustion Compared to a Landfill without Gas Collection

If the alternate fate of the MSW had been landfilled without gas collection or combustion, then $CH_{4soils}$ and $CO_{2soils}$ would increase in value while $CH_{4combustion}$ and $CO_{2combustion}$ would both be 0.

#### Step 1: Calculate the $CH_4$ and $CO_2$ Generated and Emitted by a Landfill without Gas Collection

To calculate the numerator, Equations N.19 and N.22 must first be solved:

$$\mathbf{CH_{4generated} = 90\ kg\ C\ per\ metric\ ton\ MSW\ ww \times 0.5 \times (55/100) \times (16/12)}$$

$$\mathbf{= 33.0\ kg\ CH_4\ per\ metric\ ton\ MSW\ ww}$$

$$\mathbf{CO_{2generated} = 90\ kg\ C\ per\ metric\ ton\ MSW\ ww \times 0.5 \times (45/100) \times (44/12)}$$

$$\mathbf{= 74.25\ CO_2\ per\ metric\ ton\ MSW\ ww}$$

Using the computed value from Equation N.19, Equation N.17 can be solved. The collection efficiency is 0 in this equation since there is no gas collection system.

$$\mathbf{CH_{4soils} = 33.0 \times (1 - 0) \times (1 - 0.1)}$$

$$\mathbf{= 29.7\ kg\ CH_4\ per\ metric\ ton\ MSW\ ww}$$

Using the computed values from Equations N.19 and N.22, Equation N.20can be solved:

$$\mathbf{CO_{2soils} = (74.25 + (33.0 \times 44/16 \times 0.1)) \times (1 - 0)}$$

$$\mathbf{= 83.325\ kg\ CO_2\ per\ metric\ ton\ MSW\ ww}$$

Using the computed values from Equations N.17 and N.20 the numerator (i.e., Equation N.16) can be solved:

$CO_2e$ emissions avoided by not having landfilled the MSW (kg $CO_2e$/metric ton MSW ww)

$$\mathbf{= 25 \times (29.7 + 0) + 83.325 + 0}$$

$$\mathbf{= 825.825\ kg\ CO_2e\ per\ metric\ ton\ MSW\ ww}$$

### Step 2: Calculate the $CO_2e$ Emitted Through MSW Combustion

Next, the denominator (total $CO_2e$ emissions from MSW combustion) of the *AVOIDEMIT* term can be solved (kg $CO_2e$/metric ton MSW wet weight) using Equation N.23:

$CO_2e$ emissions from MSW combustion

$$= 90 \times 0.995 \times (44/12)$$

$$= 328.350$$

### Step 3: Calculate the *BAF* Value

Next, the *AVOIDEMIT* term can be computed and input into the assessment factor equation (Equations N.1 and N.2):

$$BAF = AVOIDEMIT$$

$$= 1 - (825.825 / 328.350)$$

$$= -1.52$$

It should be noted that this calculation does not take into account any reduction in fossil fuel usage as a result of any heat, power, or both that may have been generated in the MSW incineration process.

## 3.3. Sensitivity Analysis for MSW Combustion

A simple sensitivity analysis on the key parameters in the MSW combustion methodology is presented in Table N-7 for the actual fate of MSW combustion and the alternate fate of landfilling with gas collection and flaring. Key parameters impacting the *BAF* include the destruction and collection efficiencies (DE and CE, respectively) and the GWP for $CH_4$ (21, 25, and 28). In each of the six analyses, the DE of the landfill gas was adjusted between 97% and 99%, and the CE was adjusted 60% to 95%, representing a range of low to high performing landfill gas collection system efficiencies. Sources for the parameter values used here can be found in Table N-6 of Section 3.1.2. The inputs used in the analyses are equivalent to those shown in the example calculations in Section 3.2 of this appendix. Note that the carbon content of the MSW does not impact the calculated *BAF* values because it is a factor in both treatment fates and essentially cancels out.

The *BAF* values are most negative compared to the other analyses, when MSW combustion is compared to a landfill with a 75% CE and 97% DE, indicating that MSW combustion results in a greater reduction of $CO_2e$ emissions (see Analysis 4, regardless of the GWP value). Alternatively, the most positive *BAF* value is generated when MSW combustion is compared to a highly efficient landfill gas collection and combustion system (i.e., Analysis 6).

Analyses 7 through 12 highlight the areas where the *BAF* value changes from negative to positive as a result of the CE for each GWP value when the DE is held constant at 99%. The *BAF* values were all negative when a DE of 97% was held constant despite the changes in CE and GWP.

**Table N-7. Sensitivity Analysis for MSW Combustion.**

| Analysis | Key Parameter and Value | | *BAF* | | |
|---|---|---|---|---|---|
| | CE | DE | GWP=21 | GWP=25 | GWP=28 |
| **1** | 0.60 | 0.99 | −0.174 | −0.321 | −0.431 |
| **2** | 0.60 | 0.97 | −0.196 | −0.348 | −0.462 |
| **3** | 0.75 | 0.99 | −0.141 | −0.022 | −0.386 |
| **4** | 0.75 | 0.97 | −0.241 | −0.395 | −0.510 |
| **5** | 0.95 | 0.99 | 0.398 | 0.376 | 0.359 |
| **6** | 0.95 | 0.97 | −0.302 | −0.458 | −0.575 |
| **7a** | 0.70 | 0.99 | −0.011 | | |
| **8 a** | 0.71 | 0.99 | 0.006 | | |
| **9b** | 0.76 | 0.99 | | −0.003 | |
| **10 b** | 0.77 | 0.99 | | 0.017 | |
| **11c** | 0.79 | 0.99 | | | −0.002 |
| **12 c** | 0.80 | 0.99 | | | 0.020 |

[a] The point at which the *BAF* changes from negative to positive with a GWP of 21.
[b] The point at which the *BAF* changes from negative to positive with a GWP of 25.
[c] The point at which the *BAF* changes from negative to positive with a GWP of 28.
Note: References for the key parameters and values are presented in Table N-6 of Section 3.1.2.

# 4. Livestock Waste Management through Anaerobic Processes and Associated GHG Emissions Pathways

Livestock waste management can produce $CH_4$, $CO_2$, and $N_2O$ emissions. In 2012, livestock waste management emissions in the United States were estimated at 52.9 Tg $CO_2e$ for $CH_4$;[14] and 18.0 Tg $CO_2e$ for $N_2O$ (EPA, 2014b). Waste from dairy cattle and swine had the highest $CH_4$ emissions; waste from beef and dairy cattle had the highest $N_2O$ emissions (EPA, 2014b).

Methane is produced under anaerobic conditions in livestock waste storage and treatment systems, such as liquids or slurries in lagoons, ponds, tanks, pits, or piles. In the United States, the majority of livestock waste is handled as a solid (e.g., in stacks or drylots) or deposited on pasture, range, or paddock lands, where it tends to decompose aerobically and produce little or no $CH_4$ (EPA, 2014b). Carbon dioxide is also produced under anaerobic conditions and generated when $CH_4$ in the biogas is combusted.

Both direct and indirect $N_2O$ emissions are emitted during livestock waste management. Direct $N_2O$ emissions from livestock waste are produced as part of the nitrogen cycle through the nitrification and denitrification of the organic nitrogen in livestock dung and urine. The production of direct $N_2O$ emissions from livestock waste depends on the composition of the manure and urine, the type of

[14] This accounts for $CH_4$ reductions due to capture and destruction of $CH_4$ at facilities using anaerobic digesters.

bacteria involved in the process, and the amount of oxygen and liquid in the waste management system.

- For direct $N_2O$ emissions to occur, the manure must first be handled aerobically where ammonia ($NH_3$) or organic N is converted, via nitrification, to nitrates and nitrites, and then handled anaerobically where the nitrates and nitrites are reduced to dinitrogen gas ($N_2$), with intermediate production of $N_2O$ and nitric oxide (NO), via denitrification (EPA, 2014b). These emissions are most likely to occur in dry manure handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to saturation. A very small portion of the total N excreted is expected to convert to $N_2O$ in the waste management system.
- Indirect $N_2O$ emissions are produced when nitrogen is lost from the system through volatilization (as $NH_3$ or $NO_x$) or through runoff and leaching of nitrogen during waste treatment, storage and transportation (EPA, 2014b). The vast majority of volatilization losses from these operations are $NH_3$ (EPA, 2014b).

The framework does not consider $N_2O$ emissions; therefore the methodology presented in the remainder of this section does not consider $N_2O$ emissions.

With the rise of concentrated animal feeding operations (CAFOs), the traditional use of livestock waste as a soil amendment (where it decomposes aerobically) is not practical (Santoianni et al., 2008).[15] As a result, the use of liquid-based management systems (e.g., uncovered lagoons, pits, anaerobic digesters) that promote anaerobic conditions is increasing in popularity (EPA, 2014b). There are two general pathways for GHG emissions from anaerobic management of livestock waste:[16]

- *Uncontrolled anaerobic storage and treatment*, typically in an uncovered pit or lagoon; and
- *Anaerobic digestion*, with capture and destruction of the generated biogas.

Under uncontrolled anaerobic storage and treatment systems, livestock wastes are typically deposited as a liquid slurry in uncovered lagoons, pits, ponds, or open tanks. This storage practice results in significant indirect $CH_4$ emissions (EPA, 2009b). Volatile solids contained in livestock waste degrade under anaerobic conditions, thus generating $CH_4$ biogas. If the biogas produced in uncovered lagoons, pits, ponds, or open tanks is not collected, it is released directly to the atmosphere.

Livestock waste management using anaerobic digesters allows the generated biogas to be captured and destroyed. Anaerobic digesters used for livestock waste management range in technology from contained vessels to covered lagoons. Anaerobic digesters are designed and operated for waste stabilization resulting from the microbial reduction of complex organic compounds to $CO_2$ and $CH_4$.

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[15] Aerobic management of manure may include dry lots (including feedlots), high-rise houses for poultry production (poultry without litter), poultry production with litter, deep bedding systems for cattle and swine, manure composting, aerobic treatment units, and field spreading of manure as a soil amendment.
[16] Food waste from industrial and commercial food processing may also be managed through these approaches.

The decomposition process occurs much faster and is more complete in an anaerobic digester than in an uncontrolled anaerobic storage lagoon (Manfredi and Christensen, 2009). As a result, anaerobic digesters have higher rates of $CO_2$ and $CH_4$ generation. The increase in waste degradation and stabilization is mainly accomplished by recirculating the collected leachate within the anaerobic digester. This process enhances microbial degradation of complex organic compounds to simple organics and gaseous biogas products (primarily $CO_2$ and $CH_4$).

The vast majority of anaerobic digesters used for livestock waste treatment in the United States collect the biogas for energy use, but flaring of the gas is also practiced (EPA, 2012c, 2013b). Combustion of the biogas produced in an anaerobic digester destroys (via oxidation) most of the $CH_4$ contained in biogas; the primary resulting emission is $CO_2$.[17] The 192 anaerobic digester systems used for livestock waste management in the United States avoid an estimated 1.3 million metric tons $CO_2e$, annually compared to other livestock waste management options (EPA, 2013b). Although the use of anaerobic digesters is increasing in the United States, they are still in limited use considering the number of livestock operations, and are found primarily on large-scale livestock operations. By the end of 2011, approximately 2% of U.S. livestock operations used anaerobic digesters in waste management (EPA, 2012c).

Because anaerobic digesters are designed to enhance $CH_4$ generation, poor design, operation, or maintenance of anaerobic digesters can result in significant indirect $CH_4$ emissions. For example, $CH_4$ can leak from a digester cover or can be vented during digester start-ups, shutdowns, and malfunctions (Bogner et al., 2007; EPA, 2008b; Climate Action Reserve, 2013). However, under normal working conditions, GHG emissions from controlled biological treatment in an anaerobic digester are small relative to indirect $CH_4$ emissions from uncontrolled anaerobic storage and treatment systems (Bogner et al., 2007, and references therein). As a consequence, using anaerobic digester systems with biogas combustion typically results in substantial net GHG emissions reductions compared to conventional livestock waste storage and treatment, particularly for liquid wastes.

## 4.1. Method for Calculating an Illustrative *BAF* Value Applied to Biogenic Emissions Resulting from Anaerobic Digestion of Livestock and Food Waste

The assessment factor equation can be applied to point source biogenic $CO_2$ emissions from an anaerobic digester used to store and treat livestock or food waste.[18] This section provides an illustrative method for calculating a *BAF* value that is applied to point source biogenic $CO_2$ emissions from anaerobic digesters used to manage livestock waste.

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[17] Because biogas destruction is not 100% efficient, some $CH_4$ is released without combustion (EPA, 2009b).

[18] In concept, food waste and yard trimmings managed in an anaerobic digester can be treated similarly to livestock waste managed in an anaerobic digester. However, the focus of this section is on livestock waste. With additional data, biogenic emissions from food waste and yard trimmings could be calculated. Data needs include the total volatile content of food waste and yard trimmings (may vary within and across regions), the proportion of carbon in the volatile matter, and the maximum $CH_4$ producing capacity of food waste and yard trimmings managed in an anaerobic digester.

Here, the biogenic feedstock is biogas that is collected from an anaerobic digester. As described previously, biogas combustion, whether the biogas is flared or used as a fuel to generate energy, oxidizes the $CH_4$ contained in the biogas to $CO_2$. The destruction of $CH_4$ results in a net reduction of GHG emissions relative to a scenario in which biogas produced through the anaerobic storage and treatment of livestock waste is not captured and combusted, but instead is released to the atmosphere as an indirect emission.

Equation N.1, $BAF = AVOIDEMIT$, can be used to calculate a *BAF* value for anaerobic digestion of livestock waste. The *AVOIDEMIT* term is used to represent the net $CO_2e$ emissions reductions that are achieved through biogas capture and combustion. The *AVOIDEMIT* term accounts for net $CO_2e$ emissions reductions relative to the alternative emissions pathway of indirect $CH_4$ and $CO_2$ emissions (i.e., as a result of uncontrolled, anaerobic storage and treatment of livestock waste without biogas collection and combustion).

In practice, as applied here, the *AVOIDEMIT* term is a ratio expressed as $tCO_2e$ avoided (i.e., the emissions, in $CO_2e$, that would have occurred had the livestock waste been managed in a waste management system other than an anaerobic digester) per $tCO_2e$ emitted via combustion (i.e., the emissions, in $CO_2e$, of the combusted biogas that was generated in an anaerobic digester, after accounting for both the combustion efficiency of the biogas destruction device and any losses of biogas from the anaerobic digester[19]). For the biogas feedstock collected from anaerobic digesters, the *AVOIDEMIT* term can be conceptually expressed as:

$$\boldsymbol{AVOIDEMIT} =$$

$$\mathbf{1} - \frac{(\textbf{emissions from livestock waste management system alternative to an anaerobic digester})}{(\textbf{emissions from combustion of biogas generated in an anaerobic digester})} \qquad \textbf{(EQ N.24)}$$

### 4.1.1. Calculating the Numerator

In computing *AVOIDEMIT*, the numerator (i.e., emissions from a livestock waste management system alternative to an anaerobic digester) can be calculated by assuming that if an anaerobic digester were not used to manage livestock waste, then this waste would have been managed under a different waste management option,[20] such as an uncovered anaerobic lagoon. The $CH_4$ that would have been generated under this alternative fate can be estimated using methods presented in IPCC (2006b) and EPA (2009b). Equation N.25 can be used to estimate the annual $CO_2$ and $CH_4$ emissions

[19] Biogas losses can occur from indirect $CH_4$ emissions from an anaerobic digester could occur as a result of leaks from a digester cover or through venting during digester start-ups, shutdowns, and malfunctions. Biogas leaks may occur prior to delivery of the collected biogas to the combustion unit for $CH_4$ destruction, leaks may occur as $CH_4$ emissions from digester effluent, or as a result of remaining undigested volatile solids.

[20] There are multiple livestock waste management scenarios alternative to using an anaerobic digester. Of these alternatives, aerobic treatment would produce the least amount of $CH_4$ (zero $CH_4$ production) whereas an uncovered anaerobic lagoon would generate the most (see EPA 2009b, Table A-3). Depending on ambient temperature, $CH_4$ production in an uncovered anaerobic lagoon ranges from 66% to 80% of the maximum amount of $CH_4$ that could potentially be produced from the livestock waste. The appropriate alternative livestock waste management scenario should be used when this calculation is made.

resulting from a livestock waste management strategy other than an anaerobic digester (e.g., had the waste been managed using an uncovered anaerobic lagoon).

Total $CO_2$e emissions from a livestock waste management other than anaerobic digestion =

$$\textbf{(avoided } CO_2 \textbf{ emissions) + (avoided } CH_4 \textbf{ emissions)} \qquad \textbf{(EQ. N.25)}$$

The avoided $CO_2$ emissions from a livestock waste management alternate to anaerobic digestion is equal to the degradable carbon in the volatile solids of the livestock waste after removing the amount of carbon which becomes $CH_4$ and then converting the remaining available carbon to $CO_2$ as done using Equation N.26:

Avoided $CO_2$ emissions (metric tons $CO_2$e/year) =

$$[(P_{CO_2} \times 12/44) - (Avoided_{CH_4}/GWP_{CH_4} \times 12/16)] \times (44/12) \qquad \textbf{(EQ. N.26)}$$

Where:

$P_{CO_2}$ = Potential maximum $CO_2$ emissions if all degradable carbon is converted to $CO_2$ (metric tons $CO_2$/year), see Equation N.34.

(12/44) = molecular weight ratio of C to $CO_2$ (converts potential $CO_2$ emissions to carbon).

$Avoided_{CH_4}$ = avoided $CH_4$ emissions, metric tons $CO_2$e/year (see Equation N.33).

$GWP_{CH_4}$ = 100-year GWP for $CH_4$, 25 (IPCC, 2007).

(12/16) = molecular weight ratio of C to $CH_4$ (converts $CH_4$ emissions to carbon).

(44/12) = molecular weight ratio of $CO_2$ to C (converts C less that associated with the $CH_4$ emissions back to $CO_2$ emissions).

The most accurate data from which to estimate the total amount of carbon that can be degraded is measurement data on the biogas flow rate and methane concentration from an anaerobic digester that is already in use. If an anaerobic digester is not currently used, or if no biogas measurement data are available, then the amount of carbon that can be degraded will need to be estimated from animal population data.

#### 4.1.2. Methodology When Biogas Flow Rate and Methane Concentration Data Are Available

For each anaerobic digester, the annual flow of $CH_4$ sent to the biogas combustion device can be calculated using Equation N.27:

$$CH_4F = V \times \frac{C}{100\%} \times 0.0423 \times \frac{520^\circ\ R}{T} \times \frac{P}{1\ atm} \times \frac{0.454}{1{,}000} \qquad \textbf{(EQ. N.107)}$$

Where:

| | | |
|---|---|---|
| $CH_4F$ | = | $CH_4$ flow from the anaerobic digester to the biogas combustion device (metric tons $CH_4$/year). |
| V | = | Annual volumetric flow rate of biogas to the biogas destruction device (actual cubic feet biogas per year), as determined from daily monitoring.[21] |
| C | = | Average annual $CH_4$ concentration of biogas (percent by volume, wet basis). |
| 0.0423 | = | Density of $CH_4$ pounds per standard cubic foot (at 520ºR or 15.56ºC and 1 atm). |
| T | = | Average annual temperature (ºR) at which flow is measured.[23] |
| P | = | Average annual pressure (atm) at which flow is measured.[23] |
| 0.454/1,000 | = | conversion factor from pounds to metric tons. |

To account for the biogas collection efficiency, leaks from the anaerobic digester must be estimated (such leaks are indirect emissions to the atmosphere). Equation N.28 can be used to calculate the $CH_4$ lost (metric tons per year) from an anaerobic digester:

$$\mathbf{CH_4L = CH_4F \times \frac{(1-CE)}{CE}} \qquad \textbf{(EQ. N.28)}$$

Where:

| | | |
|---|---|---|
| $CH_4L$ | = | amount of $CH_4$ lost via leaks from the anaerobic digester, prior to combustion (metric tons $CH_4$/year). |
| $CH_4F$ | = | $CH_4$ flow from the anaerobic digester to the biogas combustion device (Equation N.27). |
| CE | = | collection efficiency[22] of the anaerobic digester. |

For each anaerobic digester, the annual flow of $CO_2$ in biogas that is sent with the $CH_4$ to the biogas combustion device can be calculated using Equation N.29:

$$\mathbf{CO_2F = V \times \left(1 - \frac{C_{CH_4}}{100\%} - \frac{M}{100\%}\right) \times 0.1160 \times \frac{520°R}{T} \times \frac{P}{1\ atm} \times \frac{0.454\ metric\ ton}{1{,}000\ lbs}} \qquad \textbf{(EQ. N.119)}$$

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[21] If the pressure or temperature fluctuates significantly during the year, it would be more accurate to calculate the annual methane flow as the sum of monthly flow volumes, corrected to standard conditions by the monthly average temperature and pressure.

[22] Biogas collection efficiency is dependent upon the type of anaerobic digester and its cover. Biogas collection efficiency for a covered anaerobic lagoon depends on the cover type: collection efficiency for a bank to bank, impermeable cover is 0.975; collection efficiency for a modular, impermeable cover is 0.70. Biogas collection efficiency is 0.99 for a complete mix, fixed film, or plug flow digester that is an enclosed vessel (EPA 2009b, Table A-4; 40 CFR 98.363, Table JJ-6). Collection efficiency is the amount of biogas flow from the digester to the combustion device divided by the total amount of biogas generated.

Where:

$CO_2F$ = $CO_2$ flow from the anaerobic digester to the biogas combustion device (metric tons $CO_2$/year).

V = Annual volumetric flow rate of biogas to the biogas destruction device (cubic feet biogas per year), as determined from daily monitoring.[23]

$(1 - C_{CH_4}/100\%\text{-}M/100\%)$ = Average annual $CO_2$ concentration of biogas (volume fraction, wet basis), where C = average annual $CH_4$ concentration of biogas (volume percent, wet basis) and M = moisture content of biogas (volume percent, wet basis).

0.1160 = Density of $CO_2$ pounds per standard cubic foot (at 520ºR or 15.74ºC and 1 atm).

T = Annual average temperature (ºR) at which flow is measured.[25]

P = Annual average pressure (atm) at which flow is measured.[25]

0.454 /1,000 = conversion factor from pounds to metric tons.

Equation N.30 can be used to calculate the $CO_2$ lost (metric tons per year) from each anaerobic digester:

$$\mathbf{CO_2L = CO_2F \times \frac{(1-CE)}{CE}} \qquad \textbf{(EQ. N.30)}$$

Where:

$CO_2L$ = amount of $CO_2$ lost via leaks from the anaerobic digester, prior to combustion (metric tons $CO_2$/year).

$CO_2F$ = $CO_2$ flow from the anaerobic digester to the biogas combustion device (Equation N.29).

CE = collection efficiency of the anaerobic digester.

Equation N.31 can be used to calculate the total $CH_4$ generation from the anaerobic digester.

$$\mathbf{Total\ CH_4\ Generation = CH_4F + CH_4L} \qquad \textbf{(EQ. N.31)}$$

Where:

[23]If the pressure or temperature fluctuates significantly during the year, it would be more accurate to calculate the annual $CO_2$ flow as the sum of monthly flow volumes, corrected to standard conditions by the monthly average temperature and pressure

Total $CH_4$ Generation = the quantity of methane generated from the anaerobic digester (metric tons $CH_4$/year).

$CH_4F$ = $CH_4$ flow from the anaerobic digester to the biogas combustion device in metric tons $CH_4$/year (Equation N.27).

$CH_4L$ = amount of $CH_4$ lost via leaks from the anaerobic digester, prior to combustion in metric tons $CH_4$/year (Equation N.28).

Equation N.32 can be used to calculate the total $CO_2$ generation from the anaerobic digester.

$$\textbf{Total CO}_\mathbf{2}\textbf{ Generation} = \mathbf{CO_2F + CO_2L} \qquad \textbf{(EQ. N.32)}$$

Where:

Total $CO_2$ Generation = the quantity of $CO_2$ generated from the anaerobic digester (metric tons $CO_2$/year).

$CO_2F$ = $CO_2$ flow from the anaerobic digester to the biogas combustion device in metric tons $CO_2$/year (Equation N.29).

$CO_2L$ = amount of $CO_2$ lost via leaks from the anaerobic digester, prior to combustion in metric tons $CO_2$/year (Equation N.30).

The $CH_4$-producing potential of a specific livestock waste management system is represented by a methane conversion factor (MCF). An anaerobic digester is expected to produce methane at near the maximum methane generation potential (i.e., at a MCF of 1). Most manure management systems will not produce the maximum amount of $CH_4$ possible because the conditions in the systems are not ideal for $CH_4$ production. The value of this parameter ranges from 0% to 100%, reflecting the capability of a system to produce the maximum achievable $CH_4$ (the higher the MCF, the greater the potential for $CH_4$ production). For liquid systems (e.g., uncovered anaerobic lagoons), MCF values are temperature dependent; in order to assign the appropriate MCF for the type of liquid system used, the average ambient temperature at the system's location must be known (see EPA, 2009b, Table A-3).

The avoided $CH_4$ emissions parameter in Equation N.25 can be estimated using Equation N.33 when $CH_4$ generation data are available from an anaerobic digester:

$$\textbf{Avoided CH}_\mathbf{4}\textbf{ Emissions} = \textbf{Total CH}_\mathbf{4}\textbf{ Generation} \times \mathbf{MCF_{WMS}} \times \mathbf{GWP}_{\boldsymbol{CH4}} \qquad \textbf{(EQ. N.123)}$$

Where:

Avoided $CH_4$ Emissions = the quantity of methane emitted, in $CO_2$ equivalence, from the alternative waste management system (metric tons $CO_2$e/year).

Total $CH_4$ Generation = the quantity of methane generated from the anaerobic digester (metric tons $CH_4$/year).

$MCF_{WMS}$ = $CH_4$ conversion factor (proportion represented as a decimal) for the alternative-scenario, waste management system (see EPA, 2009b, Table A-3).

$GWP_{CH4}$ = 100-year GWP of $CH_4$, 25 (IPCC, 2007).

The potential $CO_2$ emissions can be calculated from the total $CH_4$ and $CO_2$ generation from the anaerobic digester using Equation N.34 as follows.

$$\textbf{Potential CO}_\mathbf{2}\textbf{ Emissions} = (\textbf{Total CH}_\mathbf{4}\textbf{ Generation} \times \mathbf{44/16}) + \textbf{Total CO}_\mathbf{2}\textbf{ Generation}$$

**(EQ. N.134)**

Where:

Potential $CO_2$ emissions = maximum $CO_2$ emissions if all degradable carbon is converted to $CO_2$ (metric tons $CO_2$/year).

Total $CH_4$ Generation = the quantity of methane generated from the anaerobic digester (metric tons $CH_4$/year).

Total $CO_2$ Generation = the quantity of $CO_2$ generated from the anaerobic digester (metric tons $CO_2$/year).

44/16 = molecular weight ratio of $CO_2$ to $CH_4$ emissions.

Equation N.26 can then be used to calculate the avoided $CO_2$ emissions and Equation N.25 can be used to calculate the total $CO_2$e emissions from a livestock waste management other than anaerobic digestion.

### 4.1.3. Calculating the Denominator

The total amount of $CH_4$ and $CO_2$ emissions from an anaerobic digester, as represented in the denominator of the *AVOIDEMIT* term, is a calculation of the amount of $CH_4$ sent to the biogas destruction device, minus the amount of $CH_4$ destroyed during combustion, plus the amount of $CH_4$ leaked to the atmosphere (the latter accounts for $CH_4$ collection efficiency). To convert that calculation to $CO_2$e, it is multiplied by the GWP of $CH_4$. To account for the $CO_2$ emitted as a result of $CH_4$ combustion, the amount of $CH_4$ destroyed during combustion is added. To this is added the amount of $CO_2$ in the biogas that flows to the biogas combustion device where it is then emitted to the atmosphere. The following Equation (N.35) can be used to calculate the total amount of $CO_2$e emissions from an anaerobic digester:

$$\mathbf{CO_2e\ Emissions_{AD}} = \boldsymbol{GWP_{CH_4}}(\mathbf{CH_4F} - \mathbf{CH_4D} + \mathbf{CH_4L}) + \mathbf{CH_4D} \times \mathbf{44/16} + \mathbf{CO_2F} + \mathbf{CO_2L}$$

**(EQ. N.145)**

Where:

$CO_2e\ Emissions_{AD}$ = $CO_2$e emissions from anaerobic digestion (metric tons/year).

$GWP_{CH_4}$ = 100-year GWP of $CH_4$, 25 (IPCC, 2007).

$CH_4F$ = $CH_4$ flow to the biogas combustion device (Equation N.27).

$CH_4D$ = amount of $CH_4$ destroyed via combustion (Equation N.36).

$CH_4L$ = amount of $CH_4$ lost via leaks prior to combustion (Equation N.28).

$CO_2F$ = $CO_2$ flow to the biogas combustion device (Equation N.29).

$CO_2L$ = amount of $CO_2$ lost via leaks prior to combustion (Equation N.30).

44/16 = molecular weight ratio of $CO_2$ to $CH_4$.

Equations for most of these terms have already been presented. Equation N.36 can be used to calculate the metric tons of $CH_4$ destroyed (per year) in a biogas destruction device:

$$\mathbf{CH_4D = CH_4F \times DE} \qquad \textbf{(EQ. N.156)}$$

Where:

$CH_4D$ = $CH_4$ destroyed at a biogas combustion device (metric tons $CH_4$/year).

$CH_4F$ = $CH_4$ flow from the anaerobic digester to the biogas combustion device (Equation N.27).

DE = $CH_4$ destruction efficiency from flaring or combustion in an EGU. DE varies with the type of biogas destruction device used; it can be estimated as the lesser of the manufacturer's specified destruction efficiency and 0.99 (EPA, 2013c).

Section 4.2.1 provides an illustrative example calculation of the *AVOIDEMIT* term and its subsequent application in estimating a *BAF* for the management of livestock waste in an anaerobic digester when biogas measurement data are available.

Several parameters are presented and used in the equations in the remainder of this section. Table N-8 presents the parameters used, default values, ranges presented in the literature, and references.

**Table N-8. Summary of Parameters Used When Calculating an Illustrative *BAF* for Livestock Waste Management through Anaerobic Digestion.**

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Concentration of $CH_4$ in the biogas** | $C_{CH_4}$ | 0.55 | 0.40 to 0.60 | Fraction | | EPA, 2013c |
| **Density of $CH_4$** | – | 0.662 | – | kg $CH_4$/$m^3$ | At 532°R, or 22.22°C, and 1 atm | EPA, 2009b |

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Density of $CO_2$** | – | 0.1160 | – | Pounds per standard cubic foot | At 520ºR or 15.74ºC and 1 atm | Calculated value[3] |
| **Methane conversion factor for the specific waste management strategy** | $MCF_{WMS}$ | 0.66 | Depends on the type of system and temperature | Percent | Value presented is for an uncovered anaerobic lagoon in a cool climate below 10°C | EPA, 2009b, Table A-3 |
| **Typical animal mass, by animal type** | $TAM_{AT}$ | 604 | Numerous | kg/head | Determined using either default values or farm-specific data | EPA, 2009b, Table A-2; IPCC, 2006, Table 10A4-10A9 |
| **Volatile solids excretion rate by animal type** | $VS_{AT}$ | 9.34 | Depends on the type of animal group | kg VS/day/kg animal mass | Value presented is used in the example calculations in Section 4.0.[1] | EPA, 2009b, Table A-2; EPA, 2013c, Tables JJ-2 and JJ-3 |
| **Maximum $CH_4$-producing capacity for each animal type** | $B_0$ | 0.24 | 0.17 to 0.78 | $m^3$ $CH_4$ /kg volatile solids | Value presented is for dairy cows | EPA, 2009b, Table A-2 |
| **Destruction efficiency** | DE | 0.99 | 0.90 to 0.9977 | Fraction | 0.99 is considered a default | EPA, 2011a; EPA, 2013c |
| **Collection efficiency** | CE | 0.99 | 0.70 to 0.99 | Decimal percent | 0.99 is for an enclosed vessel, plug flow digester | EPA, 2009b, Table A-4 |
| **Fraction of volatile solids in livestock waste** | VolatileCarbon_AT | 0.2979 | 0.20 to 0.40 | kg volatile solids in total dried solids/kg of total dried solids, dry basis | Value is determined from waste volatile solids analysis[2] | Sweeten et al., 2002 |

[1] $VS_{AT}$ can be determined using either default values or farm-specific data.

[2] If only fuels proximate analysis is available, estimate the volatile solids as the sum of the volatile matter and fixed carbon from the fuels proximate analysis (see Figure N-3).
[3] This value is calculated using a 60 degree Fahrenheit conversion: 44.01 * (2.20462/836.6) = 0.1160, where 44.01 = the molecular weight of $CO_2$; 2.20462 is a unit conversion factor from kilograms to pounds; and 836.6 scf/kg-mol is the molar volume conversion factor.

#### 4.1.4. Methodology When Biogas Flow Rate and Methane Concentration Data Are Not Available

When biogas measurement data are not available, the $CH_4$ and $CO_2$ emissions must be estimated based on animal type and population data. Potential $CO_2$ emissions can be solved using the following equation:

Potential $CO_2$ emissions =

$$\boldsymbol{\Sigma_{\text{animal type}} (TVS_{AT} \times VolatileCarbon_{AT} \times (44/12) \times 365 \times 1/1000)} \qquad \textbf{(EQ. N.167)}$$

Where:

| | |
|---|---|
| $\Sigma_{\text{animal type}}$ | = If the alternate waste management system accepts waste from more than one animal type then this calculation must be computed for each animal type and then summed across animal types. |
| $TVS_{AT}$ | = Total volatile solids excreted by animal type (kg/day); see Equation N.38 to calculate $TVS_{AT}$. |
| $VolatileCarbon_{AT}$ | = Fraction of degradable carbon in the volatile solids of the livestock waste (see Equation N.39). |
| (44/12) | = molecular weight ratio of $CO_2$ to C. |
| 365 | = number of days per year (i.e., 365 days/year). |
| 1/1,000 | = conversion factor from kg to metric tons. |

Total volatile solids excreted by animal type ($TVS_{AT}$) may be calculated using Equation N.38 and by referring to tables external to this appendix (Table A-2, EPA, 2009b and Tables JJ-2 and JJ-3, 40 CFR 98.363):

$$\boldsymbol{TVS_{AT} = (Population_{AT} \times TAM_{AT} \times VS_{AT}/1000)} \qquad \textbf{(EQ. N.178)}$$

Where:

| | |
|---|---|
| $TVS_{AT}$ | = Total volatile solids excreted per animal type (kg/day). |
| $Population_{AT}$ | = Average annual animal population (head), by animal type.[24] |

[24] For static populations (e.g., dairy cows, breeding swine), average annual animal populations are estimated using

$TAM_{AT}$ = Typical animal mass, by animal type; determined using either default values (see EPA, 2009b, Table A-2) or farm specific data (kg/head).

$VS_{AT}$ = Volatile solids excretion rate by animal type, using either default values (see 40 CFR 98.363, Tables JJ-2 and JJ-3) or farm specific data (kg VS/day/kg animal mass).

The fraction of degradable carbon in the volatile solids of the livestock waste ($VolatileCarbon_{AT}$) can be estimated using results from proximate and ultimate analyses[25] of the livestock waste specific to the waste of animal type being managed (Equation N.39). Data needed to estimate this parameter can be directly measured or, more simply, can be taken from the body of published scientific literature.[26] However, it is important to understand the differences between the volatile solids measurement methods and the proximate fuel analysis methods. Figure N-3 compares the methods and nomenclature typically used for these different analytical methods.

annual animal inventory or equivalent. For growing populations (e.g., meat animals such as beef and veal cattle), average annual animal populations are estimated using the average number of days each animal is kept at the facility and the number of animals produced annually (e.g., growing population = days onsite × (number of animals produced annually / 365)).

[25] Characteristics of a biogenic fuel can be described using proximate and ultimate analyses based on a sample's complete combustion to $CO_2$ and liquid water. The proximate analysis gives moisture content, volatile content, carbon remaining (fixed carbon), and mineral ash. The ultimate analysis gives the sample's elemental composition as proportions of carbon, hydrogen, oxygen, nitrogen, and sulfur. Standardized test methods have been developed, for example, see Table 3 in Demirbas (2004).

[26] For example, ASAE Standard D384.2 (2005) is useful for estimating general characteristics of livestock and poultry manure. Li et al. (2008) and Henihan et al. (2003) present specific results of proximate and ultimate analyses of chicken litter characteristics; Sweeten et al. (2002 and 2003) present similar specific results but of cattle manure. It should be noted that Sweeten et al. (2002 and 2003) pertain to the composition of Texas feedlot beef cattle manure, values may not be suitable for calculating *BAF* values across all cattle types, regions, etc.

Diagram of Waste Analysis Methods and Nomenclature

<table>
<tr><td rowspan="4">Waste Analysis Method</td><td colspan="4">Total Solids</td></tr>
<tr><td rowspan="3">Moisture Content<br>Dry sample at 105°C to constant weight: "moisture content" is mass lost during this process, commonly expressed as ratio to mass of total solids</td><td colspan="3">Dried Solids</td></tr>
<tr><td colspan="2">Volatile Solids</td><td>Residue</td></tr>
<tr><td colspan="2">Burn in furnace with air at 500 to 550°C to constant weight: "volatile solids" = mass lost during this process, may be expressed as ratio to mass of total solids ("%VS wet basis") or dried solids ("%VS dry basis")</td><td>Mass left after burning in furnace, may be expressed as ratio to mass of total solids (wet basis) or dried solids (dry basis)</td></tr>
<tr><td rowspan="4">Proximate Fuel Analysis Method</td><td colspan="4">Total Solids</td></tr>
<tr><td rowspan="3">Moisture Content<br>Dry sample at 105°C to constant weight: "moisture content" is mass lost during this process, commonly expressed as ratio to mass of total solids</td><td colspan="3">Dried Solids</td></tr>
<tr><td>Volatile Matter</td><td>Fixed Carbon</td><td>Residue</td></tr>
<tr><td>Heat at 900°C in nitrogen for 20 minutes: "volatile matter" = mass lost during this process, commonly expressed as ratio to mass of dried solids</td><td>Burn in furnace at 600°C for about 1 hour: "fixed carbons" = mass lost during this process, commonly expressed as ratio to mass of dried solids</td><td>Mass left after burning in furnace, commonly expressed as ratio to mass of dried solids</td></tr>
</table>

**Figure N-3. Comparison of Waste and Proximate Fuel Analyses (Adapted from ASTM, 2013).**

The $\text{VolatileCarbon}_{\text{AT}}$ term represents the amount of carbon in the livestock waste solids that degrades during livestock waste management. Assuming there is negligible carbon remaining in the residue and that the fixed carbon is not readily biodegradable, $\text{VolatileCarbon}_{\text{AT}}$ term can be estimated as follows:

$$\textbf{VolatileCarbon}_{\textbf{AT}} = \frac{\textbf{Carbon} - \textbf{Fixed Carbon}}{\textbf{Volatile Solids}} = \frac{\textbf{Carbon} - \textbf{Fixed Carbon}}{\textbf{Volatile Matter} + \textbf{Fixed Carbon}} \qquad \textbf{(EQ. N.189)}$$

Where:

$\text{VolatileCarbon}_{\text{AT}}$ = fraction of the degradable carbon in the volatile solids of the livestock waste (kg degradable carbon in volatile solids/kg of volatile solids, dry basis).[27]

Carbon = fraction of carbon in livestock waste (kg carbon in total dried solids/kg of total dried solids), dry basis (from ultimate analysis).

Fixed Carbon = fraction of dry solids in livestock waste that does not volatilize when heated to 900 °C in nitrogen (kg fixed carbon in total dried solids/kg of total dried solids) but is lost when heated in air at 600 °C, dry basis (from fuels proximate analysis; see Figure N-3).

[27] If the mass of fixed carbon is not 100% carbon then the amount of carbon in the volatile matter may be underestimated, thus giving a low-biased estimate of the $\text{VolatileCarbon}_{\text{AT}}$ term (though the bias is likely small).

Volatile Solids = fraction of volatile solids in livestock waste (kg volatile solids in total dried solids/kg of total dried solids), dry basis (from waste volatile solids analysis). If only fuels proximate analysis is available, estimate the volatile solids as the sum of the volatile matter and fixed carbon from the fuels proximate analysis (see Figure N-3).

Volatile Matter = fraction of dry solids that does is lost when heated to 900 °C in nitrogen (kg volatile matter/kg of total dried solids), dry basis.

The avoided $CH_4$ emissions parameter in Equation N.25 can be populated using Equation N.40, which estimates the annual avoided $CH_4$ emissions generated from a manure management strategy alternate to anaerobic digestion:

Avoided $CH_4$ emissions (metric tons $CO_2$e/year) =

$$\boldsymbol{\Sigma_{\text{animal type}} (TVS_{AT} \times VS_{WMS} \times Days \times B_0 \times MCF_{WMS} \times 0.662 \times 1/1000 \times GWP_{CH_4})} \quad \textbf{(EQ. N.40)}$$

Where:

$\Sigma_{\text{animal type}}$ = If the alternate waste management system accepts waste from more than one animal type then this calculation must be computed for each animal type and then summed across animal types.

$TVS_{AT}$ = Total volatile solids excreted by animal type (kg/day); the $TVS_{AT}$ equation is presented above (Equation N.38).

$VS_{WMS}$ = Proportion of total manure for each animal type that is managed in each waste management system (assumed to be equivalent to the amount of volatile solids in each waste management system).

Days = Number of days per year (i.e., 365 days/year).

$B_0$ = Maximum $CH_4$-producing capacity for each animal type ($m^3$ $CH_4$/kg volatile solids; see EPA, 2009b, Table A-2).

$MCF_{WMS}$ = $CH_4$ conversion factor (proportion represented as a decimal) for the alternative-scenario, waste management system (see EPA, 2009b, Table A-3).

0.662 = density of $CH_4$, kg $CH_4$/$m^3$ (at 532°R, or 22.22°C, and 1 atm).

1/1000 = conversion factor from kg to metric tons.

$GWP_{CH_4}$ = 100-year GWP of $CH_4$, 25 (IPCC, 2007).

The maximum amount of $CH_4$ that could potentially be produced from livestock waste managed under ideal conditions is calculated by multiplying the total volatile solids by the maximum $CH_4$-producing capacity of the livestock waste ($B_0$). The $B_0$ values vary by animal type and diet (see EPA, 2009b, Table A-2). Most manure management systems will not produce the maximum amount of $CH_4$ possible because the conditions in the systems are not ideal for $CH_4$ production. The $CH_4$-producing potential of a specific livestock waste management system is represented by a methane conversion factor (MCF). The value of this parameter ranges from 0% to 100% and reflects the capability of a system to produce the maximum achievable $CH_4$ (the higher the MCF, the greater the potential for $CH_4$ production). For liquid systems (e.g., uncovered anaerobic lagoons), MCF values are temperature dependent: in order to assign the appropriate MCF for the type of liquid system used, the average ambient temperature at the system's location must be known (see EPA, 2009b, Table A-3).

Summing the avoided $CO_2$ emissions and the avoided $CH_4$ emissions (Equation N.25, metric tons $CO_2$e/year) is the final computation in estimating the numerator in the *AVOIDEMIT* term for a livestock waste management strategy alternative to an anaerobic digester.

As before, the denominator of the *AVOIDEMIT* emission ratio term (i.e., the emissions from combustion of biogas generated in an anaerobic digester) is based on the flow, loss, and destruction terms presented previously (see Equation N.35). However, in the lack of direct biogas measurement data, these terms must be estimated based on the animal population equations just presented.

As noted previously, the maximum amount of $CH_4$ that could potentially be produced from livestock waste managed under ideal conditions is calculated by multiplying the total volatile solids by the maximum $CH_4$-producing capacity of the livestock waste ($B_0$). Thus, total $CH_4$ generation can be estimated using Equation N.41, which is similar to Equation N.40 except that $MCF_{WMS}$ is assumed to equal 1 and the $CH_4$ emissions are not converted to $CO_2$ equivalence.

Total $CH_4$ generation (tons $CH_4$/yr) =

$$\boldsymbol{\Sigma_{\text{animal type}} (TVS_{AT} \times VS_{WMS} \times Days \times B_0 \times 0.662 \times 1/1000)} \qquad \textbf{(EQ. N.41)}$$

Where:

$\Sigma_{\text{animal type}}$ = If the alternate waste management system accepts waste from more than one animal type then this calculation must be computed for each animal type and then summed across animal types.

$TVS_{AT}$ = Total volatile solids excreted by animal type (kg/day); the $TVS_{AT}$ equation is presented above (Equation N.38).

$VS_{WMS}$ = Proportion of total manure for each animal type that is managed in each waste management system (assumed to be equivalent to the amount of volatile solids in each waste management system).

Days = Number of days per year (i.e., 365 days/year).

$B_0$ = Maximum $CH_4$-producing capacity for each animal type ($m^3$ $CH_4$/kg volatile solids; see EPA, 2009b, Table A-2).

0.662 = density of $CH_4$, kg $CH_4/m^3$ (at 532°R, or 22.22°C, and 1 atm).

1/1000 = conversion factor from kg to metric tons.

$CH_4$ sent to the biogas destruction device can be calculated based on the collection efficiency of the anaerobic digester using Equation N.42:

$$\mathbf{CH_4F = Total\ CH_4\ Generation \times CE} \qquad \textbf{(EQ. N.42)}$$

Where:

| | |
|---|---|
| $CH_4F$ | = $CH_4$ flow from the anaerobic digester to the biogas combustion device (tons $CH_4$/yr). |
| Total $CH_4$ generation | = total quantity of $CH_4$ generated in the anaerobic digester (tons $CH_4$/yr; see Equation N.41). |
| CE | = collection efficiency[28] of the anaerobic digester. |

The $CH_4$ that is not collected and sent to the biogas destruction device can be calculated by rearranging Equation N.28:

$$\mathbf{CH_4L = Total\ CH_4\ Generation - CH_4F} \qquad \textbf{(EQ. N.193)}$$

Where:

| | |
|---|---|
| $CH_4L$ | = amount of $CH_4$ lost via leaks from the anaerobic digester, prior to combustion (metric tons $CH_4$/year). |
| Total $CH_4$ generation | = total quantity of $CH_4$ generated in the anaerobic digester (tons $CH_4$/yr; see Equation N.31) |
| $CH_4F$ | = $CH_4$ flow from the anaerobic digester to the biogas combustion device (Equation N.42). |

Given the potential $CO_2$ emissions from Equation 37 and re-arranging Equation N.34 yields the following equation for estimating the total $CO_2$ generation from the anaerobic digester.

$$\mathbf{Total\ CO_2\ Generation = Potential\ CO_2\ Emissions - (Total\ CH_4\ Generation \times 44/16)} \qquad \textbf{(EQ. N.204)}$$

Where:

[28] Biogas collection efficiency is dependent upon the type of anaerobic digester and its cover. Biogas collection efficiency for a covered anaerobic lagoon depends on the cover type: collection efficiency for a bank to bank, impermeable cover is 0.975; collection efficiency for a modular, impermeable cover is 0.70. Biogas collection efficiency is 0.99 for a complete mix, fixed film, or plug flow digester that is an enclosed vessel (EPA 2009b, Table A-4; 40 CFR 98.363, Table JJ-6). Collection efficiency is the amount of biogas flow from the digester to the combustion device divided by the total amount of biogas generated.

Total $CO_2$ Generation = the quantity of $CO_2$ generated from the anaerobic digester (metric tons $CO_2$/year).

Potential $CO_2$ emissions = maximum $CO_2$ emissions if all degradable carbon is converted to $CO_2$ (metric tons $CO_2$/year from equation N.34).

Total $CH_4$ Generation = the quantity of methane generated from the anaerobic digester (metric tons $CH_4$/year; Equation N.41).

44/16 = molecular weight ratio of $CO_2$ to $CH_4$ emissions.

Similar to the $CH_4$ flow and loss terms, the $CO_2$ flow and loss terms can be calculated based on the collection efficiency of the anaerobic digester as follows:

$$\mathbf{CO_2F = Total\ CO_2\ Generation \times CE} \qquad \textbf{(EQ. N.215)}$$

$$\mathbf{CO_2L = Total\ CO_2\ Generation - CO_2F} \qquad \textbf{(EQ. N.226)}$$

Where:

$CO_2F$ = $CO_2$ flow from the anaerobic digester to the biogas combustion device (tons $CO_2$/yr).

Total $CO_2$ generation = total quantity of $CO_2$ generated in the anaerobic digester (metric tons $CO_2$ /year; see Equation N.44).

CE = collection efficiency[29] of the anaerobic digester.

$CO_2L$ = amount of $CO_2$ lost via leaks from the anaerobic digester, prior to combustion (metric tons $CO_2$/year).

Equation N.36, presented previously, can be used to calculate the metric tons of $CH_4$ destroyed (per year) in a biogas destruction device. All of the parameters needed to determine the $CO_2e$ emissions from the anaerobic digester using Equation N.35 are then available.

Section 4.2.2 provides an illustrative example calculation of the *AVOIDEMIT* term and its subsequent application in estimating a *BAF* for the management of livestock waste in an anaerobic digester prior to the availability of biogas measurement data

[29] Biogas collection efficiency is dependent upon the type of anaerobic digester and its cover. Biogas collection efficiency for a covered anaerobic lagoon depends on the cover type: collection efficiency for a bank to bank, impermeable cover is 0.975; collection efficiency for a modular, impermeable cover is 0.70. Biogas collection efficiency is 0.99 for a complete mix, fixed film, or plug flow digester that is an enclosed vessel (EPA 2009b, Table A-4; 40 CFR 98.363, Table JJ-6). Collection efficiency is the amount of biogas flow from the digester to the combustion device divided by the total amount of biogas generated.

## 4.2. Illustrative *AVOIDEMIT* and *BAF* Calculations for Livestock Waste Management

### 4.2.1. Illustrative Calculations when Anaerobic Digester Measurement Data Are Available

When anaerobic digester flow and concentration data are available, these data provide a more accurate estimate of the degradable carbon quantities in the livestock wasted. Therefore, the potential $CH_4$ and $CO_2$ emissions from the alternate treatment pathway should be estimated from these measurement data rather than from animal population data. Again, this example is for a dairy farm in a cool climate (average ambient temperature below 10°C) and is comparing the alternate treatment of the livestock waste in an uncovered anaerobic lagoon to an anaerobic digester.

In this hypothetical example, the daily average volumetric biogas flow rate from the anaerobic digester is 54,500 $ft^3$ per day and the annual volumetric flow volume is 19,892,500 (e.g., 54,500 $ft^3$/day × 365 days/year). In this hypothetical example, the average annual $CH_4$ concentration of biogas was measured to be 52.1% (wet basis). Based on daily monitoring, the annual average temperature of the biogas from the anaerobic digester was 77ºF (537ºR) at 1.005 atm, both of which were measured where the flow is measured.

#### Step 1: Calculate the $CH_4$ Emissions from the Anaerobic Digester

The amount (metric tons/year) of $CH_4$ sent from the anaerobic digester to the biogas combustion device, $CH_4F$, is calculated as (Equation N.42):

$$\mathbf{CH_4F = V \times \frac{C_{\mathit{CH4}}}{100} \times 0.0423 \times \frac{520}{T} \times \frac{P}{1\ atm} \times \frac{0.454}{1,000}}$$

$$\mathbf{CH_4F = 19,892,500 \times 0.521 \times 0.0423 \times (520/537) \times (1.005/1) \times (0.454/1000)}$$

$$\mathbf{= 193.6950\ metric\ tons\ CH_4/year.}$$

The next term to solve in calculating the denominator of the *AVOIDEMIT* term is $CH_4D$, the amount of $CH_4$ destroyed at a biogas combustion device (metric tons $CH_4$/year). This can be calculated using Equation N.36:

$$\mathbf{CH_4D = CH_4F \times DE}$$

The $CH_4$ DE is estimated as the lesser of the manufacturer's specified DE and 0.99 (EPA, 2013c). A DE value of 0.99 will be used for this example. Thus, the $CH_4$ destroyed at biogas combustion device can be estimated as:

$$\mathbf{CH_4D = 193.6950 \times 0.99}$$

$$\mathbf{= 191.7581\ MT\ CH_4/year.}$$

The next term in the denominator of the *AVOIDEMIT* term, $CH_4L$, accounts for the CE. To calculate $CH_4$ leakage (metric tons $CH_4$/year) from an anaerobic digester, Equation N.28 can be used:

$$\mathbf{CH_4L = \ CH_4F \times \frac{(1-CE)}{CE}}$$

The previously computed value for $CH_4F$ can be used in this example. The CE is dependent upon the type of anaerobic digester and its cover (for default values, see EPA, 2009b, Table A-4; 40 CFR 98.363, Table JJ-6). This hypothetical example is for an enclosed vessel, mixed plug flow digester where the CE is 0.99, such that:

**$CH_4L$ = 193.6950 × (1 – 0.99) / 0.99**

**= 1.9565 metric tons $CH_4$/year.**

### Step 2: Calculate the $CO_2$ Emissions from the Anaerobic Digester

The next parameter to solve for in the denominator of the *AVOIDEMIT* term is $CO_2F$, the annual flow of $CO_2$ in biogas sent (mixed with $CH_4$) to the biogas combustion device. This can be calculated using Equation N.29:

$$\mathbf{CO_2F = V \times \left[1 - \left(\frac{C_{CH_4}}{100\%}\right) - \left(\frac{M}{100\%}\right)\right] \times 0.1160 \times \frac{520}{T} \times \frac{P}{1\ atm} \times \frac{0.454}{1,000}}$$

The values for V, C, T, and P are the same when solving for $CO_2F$ (Equation N.29) as for when solving for $CH_4F$ (Equation N.27). The only change is in the density of the gas, from $CH_4$ (0.0423 lbs/ft$^3$) to that of $CO_2$ (0.1160 lbs/ft$^3$) and that Equation N.29 incorporates the fraction of biogas that is not $CH_4$ or water vapor (i.e., 1 – $C_{CH_4}$/100%-M/100%). As most anaerobic digesters operate at temperature above 30°C (above 86°F), it can be assumed the cooled biogas (at the flow measurement point) is saturated with water (relative humidity of 100%). Using a psychometric chart, 77°F air holds approximately 20 grams water per kg dry air. Using the molecular weight of 18 g/mol for water and 29 g/mol for air, the moisture content of the biogas is estimated to be 3.1% (i.e., (20/18)/[(1000/29)+(20/18)]). Using Equation N.29, $CO_2F$ can be estimated as:

**$CO_2F$ = 19,892,500 × (1 – 0.521-0.031) × 0.1160 × 520/537 × 1.005 × 0.454/1,000**

**= 459.1102 MT $CO_2$/year**

The final parameter to solve for in the denominator of the *AVOIDEMIT* term is $CO_2L$, the annual flow of $CO_2$ in biogas lost from the digester. This can be calculated using Equation N.30, using the value of $CO_2F$ just calculated and the gas collection efficiency (0.99; same as used to determine $CH_4L$), as follows:

$$\mathbf{CO_2L = \ CO_2F \times \frac{(1 - CE)}{CE}}$$

**$CO_2$ = 459.1102 × (1 – 0.99) / 0.99**

**= 4.6375 MT $CO_2$/year.**

### Step 3: Calculate the $CO_2e$ Emissions from the Anaerobic Digester (Denominator)

With estimated values for each of the terms in the denominator of the *AVOIDEMIT* term, the denominator can be solved using Equation N.35:

$$\mathbf{CO_2e\ Emissions_{AD} = 25(CH_4F - CH_4D + CH_4L) + CH_4D \times 44/16 + CO_2F + CO_2L}$$

$CO_2e$ emissions from the anaerobic digester (metric tons /year)

**= 25 × (193.6950 – 191.7581 + 1.9565) + 191.7581 × 44/16 + 459.1102+4.6375**

**= 1,088.4175 metric tons $CO_2e$ /year**

### Step 4: Calculate the $CO_2e$ Emissions from the Alternate Fate (Numerator)

Now that the terms for the denominator are determined, these values can be used to estimate the total $CH_4$ and $CO_2$ generation from the alternate treatment fate (uncovered lagoon) using Equations N.31 and N.32.

**Total $CH_4$ generation (MT $CH_4$/year) = $CH_4F$ + $CH_4L$**

**Total $CH_4$ generation = 193.6950 + 1.9565**

**= 195.6515**

**Total $CO_2$ generation = $CO_2F$ + $CO_2L$**

**Total $CO_2$ generation = 459.1102 + 4.6375**

**= 463.7477**

For an anaerobic digester, the $CH_4$ conversion factor, MCF, is assumed to be 1; for the alternative-scenario's waste management system, the $CH_4$ conversion factor is projected to be 0.66 (see EPA, 2009b, Table A-3; for uncovered anaerobic lagoon in a cool climate below 10°C,[30] $MCF_{WMS}$ = 0.66). Applying Equation N.33, the avoided $CH_4$ emissions:

Avoided $CH_4$ emissions (metric tons/year) = Total $CH_4$ generation × $MCF_{WMS}$ × $GWP_{CH4}$

**= 195.6515 × 0.66 × 25**

**= 3,228.2498 MT $CO_2e$ per year.**

The potential $CO_2$ emissions are calculated from the total $CH_4$ and $CO_2$ generation from the anaerobic digester as:

**Potential $CO_2$ emissions = (Total $CH_4$ generation × 44/16) + Total $CO_2$ generation**

---

[30] Table A-3 in EPA 2009b assigns $CH_4$ conversion factors based on ambient temperature, thus accounting for the influence of climate on $CH_4$ production.

**Potential $CO_2$ emissions = (195.6515 × 44/16) +463.7477**

**= 1,001.7893 MT $CO_2$ per year**

With estimates of both the potential $CO_2$ emissions and the avoided $CH_4$ emissions, the avoided $CO_2$ emissions resulting from a dairy manure management strategy other than an anaerobic digester can be calculated using Equation N.26 as follows.

**Avoided $CO_2$ emissions = ((Potential $CO_2$ emissions × 12/44) – (Avoided $CH_4$ emissions/25 × 12/16)) × (44/12)**

**= ((1,001.7893 × 12/44) – (3,228.2498/25 × 12/16)) × (44/12)**

**= 646.6818 MT $CO_2$ per year**

Summing the estimated, avoided $CH_4$ and $CO_2$ emissions (both in metric tons $CO_2e$ per year), that result from a dairy manure management strategy other than an anaerobic digester, the total avoided $CO_2e$ emissions can be estimated in units of metric tons $CO_2e$ per year, thus solving the numerator of the *AVOIDEMIT* term (Equation N.24):

Total avoided $CO_2e$ emissions

**= (avoided $CO_2$ emissions) + (avoided $CH_4$ emissions)**

**= 646.6818 + 3,228.2498**

**= 3,874.9317 MT $CO_2e$ per year**

### Step 5: Calculate the *BAF* Value

With both the numerator and denominator of the *AVOIDEMIT* term having been computed, a *BAF* value can be estimated using Equations N.1 and N.2:

$$\boldsymbol{BAF = AVOIDEMIT}$$

$$\boldsymbol{BAF = 1 - \frac{3{,}874.9317}{1{,}088.4175}}$$

$$\boldsymbol{BAF = -2.56}$$

A negative *BAF* value calculated for this hypothetical scenario indicates that a biogas feedstock produced in an anaerobic digester from the treatment of dairy manure and flared by a stationary source results in net $CO_2e$ emissions reductions.

#### 4.2.2. Example Calculations for Livestock Waste Management Prior to Installation of an Anaerobic Digester (When Measurement Data are Not Available)

Prior to the installation of an anaerobic digester, the only information available to determine the carbon content of the livestock waste and to project the methane generation potential of the anaerobic digester are the equations presented correlating the potential $CO_2$ emissions and the avoided methane emissions to animal type and population. This example illustrates how to determine the assessment factor based only on the animal population data.

### Step 1: Calculate the $CO_2$e Emissions from the Alternate Fate (Numerator)

To calculate the numerator of the *AVOIDEMIT* term, the total volatile solids in the managed livestock waste must be estimated in order to calculate the avoided $CO_2$ and $CH_4$ emissions. Parameters in equation N.25 can be estimated using a hypothetical example of a dairy farm in a cool climate (average ambient temperature below 10°C) consisting of 500 dairy cows with a typical animal mass of 604 kg, and a volatile solids excretion rate of 9.34 kgVS/day/1,000 kg animal mass (see 40 CFR 98.363, Tables JJ-2 and JJ-3). Equation N.38 may be used to calculate total volatile solids excreted per animal type:

$$\mathbf{TVS_{AT} = (Population_{AT} \times TAM_{AT} \times VS_{AT}/1{,}000)}$$

$$\mathbf{= 500 \times 604 \times 9.34 \;/\; 1{,}000}$$

$$\mathbf{= 2{,}820.68\ kg/day}$$

With $TVS_{AT}$ estimated, the avoided $CH_4$ emissions resulting from a dairy manure management strategy other than an anaerobic digester can be calculated. In this hypothetical example, the alternative fate evaluated is management of the manure using an uncovered anaerobic lagoon. When calculating the avoided $CH_4$ emissions associated with the alternative fate of this scenario's dairy manure, several parameters are needed, including the maximum $CH_4$-producing capacity for dairy cattle (see EPA, 2009b, Table A-2, for the appropriate default value; for dairy cows, $B_0$ = 0.24 $m^3$ $CH_4$/kg); and a $CH_4$ conversion factor for the alternative-scenario's waste management system (see EPA, 2009b, Table A-3; for uncovered anaerobic lagoon in a cool climate below 10°C,[31] $MCF_{WMS}$ = 0.66). Equation N.40 can be used to calculate the avoided $CH_4$ emissions:

Avoided $CH_4$ emissions (metric tons/year)

$$\mathbf{= (TVS_{AT} \times VS_{WMS} \times Days \times B_0 \times MCF_{WMS} \times 0.662 \times 1\;/1{,}000 \times 25)}$$

$$\mathbf{= (2{,}820.68\ kg/day \times 1 \times 365\ days/year \times 0.24\ m^3\ CH_4/kg \times 0.66 \times 0.662\ kg\ CH_4/m^3 \times 1\ metric\ ton/1{,}000\ kg \times 25)}$$

$$\mathbf{= 2{,}698.9812\ metric\ tons\ CO_2e\ per\ year}$$

In order to estimate avoided $CO_2$ emissions from a dairy manure management strategy other than an anaerobic digester, the proportion of the carbon in the volatile matter of the dairy manure ($VolatileCarbon_{AT}$) must be calculated. In the $VolatileCarbon_{AT}$ term, the fixed carbon is removed from the total carbon in the dairy manure because the fixed carbon is assumed not to degrade. Data

[31] Table A-3 in EPA 2009b assigns $CH_4$ conversion factors based on ambient temperature, thus accounting for the influence of climate on $CH_4$ production.

results from proximate and ultimate analyses of cattle manure (Sweeten et al., 2002)[32] were used to populate parameter values in Equation N.39:

$$\mathbf{VolatileCarbon_{AT}} = \frac{\text{Carbon–Fixed Carbon}}{\text{Volatile Solids}} = \frac{\text{Carbon–Fixed Carbon}}{\text{Volatile Matter + Fixed Carbon}}$$

$$= \frac{\mathbf{0.2959 - 0.1127}}{\mathbf{0.5022 + 0.117}}$$

**= 0.2979**

With $TVS_{AT}$ and $VolatileCarbon_{AT}$ estimated, the potential maximum $CO_2$ emissions (Equation N.25) resulting from a dairy manure management strategy other than an anaerobic digester can then be estimated. In this scenario, the alternate fate is manure management using an open anaerobic lagoon. Because there is only one animal type (dairy cows), Equation N.37 only needs to be calculated once, as follows:

Potential $CO_2$ emissions = ($TVS_{AT}$ × $VolatileCarbon_{AT}$ × (44/12) × 365 × 1 /1000)

**= 2,820.68 × 0.2979 × (44/12) × 365 × 1 /1,000**

**= 1,124.5755 MT $CO_2$ per year**

With estimates of both the potential $CO_2$ emissions and the avoided $CH_4$ emissions, the avoided $CO_2$ emissions resulting from a dairy manure management strategy other than an anaerobic digester can be calculated. For this calculation the total potential $CO_2$ emissions from the decomposition of $TVS_{AT}$ must first be converted to carbon. From this the $CH_4$ emissions, converted to carbon must be subtracted. The result is then converted back to $CO_2$ (see Equation N.26).

Avoided $CO_2$ emissions = ((Potential $CO_2$ emissions × 12/44) – (Avoided $CH_4$ emissions/25 × 12/16)) × (44/12)

**= ((1,124.575 × 12/44) – (2,698.9812/25 × 12/16)) × (44/12)**

**= 827.6877 MT $CO_2$ per year**

Summing the estimated, avoided $CH_4$ and $CO_2$ emissions (both in metric tons $CO_2e$ per year), that result from a dairy manure management strategy other than an anaerobic digester, the total avoided $CO_2e$ emissions can be estimated in units of metric tons $CO_2e$ per year, thus solving the numerator of the *AVOIDEMIT* term (Equation N.25):

Total avoided $CO_2e$ emissions

**= (avoided $CO_2$ emissions) + (avoided $CH_4$ emissions)**

[32] Data specific to Wisconsin dairy cattle would be preferred but in the absence of these data, data from Sweeten et al. (2002) were applied.

**= 827.6877 + 2,698.9812**

**= 3,526.6689 MT $CO_2$e per year**

### Step 2: Calculate the $CO_2$e Emissions from the Actual Fate (Denominator)

In order to calculate the emissions from the anaerobic digester (i.e., denominator of the *AVOIDEMIT* term), these same equations can be used to calculate the projected emissions from the total $CH_4$ and $CO_2$ generation.

The total $CH_4$ generation for the anaerobic digester is calculated using Equation N.41.

Total $CH_4$ generation (metric tons $CH_4$/year)

**= ($TVS_{AT}$ × $VS_{WMS}$ × Days × $B_0$ × 0.662 kg $CH_4$/$m^3$ × 1 metric ton/1,000 kg)**

**= (2,820.68 kg/day × 1 × 365 days/year × 0.24 $m^3$ $CH_4$/kg × 0.662 kg $CH_4$/$m^3$ × 1 metric ton/1,000 kg)**

**= 163.5746 MT $CH_4$/year.**

The potential $CO_2$ emissions have already been calculated (1,124.5755 metric tons $CO_2$/year). However, the methane produced will lower the $CO_2$ emissions from the anaerobic digester. The total $CO_2$ generation is calculated by subtracting the carbon associated the $CH_4$ generation from the potential $CO_2$ emissions using Equation N.44 as follows:

Total $CO_2$ generation (metric tons $CO_2$/year)

**= ((Potential $CO_2$ emissions) – (Total $CH_4$ generation × 44/16))**

**= ((1,124.5755 × 12/44) – (163.5746 × 12/16)) × (44/12)**

**= 674.7453 MT $CO_2$ per year**

The actual flow and loss terms can be computed from the capture efficiency of the anaerobic digester using Equations N.42, N.43, N.45, and N.46. The collection efficiency is dependent upon the type of anaerobic digester and its cover (for default values, see EPA, 2009b, Table A-4; 40 CFR 98.363, Table JJ-6). This hypothetical example is for an enclosed vessel, mixed plug flow digester where the collection efficiency is 0.99, such that:

| | | | |
|---|---|---|---|
| **$CH_4$_F** | **= Total $CH_4$ generation × CE** | **= 163.5746×0.99** | **= 161.9389** |
| **$CH_4$_L** | **= Total $CH_4$ generation − $CH_4$_F** | **= 163.5746 − 161.9389** | **= 1.6357** |
| **$CO_2$F** | **= Total $CO_2$ generation × CE** | **= 674.7453×0.99** | **= 667.9978** |
| **$CO_2$L** | **= Total $CO_2$ generation − $CO_2$F** | **= 674.7453 − 667.9978** | **= 6.7475** |

The next term to solve in calculating the denominator of the *AVOIDEMIT* term is $CH_4D$, the amount of $CH_4$ destroyed at a biogas combustion device (MT $CH_4$/year). This can be calculated using Equation N.36:

$$\mathbf{CH_4D = CH_4F \times DE}$$

$CH_4F$ was previously solved. The $CH_4$ DE is estimated as the lesser of the manufacturer's specified destruction efficiency and 0.99 (EPA, 2013c). A DE value of 0.99 will be used for this example. Thus, the $CH_4$ destroyed at biogas combustion device can be estimated as:

$$\mathbf{CH_4D = 161.9389 \times 0.99}$$

$$\mathbf{= 160.3195\ MT\ CH_4/year.}$$

With estimated values for each of the terms in the denominator of the *AVOIDEMIT* term, the denominator can be solved using Equation N.35:

$$\mathbf{CO_2e\ Emissions_{AD} = 25(CH_4F - CH_4D + CH_4L) + CH_4D \times 44/16 + CO_2F + CO_2L}$$

$CO_2e$ emissions from the anaerobic digester (MT /year)

$$\mathbf{= 25 \times (161.9389 - 160.3195 + 1.6357) + 160.3195 \times 44/16 + 667.9978 + 6.7475}$$

$$\mathbf{= 1{,}197.0014\ MT\ CO_2e\ /year}$$

### Step 3: Calculate the *BAF* Value

With both the numerator and denominator of the *AVOIDEMIT* term having been computed, a *BAF* value can be estimated using Equations N.1 and N.2:

$$\boldsymbol{BAF = AVOIDEMIT}$$

$$\boldsymbol{BAF} = \mathbf{1 - \frac{3{,}526.6689}{1{,}197.0014}}$$

$$\boldsymbol{BAF} = \mathbf{-1.95}$$

A negative *BAF* value calculated for this hypothetical scenario indicates that a biogas feedstock produced in an anaerobic digester from the treatment of dairy manure and flared by a stationary source results in net $CO_2e$ emissions reductions.

## 4.3. Sensitivity Analysis for Anaerobic Digestion of Livestock and Food Waste

For the livestock and food waste assessments, the parameter that has the greatest variability is the methane correction factor (MCF) for the waste management system employed as an alternative to anaerobic digestion. The biogas collection efficiency and the biogas destruction efficiency impact the *BAF* value, as does the global warming potential of $CH_4$. The total biogas flow rate or the total volatile solids produced (if animal population correlations are used) does not impact the *BAF* value as they impact both the numerator and denominator by a constant factor. However, the relative ratio of $CH_4$ produced versus $CO_2$ produced does impact the *BAF* value. Table N-9 presents the results of the sensitivity analysis performed when using biogas measurement data to calculate BA.

Table N-10 presents the results of the sensitivity analysis performed when using animal population data to calculate *BAF*. Sources for the parameter values used here can be found in Table N-8 of Section 4.1.3.

**Table N-9. Sensitivity Analysis of Anaerobic Digestion Using Biogas Measurement Data.[1]**

| Analysis | Key Parameter Varied | Key Parameter Value | *BAF* Value Calculated at the Specified MCF Value | | | |
|---|---|---|---|---|---|---|
| | | | MCF=0.05 | MCF=0.3 | MCF=0.5 | MCF=0.8 |
| **1** | $GWP_{CH_4}$ | 21 | −0.10 | −0.90 | −1.54 | −2.50 |
| **2** | $GWP_{CH_4}$ | 25 | −0.12 | −1.08 | −1.85 | −3.00 |
| **3** | $GWP_{CH_4}$ | 28 | −0.13 | −1.21 | −2.07 | −3.37 |
| **4** | $C_{CH_4}$ | 40% | −0.09 | −0.88 | −1.50 | −2.44 |
| **5** | $C_{CH_4}$ | 50% | −0.12 | −1.08 | −1.85 | −3.00 |
| **6** | $C_{CH_4}$ | 60% | −0.14 | −1.27 | −2.18 | −3.55 |
| **7** | DE | 0.99 | −0.12 | −1.08 | −1.85 | −3.00 |
| **8** | DE | 0.95 | 0.03 | −0.80 | −1.47 | −2.47 |
| **9** | CE | 0.99 | −0.12 | −1.08 | −1.85 | −3.00 |
| **10** | CE | 0.70 | 0.47 | 0.01 | −0.35 | −0.90 |

[1] The following central tendency values were used unless specified as the parameter varied. The moisture content was not varied as it has a limited range (1% to 5%) and did not significantly impact the calculated *BAF*.

- $GWP_{CH_4}$ = 25
- $C_{CH_4}$ = 50%
- M (moisture content) = 3%
- DE = 0.99
- CE = 0.99

Note: References for the key parameters and values are presented in Table N-8 of Section 4.1.3.

**Table N-10. Sensitivity Analysis of Anaerobic Digestion Using Animal Population Data.[1]**

| Analysis | Key Parameter Varied | Key Parameter Value | *BAF* Value Calculated at the Specified MCF Value | | | |
|---|---|---|---|---|---|---|
| | | | MCF=0.05 | MCF=0.3 | MCF=0.5 | MCF=0.8 |
| **1** | $GWP_{CH_4}$ | 21 | −0.09 | −0.87 | −1.48 | −2.41 |
| **2** | $GWP_{CH_4}$ | 25 | −0.11 | −1.04 | −1.79 | −2.90 |
| **3** | $GWP_{CH_4}$ | 28 | −0.13 | −1.17 | −2.01 | −3.26 |
| **4** | Bo | 0.15 | −0.06 | −0.54 | −0.93 | −1.51 |
| **5** | Bo | 0.30 | −0.11 | −1.04 | −1.79 | −2.90 |
| **6** | Bo | 0.50 | −0.18 | −1.65 | −2.84 | −4.61 |
| **7** | VolatileCarbon_AT | 0.20 | −0.16 | −1.51 | −2.58 | −4.20 |
| **8** | VolatileCarbon_AT | 0.30 | −0.11 | −1.04 | −1.79 | −2.90 |
| **9** | VolatileCarbon_AT | 0.40 | −0.09 | −0.80 | −1.36 | −2.22 |
| **10** | DE | 0.99 | −0.11 | −1.04 | −1.79 | −2.90 |
| **11** | DE | 0.95 | 0.03 | −0.78 | −1.43 | −2.40 |
| **12** | CE | 0.99 | −0.11 | −1.04 | −1.79 | −2.90 |

| Analysis | Key Parameter Varied | Key Parameter Value | *BAF* Value Calculated at the Specified MCF Value | | | |
|---|---|---|---|---|---|---|
| | | | MCF=0.05 | MCF=0.3 | MCF=0.5 | MCF=0.8 |
| **13** | CE | 0.70 | 0.46 | 0.01 | −0.35 | −0.89 |

[1] The following central tendency values were used unless specified as the parameter varied.

- $GWP_{CH_4}$ = 25
- Bo = 0.30
- VolatileCarbon_AT = 0.30
- DE = 0.99
- CE = 0.99

Note: References for the key parameters and values are presented in Table N-8 of Section 4.1.3.

The following observations are noted. At the selected central tendency values, the two methodologies yield very similar results. If the ratio for $CH_4$ generation to $CO_2$ generation had been exactly the same for both methodologies, identical *BAF* values would be produced. When MCF increases, it directly increases the "Avoided $CH_4$ emissions" and the *BAF* values go down (or become more negative). Increasing the global warming potential of methane ($GWP_{CH_4}$), will increase the absolute value of the calculated *BAF* (i.e., if the *BAF* is negative, increasing $GWP_{CH_4}$ will make it more negative; if the *BAF* is positive, increasing $GWP_{CH_4}$ will make *BAF* increase). When $CH_4$ generation increases at a constant overall "potential $CO_2$ emissions" rate (increasing $C_{CH_4}$ or increasing Bo), the impact will be similar to increasing the global warming potential of methane (i.e., it will increase the absolute value of the calculated *BAF*). In the biogas measurement method, the $CO_2$ generation is inversely related to the biogas methane concentration ($C_{CH_4}$) and is not really an independent variable. In the animal population method, the "potential $CO_2$ emissions" is a function of the carbon content of the volatile solids (VolatileCarbon_AT) and can vary independently of the maximum methane generation (Bo). Increasing the VolatileCarbon_AT increases the $CO_2$ emissions relative to $CH_4$ emissions, so increasing VolatileCarbon_AT acts similar to decreasing $C_{CH_4}$ or Bo. If the fraction of methane emitted from anaerobic digestion, estimated as $1 - DE \times CE$, exceeds the fraction of methane emitted from the alternative waste management system, estimated as MCF, then the *BAF* value will be positive.

# 5. Livestock Waste Management through Direct Combustion or Thermochemical Processing and Associated GHG Emissions Pathways

Direct combustion of livestock waste presents an alternative to management through anaerobic storage and treatment, or aerobic treatment, such as composting or field spreading as a soil amendment. Direct combustion of livestock waste is currently not a common management practice in the United States. Management applications typically involve combustion of poultry litter for electricity generation, space heating (e.g., of poultry houses), or combined heat and power (Kelleher et al., 2002; Santoianni et al., 2008).[33] There are also emerging thermochemical conversion

[33] Poultry litter is a mixture of animal bedding materials (e.g., straw, wood chips, or corn husks), manure, and feathers (Santoianni et al., 2008).

processes that can be used to convert and capture the energy in livestock waste via pyrolysis, gasification, co-firing, or direct liquefaction (Cantrell et al., 2008; Santoianni et al., 2008).

Direct combustion or thermochemical conversion of livestock waste is not expected to result in a net increase in $CO_2e$ emissions relative to alternative GHG emissions pathways that could be used to manage the livestock waste.[34] Combustion or thermochemical processing is expected to result in net GHG emissions reductions compared to GHG emissions pathways that involve anaerobic storage and treatment of livestock waste. In short, if the livestock waste were not combusted or processed at high temperatures at a stationary source, resulting in biogenic $CO_2$ emissions, its alternate fate would have resulted in $CH_4$ and/or $CO_2$ emissions through anaerobic decay, aerobic decay, or both. Combustion can, however, introduce toxic metals into the environment.

The biomass contained in livestock waste is the digestive byproducts of consumed plant and animal matter. This represents carbon originally contained in plant matter, often in the form of agricultural crops, produced on a short-rotation basis.[35] As a result, the biomass contained in livestock manure that is combusted (with resulting biogenic $CO_2$ emissions) is typically derived from plant matter $CO_2$ uptake during annual or short, multi-year growth and harvest cycles.

Stationary source combustion or processing of livestock waste can result in the removal of atmospheric $CO_2$. For example, ash produced from the combustion of livestock waste can be used as an agricultural fertilizer, and pyrolysis can be used to produce biochar—a process that stabilizes carbon and can result in long-term carbon storage (Cantrell et al., 2008; Santoianni et al., 2008).

## 5.1. Method for Calculating an Illustrative *BAF* Value Applied to Biogenic Emissions Resulting from Combustion of Livestock Waste

The assessment factor equation can be applied to point source biogenic $CO_2$ emissions from a stationary source combusting livestock waste. This section provides an illustrative method for calculating a *BAF* value that is applied to point source biogenic $CO_2$ emissions from a stationary source combusting livestock waste.

Here the biogenic feedstock is livestock waste, in a form suitable for combustion.[36] In applying the assessment factor equation to this feedstock, the avoided GHG emissions reductions that would have occurred in the absence of combustion of livestock waste at a stationary source are accounted for in the *AVOIDEMIT* term of Equation N.1 (*BAF* = *AVOIDEMIT*). The *AVOIDEMIT* term represents the net GHG emissions reductions that are achieved through combustion of livestock waste, as compared to the emissions pathway from an alternate fate (e.g., treatment via field spreading or in an uncovered anaerobic lagoon). Similar to the other waste-derived biogenic feedstocks, the alternative GHG emissions pathway is accounted for in the numerator of the *AVOIDEMIT* term,

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[34] In the case of thermochemical processing, this does not include fossil fuel energy inputs used to generate process heat.
[35] This would apply to both plant- and animal-based livestock feeds.
[36] Combustion of biomass with high moisture content can be problematic; pre-drying of livestock waste may be required. Some operations have addressed this problem by mixing livestock waste with woody materials (e.g., saw dust) (Santoianni et al., 2008).

whereas the denominator accounts for the GHG emissions resulting from the actual waste management strategy used.

In practice, as applied here, the *AVOIDEMIT* term contains a ratio of the emissions, in $CO_2e$, of the alternative livestock waste management process (had that feedstock not been combusted) to the emissions, in $tCO_2e$, from the combustion of the livestock waste, after accounting for the combustion efficiency of the manure combustor. For the feedstock of livestock waste suitable for combustion, the *AVOIDEMIT* term can be conceptually expressed by the simplified ratio of:

$$AVOIDEMIT = 1 - \frac{(\textbf{emissions from treatment alternative to combustion})}{(\textbf{emissions from treatment by combustion})} \qquad \textbf{(EQ N.47)}$$

### 5.1.1. Calculating the Numerator

In computing *AVOIDEMIT*, the numerator (i.e., emissions from treatment alternative to combustion) can be calculated by assuming that if livestock waste were not managed using combustion, then this waste would have been managed under one or more different waste management systems. For example, the alternative pathway may be associated with waste storage followed by an aerobic land application or it could be anaerobic stored and treated in an uncovered lagoon.[37] The GHGs that would have been generated under these alternative fates can be estimated using methods presented in IPCC (2006b) and EPA (2009b). To estimate the annual $CO_2$ and $CH_4$ emissions resulting from a livestock waste management strategy other than combustion, the numerator of the *AVOIDEMIT* term can be estimated using Equation N.48.

Total $CO_2e$ emissions from a livestock waste management alternate to combustion =

$$\textbf{(avoided } \mathbf{CO_2} \textbf{ emissions) + (avoided } \mathbf{CH_4} \textbf{ emissions)} \qquad \textbf{(EQ. N.48)}$$

The avoided $CO_2$ emissions from a livestock waste management alternate to combustion is equal to the available carbon in the volatile solids of the livestock waste after removing the amount of carbon which becomes $CH_4$ and then converting the remaining available carbon to $CO_2$. This calculation (Equation N.48) is the same as Equation N.25 which was previously used in the calculation of emissions from anaerobic digesters for livestock waste management:

Avoided $CO_2$ emissions (metric tons $CO_2e$/year) =

$$\mathbf{[(PCO_2 \times 12/44) - (Avoided\ CH_4\ emissions/GWP_{CH_4} \times 12/16)] \times (44/12)} \qquad \textbf{(EQ. N.239)}$$

Where:

$PCO_2$ = Potential maximum $CO_2$ emissions, see Equation N.50.

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[37] There are multiple livestock waste management scenarios alternative to combustion of livestock waste. An uncovered anaerobic lagoon would generate the most (see EPA, 2009b, Table A-3). Depending on ambient temperature, $CH_4$ production in an uncovered anaerobic lagoon ranges from 66% to 80% of the maximum amount of $CH_4$ that could potentially be produced from the livestock waste. The appropriate alternative livestock waste management scenario should be used when this calculation is made.

(12/44) = molecular weight ratio of C to $CO_2$ (converts potential $CO_2$ emissions to carbon).

Avoided $CH_4$ emissions = avoided $CH_4$ emissions, metric tons $CO_2e$/year (see Equation N.53, below).

$GWP_{CH_4}$ = 100-year GWP for $CH_4$.

(12/16) = molecular weight ratio of C to $CH_4$ (converts $CH_4$ emissions to carbon).

(44/12) = molecular weight ratio of $CO_2$ to C (converts C less that associated with the $CH_4$ emissions back to $CO_2$ emissions).

Potential $CO_2$ emissions can be solved using Equation N.50, which was also used in estimating the emissions associated with livestock waste management via an anaerobic digester:

Potential $CO_2$ emissions =

$$\boldsymbol{\Sigma_{\text{animal type}} (TVS_{AT} \times VolatileCarbon_{AT} \times (44/12) \times 365 \times 1/1{,}000)} \qquad \textbf{(EQ. N.50)}$$

Where:

$\Sigma_{\text{animal type}}$ = If the alternate waste management system accepts waste from more than one animal type then this calculation must be computed for each animal type and then summed across animal types.

$TVS_{AT}$ = Total volatile solids excreted by animal type (kg/day); the $TVS_{AT}$ equation is presented below (Equation N.51).

$VolatileCarbon_{AT}$ = Fraction of degradable carbon in the volatile solids of the livestock waste (see Equation N.52).

44/12 = molecular weight ratio of $CO_2$ to C.

365 = number of days per year (i.e., 365 days/year).

1/1,000 = conversion factor from kg to metric tons.

Total volatile solids excreted by animal type ($TVS_{AT}$) may be calculated using the following equation (previously presented under livestock waste management using anaerobic digester) and referring to tables external to this appendix (Table A-2, EPA, 2009b and Tables JJ-2 and JJ-3, 40 CFR 98.363):

$$\boldsymbol{TVS_{AT} = (Population_{AT} \times TAM_{AT} \times VS_{AT}/1{,}000)} \qquad \textbf{(EQ. N.51)}$$

Where:

$TVS_{AT}$ = Total volatile solids excreted per animal type (kg/day).

$Population_{AT}$ = Average annual animal population (head), by animal type.[38]

$TAM_{AT}$ = Typical animal mass, by animal type; determined using either default values (see EPA, 2009b, Table A-2) or farm specific data (kg/head).

$VS_{AT}$ = Volatile solids excretion rate by animal type, using either default values (see 40 CFR 98.363, Tables JJ-2 and JJ-3) or farm specific data (kg VS/day/kg animal mass).

The fraction of degradable carbon in the volatile solids of the livestock waste ($VolatileCarbon_{AT}$) can be estimated using results from proximate and ultimate analyses[39] of the livestock waste specific to the waste of animal type being managed (see Equation N.52). Data needed to estimate this parameter can be directly measured or, more simply, can be taken from the body of published scientific literature.[40]

$$\mathbf{VolatileCarbon_{AT}} = \frac{\text{Carbon} - \text{Fixed Carbon}}{\text{Volatile Solids}} = \frac{\text{Carbon} - \text{Fixed Carbon}}{\text{Volatile Matter} + \text{Fixed Carbon}} \qquad \textbf{(EQ. N.52)}$$

Where:

$VolatileCarbon_{AT}$ = Fraction of the degradable carbon in the volatile solids of the livestock waste (kg degradable carbon in volatile solids/kg of volatile solids, dry basis).[41]

Carbon = Fraction of carbon in livestock waste (kg carbon in total dried solids/kg of total dried solids), dry basis (from ultimate analysis).

Fixed Carbon = Fraction of dry solids in livestock waste that does not volatilize when heated to 900 °C in nitrogen (kg fixed carbon in total dried solids/kg of total dried solids) but is lost when heated in air at 600 °C, dry basis (from fuels proximate analysis; see Figure N-3).

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38 For static populations (e.g., dairy cows, breeding swine), average annual animal populations are estimated using annual animal inventory or equivalent. For growing populations (e.g., meat animals such as beef and veal cattle), average annual animal populations are estimated using the average number of days each animal is kept at the facility and the number of animals produced annually (e.g., growing population = days onsite × (number of animals produced annually / 365)).

39 Characteristics of a biogenic feedstock can be described using proximate and ultimate analyses based on a sample's complete combustion to $CO_2$ and liquid water. The proximate analysis gives moisture content, volatile content, carbon remaining (fixed carbon), and mineral ash. The ultimate analysis gives the sample's elemental composition as proportions of carbon, hydrogen, oxygen, nitrogen, and sulfur. Standardized test methods have been developed, for example, see Table 3 in Demirbas (2004).

40 For example, ASAE Standard D384.2 (2005) is useful for estimating general characteristics of livestock and poultry manure. Li et al. (2008) and Henihan et al. (2003) present specific results of proximate and ultimate analyses of chicken litter characteristics; Sweeten et al. (2002 and 2003) present similar specific results but of cattle manure.

41 If the mass of fixed carbon is not 100% carbon then the amount of carbon in the volatile solids may be underestimated, thus giving a low-biased estimate of the $VolatileCarbon_{AT}$ term (though the bias is likely small).

Volatile Solids = Fraction of volatile solids in livestock waste (kg volatile solids in total dried solids/kg of total dried solids), dry basis (from waste volatile solids analysis). If only fuels proximate analysis is available, estimate the volatile solids as the sum of the volatile matter and fixed carbon from the fuels proximate analysis (see Figure N-3).

Volatile Matter = fraction of dry solids that does is lost when heated to 900 °C in nitrogen (kg volatile matter/kg of total dried solids), dry basis.

Equation N.53 can be used to estimate the annual avoided $CH_4$ emissions generated from a manure management strategy alternate to combustion:

Avoided $CH_4$ emissions (metric tons $CO_2e$/year) =

$$\mathbf{\Sigma_{animal\ type}\ (TVS_{AT} \times VS_{WMS} \times Days \times B_0 \times MCF_{WMS} \times 0.662 \times 1/1{,}000 \times GWP_{CH_4})} \quad \textbf{(EQ. N.53)}$$

Where:

$\Sigma_{animal\ type}$ = If the alternate waste management system accepts waste from more than one animal type then this calculation must be computed for each animal type and then summed across animal types.

$TVS_{AT}$ = Total volatile solids excreted by animal type (kg/day); (Equation N.51).

$VS_{WMS}$ = Proportion of total manure for each animal type that is managed in each waste management system (assumed to be equivalent to the amount of volatile solids in each waste management system).

Days = Number of days per year (i.e., 365 days/year).

$B_0$ = Maximum $CH_4$-producing capacity for each animal type ($m^3$ $CH_4$/kg volatile solids; see EPA, 2009b, Table A-2).

$MCF_{WMS}$ = $CH_4$ conversion factor (proportion represented as a decimal value) for the alternative-scenario, waste management system (see EPA, 2009b, Table A-3).

0.662 = density of $CH_4$, kg $CH_4$ / $m^3$ (at 531.67°R, or 22.22°C, and 1 atm).

1/1000 = conversion factor from kg to metric tons.

$GWP_{CH_4}$ = 100-year GWP of $CH_4$, 25 (IPCC, 1996).

The maximum amount of $CH_4$ that could potentially be produced from livestock waste managed under ideal conditions is calculated by multiplying the total volatile solids by the maximum $CH_4$-producing capacity of the livestock waste ($B_0$). The $B_0$ values vary by animal type and diet (see EPA, 2009b, Table A-2). Most manure management systems will not produce the maximum amount of $CH_4$ possible because the conditions in the systems are not ideal for $CH_4$ production. The $CH_4$-

producing potential of a specific livestock waste management system is represented by a methane conversion factor (MCF). The value of this parameter ranges from 0% to 100% and reflects the capability of a system to produce the maximum achievable $CH_4$ (the higher the MCF, the greater the potential for $CH_4$ production). For liquid systems (e.g., uncovered anaerobic lagoons), MCF values are temperature dependent: in order to assign the appropriate MCF for the type of liquid system used, the average ambient temperature at the system's location must be known (see EPA, 2009b, Table A-3).

Summing the avoided $CO_2$ emissions and the avoided $CH_4$ emissions (Equation N.48, metric tons $CO_2$e/year) is the final computation in estimating the numerator in the *AVOIDEMIT* term for a livestock waste management strategy alternative to combustion. See Section 5.2.2 for two illustrative example calculations of the numerator in the *AVOIDEMIT* term and its subsequent application in estimating a *BAF*.

### 5.1.2. Calculating the Denominator

In the derivation of *AVOIDEMIT*, the denominator (emissions from livestock waste treatment by combustion) is based on the carbon content of the point source, stack emissions from the waste combustion unit. The value of the denominator is equal to the $CO_2$e of the combusted livestock waste, adjusted by the combustion efficiency of the incinerator.[42] This adjustment is to account for the proportion of feedstock that is neither combusted nor emitted to the atmosphere as a point-source emission. Combustion of livestock waste does not create or emit $CH_4$; the only GHG associated with this process is $CO_2$. The principal products of livestock waste combustion include $CO_2$, carbon monoxide (CO), hydrocarbons (HC), nitrogen oxides ($NO_x$), sulfur oxides ($SO_x$), inorganic bottom ash, and fly ash (Antares Group Inc. et al., 1999). Carbon monoxide and HC are important indicators of incomplete combustion; with complete combustion, $CO_2$ is the primary carbon-based emission (Antares Group Inc. et al., 1999).

Total $CO_2$e emissions from combustion of livestock waste can be computed using the following equation:

$CO_2$ emissions from incineration of livestock waste (MT $CO_2$e/year) =

$$\boldsymbol{\Sigma_{\text{animal type}} (TVS_{AT} \times TotalCarbon_{AT} \times DE \times (44/12) \times 365 \times 1/1{,}000)} \qquad \textbf{(EQ. N.244)}$$

Where:

$TVS_{AT}$ = total volatile solids excreted per animal type (kg/day), see Equation N.51.

[42] Combustion efficiency of livestock waste varies with the type of boiler used, moisture content, and particle size of the feedstock. In an efficient combustor, very little carbon in poultry litter is left unburned (Antares Group Inc. et al. 1999). One published value of combustion efficiency for combustion of broiler litter, based on the carbon content in the ash is 96% (Costello, 2007).

$TotalCarbon_{AT}$ = fraction of the carbon in the volatile solids of the livestock waste, see Equation N.55.

DE = livestock waste destruction efficiency (i.e., combustion efficiency of incinerator).

44/12 = molecular weight ratio of $CO_2$ to C.

365 = number of days per year (i.e., 365 days/year).

1/1,000 = conversion factor from kg to metric tons.

The fraction of total carbon in the volatile solids of the livestock waste ($TotalCarbon_{AT}$) can be estimated using results from proximate and ultimate analyses[43] of the livestock waste specific to the waste of animal type being managed. It is again assumed that there is negligible carbon in the residue, but the fixed carbon is expected to oxidize during the manure combustion process. Thus, it is assumed that all of the combustible carbon exists in the volatile solids fraction of the dried solids. Data needed to estimate this parameter can be directly measured or, more simply, can be taken from the body of published scientific literature.[44]

$$\mathbf{TotalCarbon_{AT}} = \frac{\textbf{Carbon}}{\textbf{Volatile Solids}} = \frac{\textbf{Carbon}}{\textbf{Volatile Matter + Fixed Carbon}} \qquad \textbf{(EQ. N.255)}$$

Where:

$TotalCarbon_{AT}$ = fraction of carbon in the volatile solids of the livestock waste, (kg carbon in volatile solids/kg of volatile solids, dry basis).

Carbon = fraction of carbon in livestock waste (kg carbon in total dried solids/kg of total dried solids), dry basis (from ultimate analysis).

Volatile Solids = fraction of volatile solids in livestock waste (kg volatile solids in total dried solids/kg of total dried solids), dry basis (from waste volatile solids analysis). If only fuels proximate analysis is available, estimate the volatile solids as the sum of the volatile matter and fixed carbon from the fuels proximate analysis (see Figure N-3).

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[43] Characteristics of a biogenic feedstock can be described using proximate and ultimate analyses based on a sample's complete combustion to $CO_2$ and liquid water. The proximate analysis gives moisture content, volatile content, carbon remaining (fixed carbon), and mineral ash. The ultimate analysis gives the sample's elemental composition as proportions of carbon, hydrogen, oxygen, nitrogen, and sulfur. Standardized test methods have been developed, for example, see Table 3 in Demirbas (2004).

[44] For example, ASAE Standard D384.2 (2005) is useful for estimating general characteristics of livestock and poultry manure. Li et al. (2008) and Henihan et al. (2003) present specific results of proximate and ultimate analyses of chicken litter characteristics; Sweeten et al. (2002 and 2003) present similar specific results but of cattle manure.

Volatile Matter = fraction of dry solids that does is lost when heated to 900 °C in nitrogen (kg volatile matter/kg of total dried solids), dry basis (from fuels proximate analysis; see Figure N-3)

Fixed Carbon = fraction of dry solids in livestock waste that does not volatilize when heated to 900 °C in nitrogen (kg fixed carbon in total dried solids/kg of total dried solids) but is lost when heated in air at 600 °C, dry basis (from fuels proximate analysis; see Figure N-3).

After solving for the numerator and the denominator of the *AVOIDEMIT* term, the *BAF* can be calculated using Equation N.1. See Section 5.2.2 for two illustrative example calculations of the numerator and denominator in the *AVOIDEMIT* term and its subsequent application in estimating a *BAF* for livestock waste management by way of direct combustion.

Several parameters are presented and used in the equations in the remainder of this section. Table N-11 presents the parameters used, typical or default values, ranges presented in the literature, and references.

**Table N-11. Summary of Parameters Used When Calculating an Illustrative *BAF* for Livestock Waste Management through Combustion.**

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Typical animal mass, by animal type** | $TAM_{AT}$ | 604 | Numerous | kg/head | Determined using either default values or farm-specific data | EPA, 2009b, Table A-2; IPCC, 2006, Table 10A4-10A9 |
| **Volatile solids excretion rate by animal type** | $VS_{AT}$ | 9.34 | Depends on the type of animal group | kg VS/day/kg animal mass | Value presented is used in the example calculations in Section 4.0.[1] | EPA, 2009b, Table A-2; EPA, 2013c, Tables JJ-2 and JJ-3 |
| **Maximum $CH_4$-producing capacity for each animal type** | $B_0$ | 0.24 | 0.17 to 0.78 | $m^3$ $CH_4$ /kg volatile solids | Value presented is for dairy cows | EPA, 2009b, Table A-2 |
| **Density of $CH_4$** | -- | 0.662 | -- | kg $CH_4/m^3$ | At 532°R, or 22.22°C, and 1 atm | EPA, 2009b |
| **Destruction efficiency** | DE | 0.96 | 0.90 to 0.9977 | decimal percent | | EPA, 2009b, Table A-4 |

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Fraction of volatile solids in livestock waste** | Volatile Carbon_AT | | 0.20 to 0.40 | kg volatile solids in total dried solids/kg of total dried solids, dry basis | Value is determined from waste volatile solids analysis[2] | Sweeten et al., 2002 |

[1] $VS_{AT}$ can be determined using either default values or farm-specific data.

[2] If only fuels proximate analysis is available, estimate the volatile solids as the sum of the volatile matter and fixed carbon from the fuels proximate analysis (see Figure N-3).

## 5.2. Example *AVOIDEMIT* and *BAF* Calculations for Direct Combustion of Livestock Waste

Two example scenarios of alternative management for poultry litter are presented here. Both examples are set in northern Georgia and consist of 400,000 broilers (chickens).[45] In the first example, the alternative management strategy is to store the chicken litter as a solid prior to its application as a soil amendment; the chicken litter would be stored for approximately 1 year prior to its land application (EPA, 2001).[46] In the second example, the alternative management strategy, although not as commonly used, is to store and manage the chicken litter from broilers in an uncovered anaerobic lagoon.[47] Several of the calculated parameter values can be used in both scenarios (i.e., $TVS_{AT}$, $VolatileCarbon_{AT}$, and the potential maximum $CO_2$ emissions).

### 5.2.1. Example Calculation for Direct Combustion and Land Application of Livestock Waste

#### Step 1: Calculating the Avoided $CH_4$ from the Alternate Treatment

To calculate the numerator of the *AVOIDEMIT* term, the total volatile solids must be estimated in order to calculate the avoided $CH_4$ emissions. Parameters in Equation N.48 can be estimated using a hypothetical example of an operation set in northern Georgia, consisting of 400,000 broilers per year, with a typical animal mass of 0.9 kg, and a volatile solids excretion rate of 15 kg VS/day/1000 kg animal mass (see 40 CFR 98.363, Table JJ-2). Equation N.51 may be used to calculate total volatile solids excreted per animal type:

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[45] Regional differences in ambient temperature affect the value of the $CH_4$ conversion factor ($MCF_{WMS}$) used to calculate avoided $CH_4$ emissions. Therefore, the geographic location of the manure management system has an effect on the amount of generated emissions and the value of the *BAF*. However, the number of animals (e.g., broilers) whose waste is managed does not have an effect on the value of the *BAF*.

[46] Large quantities of poultry litter are removed from the poultry house during annual clean-out. If possible, the annual clean-out typically is timed to coincide with the time land is available for land application (EPA, 2001).

[47] Although chicken litter from broilers can be managed in anaerobic lagoons, it is more common to use this management strategy for pullets.

$$\textbf{TVS}_{AT} = (\textbf{Population}_{AT} \times \textbf{TAM}_{AT} \times \textbf{VS}_{AT}/1{,}000)$$

$$= 400{,}000 \times 0.9 \text{ kg} \times 15 \text{ kgVS/day/kg animal mass}/1{,}000$$

$$= 5{,}400 \text{ kg/day}$$

With $TVS_{AT}$ estimated, the avoided $CH_4$ emissions resulting from a poultry litter management strategy other than an anaerobic digester can be calculated. In this hypothetical example set in a temperate region (e.g., northern Georgia) with an average temperature of 16ºC, the alternative fate could have been to store the chicken litter as a solid prior to its application as a soil amendment; the chicken litter would be stored for approximately 1 year prior to its land application (EPA, 2001).

In this first hypothetical scenario, the only litter-producing animal type is broilers, and the only alternate waste management system of the poultry production with litter is solid storage prior to land application. When calculating the avoided $CH_4$ emissions associated with the alternative fate of chicken litter under this scenario, several parameters are needed, including the maximum $CH_4$-producing capacity for broilers (see EPA, 2009b, Table A-2, for the appropriate default value; for broilers, $B_0$ = 0.36 $m^3$ $CH_4$/kg); and a $CH_4$ conversion factor for the alternative-scenario's waste management system (see EPA, 2009b, Table A-3; for solid storage in a temperate climate, $MCF_{WMS}$ = 0.04). Equation N.53 can be used to calculate the avoided $CH_4$ emissions:

Avoided $CH_4$ emissions (metric tons $CO_2$e per year)

$$= 25 \times (\textbf{TVS}_{AT} \times \textbf{VS}_{WMS} \times 365 \text{ days/year} \times \textbf{B}_0 \times \textbf{MCF}_{WMS} \times 0.662 \text{ kg } CH_4/m^3 \times 1 \text{ metric ton}/1{,}000 \text{ kg})$$

$$= 25 \times (5{,}400 \times 1 \times 365 \times 0.36 \text{ m}^3 \text{ CH}_4\text{/kg} \times 0.04 \times 0.662 \times 1/1{,}000)$$

$$= 469.7287 \text{ MT CO}_2\text{e per year.}$$

### Step 2: Calculating the Avoided $CO_2$ from the Alternate Treatment

To estimate the avoided $CO_2$ emissions from a poultry litter management strategy other than combustion, the fraction of degradable carbon in the volatile solids of the chicken litter must be calculated. For the scenarios presented here (Section 5.2), data (results from proximate and ultimate analyses of poultry litter) from Li et al. (2008) can be used to populate parameters in Equation N.48. Because Li et al. (2008) presented results of the proximate analysis on a wet basis, those parameters must be converted to a dry basis by dividing each of them by the proportion of the dry content of the chicken litter (i.e., divide the fixed carbon and the volatile solids each by (1–moisture content)). In the $VolatileCarbon_{AT}$ term, the fixed carbon is removed from the total carbon in the poultry litter because the fixed carbon is assumed not to degrade. Using these data, Equation N.52 can be parameterized as follows:

$$\textbf{VolatileCarbon}_{AT} = \frac{\textbf{Carbon–Fixed Carbon}}{\textbf{Volatile Solids}} = \frac{\textbf{Carbon–Fixed Carbon}}{\textbf{Volatile Matter + Fixed Carbon}}$$

$$= \frac{\mathbf{0.282 - 0.0688}}{\mathbf{0.6516 + 0.0688}}$$

$$= \mathbf{0.2959}$$

With $TVS_{AT}$ and $VolatileCarbon_{AT}$ estimated, the potential maximum $CO_2$ emissions (Equation N.50) resulting from a chicken litter management strategy alternative to combustion can then be estimated. In this scenario, the alternate fate is solid manure storage for approximately 1 year before it is ultimately used as a land application. Because waste management is limited to litter from only one animal type (broilers), Equation N.50 only needs to be calculated once, as follows:

Potential $CO_2$ emissions

$$\mathbf{= (TVS_{AT} \times VolatileCarbon_{AT} \times (44/12) \times 365\ days/year \times 1\ metric\ ton/1{,}000\ kg)}$$

$$\mathbf{= 5{,}400 \times 0.2959 \times (44/12) \times 365 \times 1/1{,}000}$$

$$\mathbf{= 2{,}138.4693\ MT\ CO_2\ per\ year}$$

With estimates of both the potential $CO_2$ emissions and the avoided $CH_4$ emissions, the avoided $CO_2$ emissions resulting from an aerobic poultry litter management strategy can be calculated. For this calculation the total potential $CO_2$ emissions from the decomposition of $TVS_{AT}$ must first be converted to carbon. From this is subtracted the $CH_4$ emissions after converting them to carbon. The result is then converted back to $CO_2$ (see Equation N.49):

Avoided $CO_2$ emissions

$$\mathbf{= ((Potential\ CO_2\ emissions \times 12/44) – (Avoided\ CH_4\ emissions/25 \times 12/16)) \times (44/12)}$$

$$\mathbf{= ((2{,}138.4693 \times 12/44) – (469.7287 / 25 \times 12/16)) \times (44/12)}$$

$$\mathbf{= 2{,}086.7990\ MT\ CO_2\ per\ year}$$

### Step 3: Calculating the $CO_2e$ from the Alternate Fate (Numerator)

The final calculation in solving for the numerator of the *AVOIDEMIT* term for this scenario of chicken litter management is to sum the estimated, avoided $CH_4$ and the avoided $CO_2$ emissions (Equation N.48). This summation represents the total avoided $CO_2e$ emissions (in metric tons $CO_2e$ per year) that result from storage of the solid litter:

Total avoided $CO_2e$ emissions

$$\mathbf{= (avoided\ CO_2\ emissions) + (avoided\ CH_4\ emissions)}$$

$$\mathbf{= 2{,}086.7990 + 469.7287}$$

$$\mathbf{= 2{,}556.5277\ metric\ tons\ CO_2e\ per\ year.}$$

### Step 4: Calculating the $CO_2e$ from the Anaerobic Digester (Denominator)

Equation N.54 can be used to calculate the denominator of the *AVOIDEMIT* term. In both of the scenarios presented here, poultry litter management is limited to that from only one animal type (i.e., broilers). The above calculated values for total volatile solids excreted per broiler ($TVS_{AT}$) is still applicable. For combustion, however, the fraction of total carbon, rather than degradable carbon, in the volatile solids must be calculated. Using Equation N.55, fraction of carbon in the volatile solids of the poultry litter ($TotalCarbon_{AT}$) is calculated as:

$$\mathbf{TotalCarbon_{AT}} = \frac{\mathbf{Carbon}}{\mathbf{Volatile\ Solids}} = \frac{\mathbf{Carbon}}{\mathbf{Volatile\ Matter + Fixed\ Carbon}}$$

$$= \frac{\mathbf{0.282}}{\mathbf{0.6516+0.0688}}$$

$$= \mathbf{0.3914}$$

Given these values, the total $CO_2e$ emissions from combusting the litter from the 400,000 broilers assuming a 96% destruction efficiency of the waste[48] is calculated as:

$CO_2$ emissions from incineration of poultry litter (metric tons $CO_2e$/year)

$$= \mathbf{TVS_{AT} \times TotalCarbon_{AT} \times (44/12) \times DE \times days/year \times 1\ metric\ ton/1{,}000kg}$$

$$= \mathbf{5{,}400 \times 0.3914 \times (44/12) \times 0.96 \times 365 \times 1/1{,}000}$$

$$= \mathbf{2{,}715.5019\ metric\ tons\ CO_2e\ per\ year}$$

### Step 5: Calculating the *BAF* Value

After solving for the numerator and the denominator of the *AVOIDEMIT* term associated with this litter management strategy (i.e., litter storage prior to a land application), the *BAF* can be calculated using Equation N.1 and Equation N.2:

$$\boldsymbol{BAF = AVOIDEMIT}$$

$$= \mathbf{1 - \frac{2{,}556.5277\ metric\ tons\ CO_2e/year}{2{,}715.5019\ metric\ tons\ CO_2e/year}}$$

$$= \mathbf{0.06}$$

48 Combustion efficiency of livestock waste varies with the type of boiler used, moisture content, and particle size of the feedstock. In an efficient combustor, very little carbon in poultry litter is left unburned (Antares Group Inc. et al., 1999). One published value of combustion efficiency for combustion of broiler litter, based on the carbon content in the ash is 96% (Costello, 2007).

The *BAF* is small, but positive, indicating that the alternate disposal scenario of storing the chicken litter as a solid prior to its application as a soil amendment has emissions similar to, but slightly lower than manure combustion.

### 5.2.2. Calculation for Direct Combustion and an Uncovered Anaerobic Lagoon

The second scenario presented is an alternate management strategy of this chicken litter via storage and treatment in an open anaerobic lagoon.[49] An uncovered anaerobic lagoon is capable of generating more $CH_4$ than is management via solid storage prior to land application. As a result the calculated avoided $CH_4$ emissions and the avoided $CO_2$ emissions will differ. However, between these two scenarios, the calculated values for $TVS_{AT}$ (5,400 kg/day), $VolatileCarbon_{AT}$ (0.2959), and the potential maximum $CO_2$ emissions (2,138.4693 metric tons $CO_2$ per year) would not be different; those values calculated above, can be imported into the equations needed to calculate the avoided $CH_4$ emissions and the avoided $CO_2$ emissions (Equations N.53, and N.49, respectively).

#### Step 1: Calculating the Avoided $CH_4$ Emissions from the Alternate Fate

When calculating the avoided $CH_4$ emissions associated with the alternate fate of the chicken litter under this scenario, several parameters are needed, including the maximum $CH_4$-producing capacity for broilers (see EPA, 2009b, Table A-2, for the appropriate default value; for broilers, $B_0$ = 0.36 $m^3$ $CH_4$/kg; this is the same as under the previous example); and a $CH_4$ conversion factor for the alternative-scenario's waste management system (see EPA, 2009b, Table A-3; for an uncovered anaerobic lagoon in a temperate climate of 16°C, $MCF_{WMS}$ = 0.75). The following equation (Equation N.53) can be used to calculate the avoided $CH_4$ emissions for treatment of this chicken litter via an uncovered anaerobic lagoon:

Avoided $CH_4$ emissions (metric tons $CO_2e$ /year)

$$\mathbf{= 25 \times (TVS_{AT} \times VS_{WMS} \times 365\ days/year \times B_0 \times MCF_{WMS} \times 0.662\ kg\ CH_4/m^3 \times 1\ metric\ ton/1{,}000\ kg)}$$

$$\mathbf{= 25 \times (5{,}400\ kg/day \times 1 \times 365 \times 0.36\ m^3\ CH_4/kg \times 0.75 \times 0.662\ kg\ CH_4/m^3 \times 1/1{,}000)}$$

$$\mathbf{= 8{,}807.4135\ MT\ CO_2e\ per\ year}$$

#### Step 2: Calculating the $CO_2$ Emissions from the Alternate Fate

With estimates of both the potential $CO_2$ emissions and the avoided $CH_4$ emissions, the avoided $CO_2$ emissions resulting from an uncovered anaerobic lagoon poultry litter management strategy can be calculated. For this calculation, the total potential $CO_2$ emissions from the decomposition of $TVS_{AT}$ must first be converted to carbon. From this is subtracted the $CH_4$ emissions after converting them to carbon. The result is then converted back to $CO_2$ (see Equation N.49):

[49] As previously mentioned, chicken litter from broilers can be managed in anaerobic lagoons, though it is more common to use this management strategy for pullets.

Avoided $CO_2$ emissions

$$= ((\text{Potential } CO_2 \text{ emissions} \times 12/44) - (\text{Avoided } CH_4 \text{ emissions}/21 \times 12/16)) \times (44/12)$$

$$= ((2{,}138.4693 \times 12/44) - (7{,}398.2273/21 \times 12/16)) \times (44/12)$$

$$= 1{,}169.6538 \text{ MT CO2 per year}$$

### Step 3: Calculating the $CO_2$e Emissions from the Alternate Fate

Summing the estimated, avoided $CH_4$ and $CO_2$ emissions (both in metric tons $CO_2$e per year) that result from an uncovered, anaerobic lagoon management strategy for poultry litter (Equation N.25), the numerator of the *AVOIDEMIT* term for this scenario can be computed as:

Total avoided $CO_2$e emissions

$$= (\text{avoided } CO_2 \text{ emissions}) + (\text{avoided } CH_4 \text{ emissions})$$

$$= 1{,}169.6538 + 8{,}807.4135$$

$$= 9{,}977.0673 \text{ MT } CO_2e \text{ per year}$$

### Step 4: Calculating the *BAF* Value

The denominator of the *AVOIDEMIT* term represents the $CO_2$e emissions associated with combustion of the poultry litter (in this case, litter from 400,000 broilers). The computed value of the denominator was previously calculated using Equation N.54 and is unchanged by the alternate fate of the managed poultry litter. Therefore, both the numerator and the denominator of the *AVOIDEMIT* term associated with this waste management strategy (i.e., an uncovered anaerobic lagoon) have been calculated. The *BAF* can be computed using Equations N.1 and N.2:

$$BAF = AVOIDEMIT$$

$$= 1 - \frac{9{,}977.0673}{2{,}715.5019}$$

$$= -2.67$$

## 5.3. Sensitivity Analysis for Livestock Waste Management through Direct Combustion

Table N-12 presents the results of the sensitivity analysis performed for the direct combustion of livestock waste. Sources for the parameter values used here can be found in Table N-11 of Section 5.1.2. The parameter that has the greatest variability is the methane correction factor (MCF) for the waste management system employed as an alternative to direct livestock waste combustion. Increasing the MCF decreases the *BAF* (including making negative *BAF* values more negative). Increasing the GWP of $CH_4$ (GWP_CH$_4$) also decreases the *BAF* value. The methane generation potential (Bo) and the volatile carbon content (VolatileCarbon_AT) of the waste has a similar affect; increasing Bo or VolatileCarbon_AT decreases *BAF*. The total carbon content (TotalCarbon_AT) is only used in the denominator of the emission ratio term of *AVOIDEMIT*, so increasing

TotalCarbon_AT increases *BAF* (including making negative *BAF* less negative). In the same manner, the waste combustor destruction efficiency (DE) only impacts the denominator. Since the destruction efficiency here reflects the fraction of the total carbon that is oxidized in the combustor, lowering DE reduces the emissions in the denominator causes *BAF* to decrease (or become more negative).

**Table N-12. Sensitivity Analysis of Anaerobic Digestion using Animal Population Data.[1]**

| Analysis | Key Parameter Varied | Key Parameter Value | *BAF* Value Calculated at the Specified MCF Value | | | |
|---|---|---|---|---|---|---|
| | | | MCF=0.05 | MCF=0.3 | MCF=0.5 | MCF=0.8 |
| 1 | $GWP_{CH_4}$ | 21 | 0.21 | −0.35 | −0.80 | −1.47 |
| 2 | $GWP_{CH_4}$ | 25 | 0.18 | −0.50 | −1.05 | −1.87 |
| 3 | $GWP_{CH_4}$ | 28 | 0.16 | −0.61 | −1.23 | −2.16 |
| 4 | Bo | 0.15 | 0.25 | −0.09 | −0.36 | −0.77 |
| 5 | Bo | 0.30 | 0.18 | −0.50 | −1.05 | −1.87 |
| 6 | Bo | 0.50 | 0.09 | −1.05 | −1.96 | −3.32 |
| 7 | VolatileCarbon_AT | 0.20 | 0.41 | −0.27 | −0.82 | −1.64 |
| 8 | VolatileCarbon_AT | 0.30 | 0.18 | −0.50 | −1.05 | −1.87 |
| 9 | VolatileCarbon_AT | 0.40 | −0.04 | −0.73 | −1.27 | −2.09 |
| 10 | TotalCarbon_AT | 0.30 | −0.23 | −1.25 | −2.07 | −3.30 |
| 11 | TotalCarbon_AT | 0.45 | 0.18 | −0.50 | −1.05 | −1.87 |
| 12 | TotalCarbon_AT | 0.60 | 0.39 | −0.13 | −0.53 | −1.15 |
| 13 | DE | 0.99 | 0.19 | −0.48 | −1.03 | −1.84 |
| 14 | DE | 0.98 | 0.18 | −0.50 | −1.05 | −1.87 |
| 15 | DE | 0.95 | 0.16 | −0.55 | −1.11 | −1.96 |

[1] The following central tendency values were used unless specified as the parameter varied.

- $GWP_{CH_4}$ = 25
- Bo = 0.30
- VolatileCarbon_AT = 0.30
- TotalCarbon_AT = 0.45
- DE = 0.99.

Note: References for the key parameters and values are presented in Table N-11 of Section 5.1.2.

# 6. Wastewater Disposal in Wastewater Treatment Facilities and Associated GHG Emission Pathways

Wastewater from domestic and industrial sources is treated to remove soluble organic matter, suspended solids, pathogenic organisms, and chemical contaminants from the wastewater prior to its discharge into natural water systems. In the United States, approximately 20% of domestic wastewater is treated in septic systems or other onsite systems, while the rest is collected and treated centrally (EPA, 2014b). Centralized wastewater treatment systems, such as publicly owned treatment works, may include a variety of processes, ranging from treatment in lagoons to advanced tertiary treatment technology for removing nutrients. In the United States, there are approximately 14,780 wastewater treatment plants (Lono-Batura et al., 2012).

Soluble organic matter in wastewater is generally removed via biological processes in which microorganisms biodegrade the organic matter under aerobic or anaerobic conditions. Sludges (also referred to as wastewater biosolids after they have been treated) are the product of most wastewater treatment systems (Bogner et al., 2007; RTI International, 2010). Carbon dioxide, $CH_4$ and $N_2O$ can be produced and released to the atmosphere at various stages between the initial point of wastewater collection and its final disposal, including wastewater transport, sewage treatment processes, and anaerobic digestion of wastewater or sludges (Bogner et al., 2007). In the United States, domestic and industrial wastewater treatment accounted for approximately 2.3% of $CH_4$ emissions in 2012 (totaling 12.8 Tg $CO_2$e) and 1.2% of $N_2O$ emissions (totaling 5.0 Tg $CO_2$e) (EPA, 2014b).

Methane is microbially produced under anaerobic conditions. Domestic wastewater $CH_4$ emissions originate from both septic systems and from centralized treatment systems. Within centralized systems, $CH_4$ emissions can arise from aerobic systems that are not well managed (resulting in anaerobic conditions) or that are designed to have periods of anaerobic activity (e.g., constructed wetlands), anaerobic systems (e.g., anaerobic lagoons), a mixed aerobic and anaerobic systems (e.g., facultative lagoons with surface aerobic zones and deeper anaerobic zones), and from anaerobic digesters if captured biogas is released through leaks, venting, or incomplete combustion (Bogner et al., 2007; RTI International, 2010; EPA, 2014b). During collection and treatment, wastewater may be accidentally or deliberately managed under anaerobic conditions. Wastewater and wastewater sludge may be further biodegraded under aerobic conditions or anaerobic conditions, including anaerobic digestion, agricultural reuse, or incineration (Bogner et al., 2007; EPA, 2014b).

$N_2O$ is an intermediate product of microbial nitrogen cycling; it is generated via the treatment of domestic wastewater during both nitrification and denitrification of the nitrogen present, usually in the form of urea, ammonia, and proteins. These compounds are converted to nitrate ($NO_3$) through the aerobic process of nitrification. Denitrification occurs under anoxic conditions, and involves the biological conversion of nitrate into $N_2$ (EPA, 2014b). The amount of nitrogen present in the influent wastewater determines the $N_2O$ generation potential.

Collection of biogas generated in the wastewater treatment process is primarily, if not entirely, restricted to treatment systems using anaerobic digesters. Anaerobic digesters are used to enhance the degradation process of wastewater and wastewater sludge, thereby producing biogas. If the $CH_4$ generated by an anaerobic wastewater treatment process or anaerobic sludge digestion process is captured and combusted (in a flare or other combustion device), then $CH_4$ is destroyed and converted to $CO_2$, resulting in a net decrease in GHG emissions. $N_2O$ is not a product of wastewater treatment via an anaerobic digester.

It is unknown how many U.S. wastewater treatment facilities use anaerobic digesters to treat wastewater and wastewater sludge. However, a survey of 5,128 U.S. wastewater treatment facilities (of the 14,780 facilities) concluded that at least 1,238 (24% of this subsample) treat sludge using anaerobic digesters and collect the biogas produced (Lono-Batura et al., 2012). The majority of wastewater treatment facilities that use anaerobic digesters are large (treating over one million gallons per day). However, this represents less than 40% of the large wastewater treatment facilities in the U.S; this sector has potential to expand (Lono-Batura et al., 2012). Collected biogas from waste water treatment facilities is most commonly flared, though some wastewater treatment facilities use it for energy generation or sell it for use off-site; several facilities reported releasing the collected biogas directly to the atmosphere (Lono-Batura et al., 2012). Because anaerobic digesters enhance the waste degradation process, thereby increasing the rate of $CH_4$ generation, biogas produced in an anaerobic digester and released directly to the atmosphere without combustion would result in greater $CH_4$ emissions than had the treatment not utilized an anaerobic digester.

## 6.1. Method for Calculating an Illustrative *BAF* Value Applied to Biogenic Emissions Resulting from Combustion of Biogas from Wastewater Treatment

The assessment factor equation can be applied to point source biogenic emissions that result from the combustion of biogas from an anaerobic digester used for wastewater treatment. An illustrative method is provided for calculating a *BAF* value that can be applied to point source biogenic emissions from anaerobic digesters used for the treatment of wastewater. Wastewater treatment via an anaerobic digester is the only treatment method that results in point source emissions.

Here the biogenic feedstock is biogas that is collected from an anaerobic digester used for waste water treatment. Biogas combustion, whether biogas is flared or used as a fuel to generate energy, oxidizes the $CH_4$ contained in biogas to $CO_2$. This results in a net reduction of GHG emissions relative to an alternate GHG emissions pathway in which biogas produced through the anaerobic treatment of wastewater is not captured and combusted, but instead is released to the atmosphere as an indirect emission.

In instances where the alternate GHG emissions pathway involves uncontrolled anaerobic treatment of wastewater, use of anaerobic digester systems typically results in substantial net GHG emissions reductions. However, because anaerobic digesters are designed to enhance $CH_4$ generation, poor design, operation, or maintenance of anaerobic digesters can result in significant indirect $CH_4$ emissions. For example, $CH_4$ can leak from a digester cover or can be vented during digester start-ups, shutdowns, and malfunctions (Bogner et al., 2007; EPA, 2008b; Climate Action Reserve, 2013). However, under normal working conditions, GHG emissions from controlled biological treatment in an anaerobic digester are small relative to indirect $CH_4$ emissions from uncontrolled anaerobic storage and treatment systems (Bogner et al., 2007, and references therein).

In instances where the alternative GHG emissions pathway involves aerobic treatment of wastewater, use of an anaerobic digestion system with biogas capture and combustion in most instances would not be expected to result in a net increase of GHG emissions. In these instances, wastewater and sludges either would have decayed aerobically, producing $CO_2$ as the primary decay product (e.g., in a shallow lagoon or in an aeration tank associated with activated sludge wastewater treatment processes), or they would have decayed anaerobically, producing both $CH_4$ and $CO_2$ (e.g., in a deep, open lagoon).

In applying the assessment factor equation (*BAF* = *AVOIDEMIT*; Equation N.1), net GHG emissions reductions from the use of an anaerobic digester (where the generated biogas is collected and destroyed) are accounted for in the *AVOIDEMIT* term. In practice, as applied here, the *AVOIDEMIT* term is a ratio expressed in units of $CO_2$e avoided (i.e., the emissions, in $CO_2$e, resulting from an alternative wastewater treatment scenario of aerobic, anaerobic, or a combination of aerobic and anaerobic treatment) per units of $CO_2$e removed via combustion (i.e., the emissions, in $CO_2$e, of the biogas generated in an anaerobic digester—accounting for biogas collection and combustion efficiencies). For the biogas feedstock collected from the treatment of wastewater in an anaerobic digester, the *AVOIDEMIT* term can be conceptually expressed using the simplified ratio of:

$$\boldsymbol{AVOIDMENT = 1 - \frac{(\textbf{emissions from treatment alternative to an anaerobic digester})}{(\textbf{emissions from treatment in an anaerobic digester})}} \quad \textbf{(EQ N.56)}$$

### 6.1.1. Calculating the Numerator

In computing *AVOIDEMIT*, the numerator (i.e., emissions from a treatment alternative to an anaerobic digester) can be calculated by building upon the methods developed for EPA by RTI International (2010). To compute the emissions profile of the treatment method that is alternate to an anaerobic digester, the $CO_2$e emissions resulting from the alternate treatments of wastewater and wastewater sludge must be summed:

$$\boldsymbol{AVOIDEMIT}\ \textbf{numerator} = \mathbf{CO_2WW + CO_2S + (GWP_{CH_4} \times (CH_4WW + CH_4S))} \qquad \textbf{(EQ. N.267)}$$

Where:

| | |
|---|---|
| $CO_2WW$ | = $CO_2$ emission rate from wastewater treatment (MT $CO_2$/year), see Equation N.58. |
| $CO_2S$ | = $CO_2$ emission rate from wastewater sludge treatment (MT $CO_2$/year), see Equation N.61. |
| $GWP_{CH_4}$ | = Global warming potential for methane, 25 (IPCC, 2007) |
| $CH_4WW$ | = $CH_4$ emission rate from wastewater treatment (MT $CH_4$/year), see Equation N.59. |
| $CH_4S$ | = $CH_4$ emission rate from wastewater sludge treatment (MT $CO_2$/year), see Equation N.62. |

$CO_2$ and $CH_4$ emissions from the aerobic treatment of wastewater can be calculated using the following two equations:

$$\mathbf{CO_2WW = 10^{-6} \times OpHrs \times Q_{ww} \times OD \times Eff_{OD} \times \frac{44}{32} \times \left[\left(1 - MCF_{WW} \times BG_{CH_4}\right)(1 - \boldsymbol{\lambda})\right]}$$

**(EQ. N.278)**

$$\mathbf{CH_4WW = 10^{-6} \times OpHrs \times Q_{ww} \times OD \times Eff_{OD} \times \frac{16}{32} \times \left[\left(MCF_{WW} \times BG_{CH_4}\right)(1 - \boldsymbol{\lambda})\right]}$$

**(EQ. N.59)**

Where:

| | |
|---|---|
| $CO_2WW$ | = $CO_2$ emission rate (MT $CO_2$/year). |
| $CH_4WW$ | = $CH_4$ emission rate (MT $CH_4$/year). |
| $10^{-6}$ | = Units conversion factor (MT/g). |
| OpHrs | = Hours wastewater treatment system is operated per year. |
| $Q_{ww}$ | = Wastewater influent flow rate ($m^3$/hr). |
| OD | = Oxygen demand[50] of influent wastewater to the biological treatment unit (mg/L = g/$m^3$). |
| $Eff_{OD}$ | = Oxygen demand removal efficiency of the biological treatment unit. |
| 44/32 | = Molar mass ratio of $CO_2$ to $O_2$; representing the conversion factor for maximum $CO_2$ generation per unit of oxygen demand. |
| 16/32 | = Molar mass ratio of $CH_4$ to $O_2$; representing the conversion factor for maximum $CH_4$ generation per unit of oxygen demand. |
| $MCF_{ww}$ | = $CH_4$ correction factor for wastewater treatment unit, indicating the fraction of the influent oxygen demand that is converted anaerobically in the wastewater treatment unit.[51] |
| $BG_{CH_4}$ | = Fraction of C as $CH_4$ in generated biogas (default is 0.65). |
| λ | = Sludge biomass yield, expressed as g C converted to sludge per g C consumed in the wastewater treatment process (see Equation N.60). |

The variable representing sludge biomass yield (λ) in Equations N.58 and N.59 is an estimate of the net sludge generated from the wastewater treatment process, as calculated with Equation N.60.

$$\boldsymbol{\lambda = \frac{Q_s \times MLVSS_s \times 0.53}{Q_{ww} \times OD \times Eff_{od} \times \frac{12}{32}}}$$ **(EQ. N.60)**

Where:

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[50] Determined as either the 5-day biochemical oxygen demand (BOD5) or the chemical oxygen demand (COD). The BOD5 and COD are two measures of the amount of degradable organic content in wastewater.
[51] $MCF_{ww}$ value ranges from 0 to 0.8 (see Table N-4). A $MCF_{ww}$ value of zero (no $CH_4$ emissions) is assigned to well-managed aerobic decomposition systems.

$\lambda$ = Sludge biomass yield, expressed as g C converted to sludge per g C consumed in the wastewater treatment process.

$Q_s$ = Wastewater sludge flow rate ($m^3$/hr).

$Q_{ww}$ = Wastewater influent flow rate ($m^3$/hr).

$MVVSS_s$ = Mixed liquor volatile suspended solids concentration of the waste sludge stream (mg/L).

OD = Oxygen demand of influent wastewater to the biological treatment unit (mg/L).

$Eff_{OD}$ = Oxygen demand removal efficiency of the biological treatment unit.

0.53 = Correction factor for carbon content of the sludge biomass.[52]

12/32 = Molar mass ratio of C to $O_2$; representing the conversion factor for maximum C consumption per unit of oxygen demand.

If the flow rate of the sludge waste stream is not directly measured then estimated representative values for sludge biomass yield can be used as an alternative to Equation N.60. Illustrative representative values for sludge biomass yield are specific to the treatment system used (Table N-13).

**Table N-13. Illustrative Representative Values for Methane Correction Factor (MCF) and Biomass Yield (λ) by Treatment System for both Wastewater and Sludge Treatment Processes (from RTI International, 2010).**

| Wastewater Treatment Process | MCF | λ |
|---|---|---|
| Aerated treatment process (e.g., activated sludge system), well managed | 0 | 0.65 |
| Aerated treatment process, overloaded (i.e., anoxic areas) | 0.3 | 0.45 |
| Anaerobic treatment process (e.g., anaerobic digester) | 0.8 | 0.1 |
| Facultative lagoon, shallow (< 2 m deep) | 0.2 | 0 |
| Facultative lagoon, deep (≥ 2 m deep) | 0.8 | 0 |
| **Sludge Treatment Process** | | |
| Aerobic sludge digestion | 0 | Use λ from wastewater treatment process |
| Anaerobic sludge digestion (e.g., anaerobic digester) | 0.8 | |

Emissions from the treatment of solids (i.e., sludge generated in the wastewater treatment system), whether sludge is treated aerobically or anaerobically, can be calculated using the following equations:

[52] Carbon accounts for 53% of the sludge biomass weight (dry basis).

$$CO_2S = 10^{-6} \times OpHrs \times Q_{ww} \times OD \times Eff_{OD} \times \frac{44}{32} \times \left[\lambda\left(1 - MCF_S \times BG_{CH_4}\right)\right] \quad \textbf{(EQ. N.61)}$$

$$CH_4S = 10^{-6} \times OpHrs \times Q_{ww} \times OD \times Eff_{OD} \times \frac{16}{32} \times \left[\lambda\left(MCF_S \times BG_{CH_4}\right)\right] \quad \textbf{(EQ. N.62)}$$

Where:

$CO_2S$ = $CO_2$ emission rate (MT $CO_2$/year).

$CH_4S$ = $CH_4$ emission rate (MT $CH_4$/year).

$10^{-6}$ = Units conversion factor (MT/g).

OpHrs = Hours wastewater treatment system is operated per year.[53]

$Q_{ww}$ = Wastewater influent flow rate ($m^3$/hr).

OD = Oxygen demand[54] of influent wastewater to the biological treatment unit (mg/L = g/$m^3$).

$Eff_{OD}$ = Oxygen demand removal efficiency of the biological treatment unit.

44/32 = Molar mass ratio of $CO_2$ to $O_2$; representing the conversion factor for maximum $CO_2$ generation per unit of oxygen demand.

16/32 = Molar mass ratio of $CH_4$ to $O_2$; representing the conversion factor for maximum $CH_4$ generation per unit of oxygen demand.

λ = Sludge biomass yield, expressed as g C converted to sludge per g C consumed in the wastewater treatment process (see Equation N.60).

$MCF_S$ = $CH_4$ correction factor for sludge digestion, indicating the fraction of the treated sludge that is converted anaerobically in the wastewater treatment unit.[55]

$BG_{CH_4}$ = Fraction of C as $CH_4$ in generated biogas (default is 0.65).

### 6.1.2. Calculating the Denominator

In computing *AVOIDEMIT*, the denominator (i.e., emissions from the treatment of wastewater and sludge in an anaerobic digester such that the generated biogas is captured and combusted) can be calculated by building upon the methods developed for EPA by RTI International (2010). To compute the emissions profile associated with the anaerobic treatment of wastewater and wastewater sludge in an anaerobic digester, the $CO_2e$ emissions resulting from these treatments

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[53] A wastewater system may operate continuously except for when it is down for maintenance, 10% of the year (e.g., 365 × 24 × 0.9 = 8760 hours).
[54] Determined as either the 5-day biochemical oxygen demand (BOD5) or the chemical oxygen demand (COD). The BOD5 and COD are two measures of the amount of degradable organic content in wastewater.
[55] $MCF_{ww}$ value ranges from 0 to 0.8 (see table 3-1, RTI International, 2010). A $MCF_{ww}$ value of zero (no $CH_4$ emissions) is assigned to well-managed aerobic decomposition systems.

must be summed while simultaneously accounting for biogas destruction efficiency (via combustion) and biogas collection efficiency:

$$\textbf{\textit{AVOIDEMIT} denominator} = CO_2WW + CO_2S + 25\times((CH_4WW - CH_4WWD + CH_4WWL) + (CH_4S - CH_4SD + CH_4SL)) + (CH_4WWD \times 44/16) + (CH_4SD \times 44/16) \qquad \textbf{(EQ. N.63)}$$

Where:

$CO_2WW$ = $CO_2$ emission rate from wastewater treatment (MT $CO_2$/year), see Equation N.58.

$CO_2S$ = $CO_2$ emission rate from wastewater sludge treatment (MT $CO_2$/year), see Equation N.61.

$CH_4WW$ = $CH_4$ emission rate from wastewater treatment (MT $CH_4$/year), see Equation N.59.

$CH_4WWD$ = $CH_4$ emission rate from wastewater treatment, adjusted for biogas destruction efficiency (MT $CH_4$/year).[56]

$CH_4WWL$ = $CH_4$ emission rate from wastewater treatment, adjusted for biogas collection efficiency (MT $CH_4$/year).[57]

$CH_4S$ = $CH_4$ emission rate from wastewater sludge treatment (metric ton $CO_2$/year), see Equation N.62.

$CH_4SD$ = $CH_4$ emission rate from wastewater sludge treatment, adjusted for biogas destruction efficiency (MT $CH_4$/year).[58]

$CH_4SL$ = $CH_4$ emission rate from wastewater sludge treatment, adjusted for biogas collection efficiency (MT $CH_4$/year).[59]

44/16 = molecular weight ratio of $CO_2$ to $CH_4$.

After solving for the numerator and the denominator of the *AVOIDEMIT* term, the *BAF* can be calculated using Equation N.1. See Section 6.2 for an illustrative example calculation of the numerator and denominator in the *AVOIDEMIT* term and its subsequent application in estimating a *BAF* for the management of wastewater and wastewater sludge.

Several parameters are presented and used in the equations in the remainder of this section.

[56] Assuming biogas destruction efficiency is 0.99 then $CH_4WWD = 0.99 \times CH_4WW$.
[57] Assuming biogas collection efficiency is 0.99 then $CH_4WWL = (1\text{-}0.99) \times CH_4WW$.
[58] Assuming biogas destruction efficiency is 0.99 then $CH_4SD = 0.99 \times CH_4S$.
[59] Assuming biogas collection efficiency is 0.99 then $CH_4SL = (1\text{-}0.99) \times CH_4S$.

Table N-14presents the parameters used, typical or default values, ranges presented in the literature, and references.

**Table N-14. Summary of Parameters Used When Calculating an Illustrative *BAF* for Wastewater Treatment.**

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **Oxygen demand of the influent wastewater to the biological treatment unit** | OD | – | – | mg/L or $g/m^3$ | Determined through either the BOD5 or the COD tests. | NA |
| **Oxygen demand removal efficiency of the biological treatment unit** | $Eff_{OD}$ | – | – | Decimal percent | Determined by the wastewater treatment facility. | NA |
| **Fraction of C as $CH_4$ in generated biogas** | $BG_{CH_4}$ | 0.65 | 0.40 to 0.70 | Decimal percent | | EPA, 2013c |
| **Sludge biomass yield** | λ | – | 0 to 0.65 | Expressed as g C converted to sludge per g C consumed in the wastewater treatment process | See Equation N.60 | RTI International, 2010 |
| **Aerated treatment process (e.g., activated sludge system), well managed** | λ | 0.65 | | | | Muller et al., 2003; Munz, 2008; Choubert et al., 2009; RTI International, 2010 |
| **Aerated treatment process, overloaded (i.e., anoxic areas)** | λ | 0.45 | | | | Muller et al., 2003; Ammary, 2004; Munz, 2008; Choubert et al., 2009; RTI International, 2010 |
| **Anaerobic treatment process (e.g., anaerobic digester)** | λ | 0.1 | | | | Ammary, 2004; Low and Chase, 1999; RTI International, 2010 |
| **Facultative lagoon, shallow (< 2 m deep)** | λ | 0 | | | | RTI International, 2010 |
| **Facultative lagoon, deep (≥ 2 m deep)** | λ | 0 | | | | RTI International, 2010 |

| Parameter Description | Symbol | Value | Range | Units | Comments | Reference (for value column) |
|---|---|---|---|---|---|---|
| **WW: Aerated treatment process (e.g., activated sludge system), well managed** | $MCF_{WW}$ | 0 | 0 to 0.8 | Fraction | Indicating the fraction of the influent oxygen demand that is converted anaerobically in the wastewater treatment unit | IPCC, 2006; RTI International, 2010 |
| **WW: Aerated treatment process, overloaded (i.e., anoxic areas)** | $MCF_{WW}$ | 0.3 | | | | |
| **WW: Anaerobic treatment process (e.g., anaerobic digester)** | $MCF_{WW}$ | 0.8 | | | | |
| **WW: Facultative lagoon, shallow (< 2 m deep)** | $MCF_{WW}$ | 0.2 | | | | |
| **WW: Facultative lagoon, deep (≥ 2 m deep)** | $MCF_{WW}$ | 0.8 | | | | |
| **Sludge: Aerobic sludge digestion** | $MCF_S$ | 0 | | | | |
| **Sludge: Anaerobic sludge digestion (e.g., anaerobic digestion)** | $MCF_S$ | 0.8 | | | | |
| **Biogas destruction efficiency** | DE; included in $CH_4WWD$ and $CH_4SD$ | 0.99 | 0.90 to 0.9977 | Decimal percent | | EPA, 2010d; EPA, 2011a; EPA, 2013c |
| **Biogas collection efficiency** | CE; included in $CH_4WWL$ and $CH_4SD$ | 0.99 | 0.70 to 0.99 | Decimal percent | 0.99 is used for an enclosed vessel, anaerobic sludge digester | EPA, 2010d, Table 6-1 |

## 6.2. Example *AVOIDEMIT* and *BAF* Calculations for the Collected Biogas from Treatment of Wastewater and Wastewater Sludge

This example calculation of the *BAF* is for a hypothetical wastewater treatment system that uses an anaerobic digester to treat wastewater and another anaerobic digester to treat sludge. In this scenario the wastewater treatment system has an average flow rate of 1 million gallons per day (or 157.71 $m^3$/hr), an inlet 5-day biochemical oxygen demand (BOD5) of 500 g/$m^3$, and the treatment system has a 95% BOD5 removal efficiency.

### Step 1: Calculating the Numerator

To calculate the numerator of *AVOIDEMIT*, the emissions from a treatment alternative to an anaerobic digester, must be computed. In this hypothetical example, the treatment alternative to an anaerobic digester would be a shallow (< 2 m deep), facultative lagoon. For this example, it is assumed that the wastewater system would operate continuously throughout the year (365 × 24 = 8760 hours). Treatment of wastewater sludge is assumed to be outside of the lagoon such that there is no sludge biomass yield (λ = 0). Emissions from this alternate treatment pathway can be computed using Equation N.57. However, values to populate Equation N.57 must first be calculated using Equations N.58 and N.59 (because λ = 0, Equations N.61 and N.62 are equal to zero, thus dropping out of Equation N.57).

$CO_2$ emissions from the wastewater treatment system (a shallow, facultative lagoon) are calculated using Equation N.58, ($MCF_{ww}$ = 0.2 and λ = 0):

$$\boldsymbol{CO_2WW = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{44}{32} \times \left[\left(1 - MCF_{WW} \times BG_{CH_4}\right)(1 - \lambda)\right]}$$

$$= 10^{-6} \times 8760 \times 157.71 \times 500 \times 0.95 \times (44/32) \times [(1 - 0.2 \times 0.65)\ (1 - 0)]$$

$$= 785.0167 \text{ MT } CO_2 \text{ per year}$$

$CH_4$ emissions from the wastewater treatment system (a shallow, facultative lagoon) are calculated using Equation N.59, ($MCF_{ww}$ = 0.2 and λ = 0):

$$\boldsymbol{CH_4WW = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{16}{32} \times \left[\left(MCF_{WW} \times BG_{CH_4}\right)(1 - \lambda)\right]}$$

$$= 10\text{-}6 \times 8760 \times 157.71 \times 500 \times 0.95 \times (16/32) \times [(0.2 \times 0.65)(1 - 0)]$$

$$= 42.6550 \text{ MT } CH_4 \text{ per year.}$$

The numerator of *AVOIDEMIT* can now be solved using Equation N.57:

$$AVOIDEMIT \text{ numerator} = CO_2WW + CO_2S + (25 \times (CH_4WW + CH_4S))$$

$$= 785.0108 + 0 + 25 \times (42.6547 + 0)$$

$$= 1{,}851.38 \text{ MT } CO_2e \text{ per year}$$

### Step 2: Calculating the Denominator

To calculate the denominator of *AVOIDEMIT*, the emissions from wastewater and wastewater sludge treatment using an anaerobic digester, must be computed. Emissions from this treatment pathway can be computed using Equation N.63. However, values to populate Equation N.63 must first be calculated using Equations N.58, N.59, N.61, and N.62.

$CO_2$ emissions from the wastewater treatment system (an anaerobic digester) are calculated using Equation N.58, ($MCF_{ww}$ = 0.8 and λ = 0.1):

$$\boldsymbol{CO_2WW = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{44}{32} \times \left[\left(1 - MCF_{WW} \times BG_{CH_4}\right)(1-\lambda)\right]}$$

$$\mathbf{= 10^{-6} \times 8760 \times 157.71 \times 500 \times 0.95 \times (44/32) \times[(1 - 0.8 \times 0.65)(1 - 0.1)]}$$

$$\mathbf{= 389.8014\ MT\ CO_2\ per\ year}$$

$CH_4$ emissions from the wastewater treatment system (an anaerobic digester) are calculated using Equation N.59, ($MCF_{ww}$ = 0.8 and λ = 0.1):

$$\boldsymbol{CH_4WW = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{16}{32} \times \left[\left(MCF_{WW} \times BG_{CH_4}\right)(1-\lambda)\right]}$$

$$\mathbf{= 10^{-6} \times 8760 \times 157.71 \times 500 \times 0.95 \times (16/32) \times[(0.8 \times 0.65)(1 - 0.1)]}$$

$$\mathbf{= 153.558\ MT\ CH_4\ per\ year.}$$

$CO_2$ emissions from the wastewater sludge treatment system (an anaerobic digester) are calculated using Equation N.61, ($MCF_S$ = 0.8 and λ = 0.1):

$$\boldsymbol{CO_2S = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{44}{32} \times \left[\lambda\left(1 - MCF_S \times BG_{CH_4}\right)\right]}$$

$$\mathbf{= 10^{-6} \times 8760 \times 157.71 \times 500 \times 0.95 \times (44/32) \times[0.1(1 - 0.8 \times 0.65)]}$$

$$\mathbf{= 43.3113\ MT\ CO_2\ per\ year}$$

$CH_4$ emissions from the wastewater sludge treatment system (an anaerobic digester) are calculated using Equation N.62, ($MCF_S$ = 0.8 and λ = 0.1):

$$\boldsymbol{CH_4S = 10^{-6} \times \frac{HO}{yr} \times Q_{WW} \times OD \times Eff_{OD} \times \frac{16}{32} \times \left[\lambda\left(MCF_S \times BG_{CH_4}\right)\right]}$$

$$\mathbf{= 10^{-6} \times 8760 \times 157.71 \times 500 \times 0.95 \times (16/32) \times[0.1(0.8 \times 0.65)]}$$

$$\mathbf{= 17.0620\ MT\ CH_4\ per\ year.}$$

Next, the calculated emissions associated with treatment of wastewater and wastewater sludge in an anaerobic digester can be used to populate Equation N.63:

$$\textbf{\textit{AVOIDEMIT} denominator} =$$

$$\mathbf{CO_2WW + CO_2S}$$

$$\mathbf{+\ 25\times\ ((CH_4WW - CH_4WWD + CH_4WWL) + (CH_4S - CH_4SD + CH_4SL))}$$

$$\mathbf{+\ CH_4WWD \times 44/16 + CH_4SD \times 44/16}$$

Assuming biogas destruction efficiency is 0.99, then:

- $CH_4WWD = 0.99 \times CH_4WW$, and
- $CH_4SD = 0.99 \times CH_4S$.

Assuming biogas collection efficiency is 0.99, then:

- $CH_4WWL = (1–0.99) \times CH_4WW$, and
- $CH_4SL = (1–0.99) \times CH_4S$.

Applying these assumptions, the denominator of *AVOIDEMIT* can be expressed as:

$$\textbf{\textit{AVOIDEMIT} denominator} = \mathbf{CO_2WW + CO_2S}$$

$$\mathbf{+\ 25\times\ (CH_4WW - (0.99 \times CH_4WW) + ((1–0.99) \times CH_4WW))}$$

$$\mathbf{+\ 25\times\ (CH_4S - (0.99 \times CH_4S) + ((1–0.99) \times CH_4S))}$$

$$\mathbf{+\ (0.99 \times CH_4WW \times 44/16) + (0.99 \times CH_4S \times 44/16)}$$

$$\textbf{\textit{AVOIDEMIT} denominator} = \mathbf{389.8014 + 43.3113}$$

$$\mathbf{+\ 25\times\ (153.558 - (0.99 \times 153.558) + ((1 - 0.99) \times 153.558))}$$

$$\mathbf{+\ 25\times\ (17.0620 - (0.99 \times 17.0620) + ((1 - 0.99) \times 17.0620))}$$

$$\mathbf{+\ (0.99 \times 153.558 \times 44/16) + (0.99 \times 17.0620 \times 44/16)}$$

$$\mathbf{= 982.9357\ MT\ CO_2e\ per\ year.}$$

### Step 3: Calculating the *BAF* Value

After solving for the numerator and the denominator of the *AVOIDEMIT* term associated with wastewater management, the *BAF* can be calculated using Equations N.1 and N.2:

$$\boldsymbol{BAF = AVOIDEMIT}$$

$$\mathbf{= 1 - \frac{1{,}851.38}{982.9357}}$$

$$\mathbf{= –0.88}$$

A negative calculated *BAF* value, such as that above, indicates that a biogas feedstock produced in an anaerobic digester from the treatment of wastewater and wastewater sludge, and used by a stationary source results in net $CO_2e$ emissions reductions.

## 6.3. Sensitivity Analysis for Wastewater Treatment

A simple sensitivity analysis on the key parameters in the wastewater treatment methodology is presented in Table N-15 for the actual fate of wastewater treatment in an anaerobic digester and the alternate fate of placing the waste in a shallow, facultative lagoon. Key parameters impacting the *BAF* include the DE and CE for the anaerobic digester, and the GWP for $CH_4$ (21, 25, and 28). In each of the six analyses, the DE of the biogas was adjusted between 95% and 99%, and the CE was adjusted to 75%, 90%, and 99%, representing a range of low to high performing biogas collection system efficiencies. Sources for the parameter values used here can be found in Table N-14of Section 6.1.2. The inputs used in the analyses are equivalent to those shown in the example calculations in Section 6.2 of this appendix.

The *BAF* values are positive when performing the calculations with a CE lower than 78% despite the DE value (95% or 99%) used. Higher CEs yield negative *BAF* values regardless of the GWP, indicating that wastewater management through anaerobic digestion may be a better treatment option with respect to $CO_2$ and $CH_4$ emissions pathways. The turning point for the *BAF* values with respect to either a 95% of 99% DE is presented in Analyses 7 through 18. Note that, in order for a net $CO_2e$ emissions reduction to occur, the anaerobic digester may require a CE of at least 80% to 85%.

**Table N-15. Sensitivity Analysis for Wastewater Treatment.**

| Analysis | Key Parameter and Value | | *BAF* | | |
|---|---|---|---|---|---|
| | CE | DE | GWP=21 | GWP=25 | GWP=28 |
| **1** | 0.75 | 0.99 | 0.081 | 0.077 | 0.075 |
| **2** | 0.75 | 0.95 | 0.140 | 0.142 | 0.144 |
| **3** | 0.85 | 0.99 | −0.143 | −0.172 | −0.191 |
| **4** | 0.85 | 0.95 | −0.053 | −0.069 | −0.079 |
| **5** | 0.99 | 0.99 | −0.734 | −0.884 | −0.993 |
| **6** | 0.99 | 0.95 | −0.537 | −0.631 | −0.698 |
| **7**[a] | 0.79 | 0.99 | 0.003 | | |
| **8**[a] | 0.80 | 0.99 | −0.019 | | |
| **9**[b] | 0.78 | 0.99 | | 0.015 | |
| **10**[b] | 0.79 | 0.99 | | −0.008 | |
| **11**[c] | 0.78 | 0.99 | | | 0.009 |
| **12**[c] | 0.79 | 0.99 | | | −0.016 |
| **13**[d] | 0.82 | 0.95 | 0.013 | | |
| **14**[d] | 0.83 | 0.95 | −0.008 | | |
| **15**[e] | 0.82 | 0.95 | | 0.005 | |
| **16**[e] | 0.83 | 0.95 | | −0.019 | |
| **17**[f] | 0.81 | 0.95 | | | 0.023 |
| **18**[f] | 0.82 | 0.95 | | | −0.001 |

[a] The point at which the *BAF* changes from negative to positive with a GWP of 21 and DE = 0.99.
[b] The point at which the *BAF* changes from negative to positive with a GWP of 25 and DE = 0.99.

[c] The point at which the *BAF* changes from negative to positive with a GWP of 28 and DE = 0.99.
[d] The point at which the *BAF* changes from negative to positive with a GWP of 21 and DE = 0.95.
[e] The point at which the *BAF* changes from negative to positive with a GWP of 25 and DE = 0.95.
[f] The point at which the *BAF* changes from negative to positive with a GWP of 28 and DE = 0.95.
Note: References for the key parameters and values are presented in Section 6.1.

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