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CENTER for BIOLOGICAL DIVERSITY Because life is good

March 17, 2017

Via U.S. Mail

California Energy Commission
Dockets Office, MS-4
Re: Docket No. 16-OIR-05
1516 Ninth Street
Sacramento, CA 95814-5512

Re: Docket No. 16-OIR-05: Electronic Copies of References Cited in the Center for Biological Diversity's Comments on Pre-Rulemaking Updates to the Power Source Disclosure Regulations (AB 1110 Implementation)

Dear Commissioners:

Enclosed please find a CD containing PDF copies of references cited in the Center for Biological Diversity's March 15, 2017 comments on pre-rulemaking updates to the Power Source Disclosure regulations (Docket No. 16-OIR-05). Please include these references in the record of proceedings for this matter.

Sincerely,

Kevin P. Bundy
Senior Attorney

Encl.: CD of References Cited

Kevin Bundy, Climate Legal Director and Senior Attorney . 1212 Broadway, Suite 800 . Oakland, CA 94612
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References Cited
Center for Biological Diversity Comments
Re: Docket No. 16-OIR-05: Pre-Rulemaking Updates to the Power Source Disclosure Regulations (AB 1110 Implementation)
March 15, 2017

(Copies of references to be provided on CD under separate cover)

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SCIENTIFIC REPORTS

OPEN

Forest soil carbon is threatened by intensive biomass harvesting

David L. Achat[1], Mathieu Fortin[2,3], Guy Landmann[4], Bruno Ringeval[1] & Laurent Augusto[1]

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Forests play a key role in the carbon cycle as they store huge quantities of organic carbon, most of which is stored in soils, with a smaller part being held in vegetation. While the carbon storage capacity of forests is influenced by forestry, the long-term impacts of forest managers' decisions on soil organic carbon (SOC) remain unclear. Using a meta-analysis approach, we showed that conventional biomass harvests preserved the SOC of forests, unlike intensive harvests where logging residues were harvested to produce fuelwood. Conventional harvests caused a decrease in carbon storage in the forest floor, but when the whole soil profile was taken into account, we found that this loss in the forest floor was compensated by an accumulation of SOC in deeper soil layers. Conversely, we found that intensive harvests led to SOC losses in all layers of forest soils. We assessed the potential impact of intensive harvests on the carbon budget, focusing on managed European forests. Estimated carbon losses from forest soils suggested that intensive biomass harvests could constitute an important source of carbon transfer from forests to the atmosphere (142–497 Tg-C), partly neutralizing the role of a carbon sink played by forest soils.

Forests contain more carbon than the atmosphere[1–3] and, as such, are a major component of the carbon cycle on Earth. Compared with other terrestrial ecosystems, forests store some of the largest quantities of carbon per surface area of land[4]. As a result, the carbon storage capacity of land could be improved through afforestation, or decreased by deforestation[4,5]. While such land-use changes have well-known consequences on land carbon, the long-term impact of forest managers' decisions remains unclear relative to the global carbon cycle, and strategies regarding carbon by management of forests are conflicting[6]. One school of thought proposes that forests should be allowed to accumulate carbon in the long-term because old-growth forests are active carbon sinks[7]. An alternative approach proposes an intensification of wood harvesting to replace fossil carbon in the production of manufactured objects and energy[2]. The best strategy for managing forest carbon as a means of mitigating climate change is still a controversial issue[1]. Indeed, while collecting more biomass can help in the substitution of fossil energy by fuelwood, it also results in the reduction of carbon stocks sequestered in trees[8], and in turn, a possible reduction in the future rate of carbon accumulation, due to the removal of the largest trees which have the highest accumulation rates[9]. Furthermore, although it has been established that forest management can modify stocks of soil organic carbon (SOC)[10], the extent to which the intensity and frequency of biomass harvests might be deleterious to forest SOC remains unclear because of the difficulty in monitoring this compartment of the ecosystem accurately[6,10], and due to the high number of factors involved[11]. The complexity of this question has led to many uncertainties[1] and inconclusive debates[12–14].

Here we report a global assessment of the consequences of different management practices on soil organic carbon storage in forests. We focused on soils because they are generally the largest carbon pools of forest ecosystems[15], are less exposed to climatic extremes than trees[16], and because little is known about their responses to changes in management or the environment[2,10]. The assemblage of results published on this topic in peer-reviewed journals yielded large databases comprising experimental forest sites distributed worldwide. In each forest, different practices of biomass harvest were tested, and their

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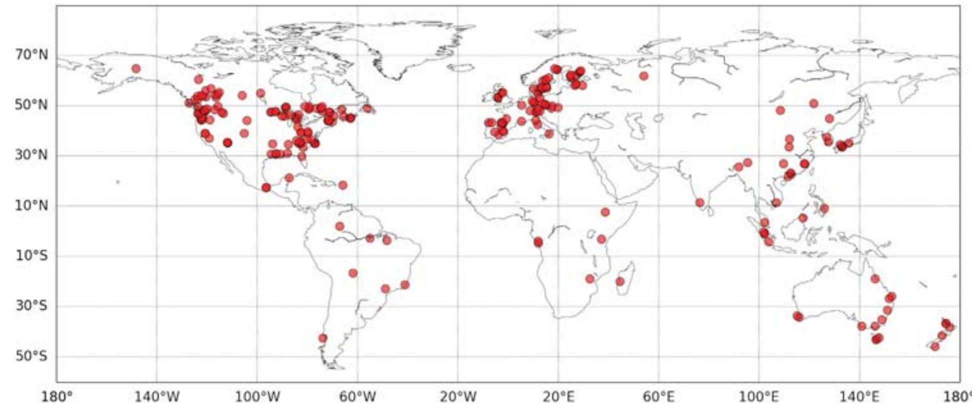

Figure 1. Distribution of the sites used in this meta-analysis on the effects of conventional and intensive harvests on soil C stocks. See more details on the geographical location of the sites in Fig. S1. Map created in Python Language version 2.7 (Python Software Foundation; www.python.org), using the *basemap* package (https://pypi.python.org/pypi/basemap/1.0.7) of the *matplotlib* library (http://matplotlib.org).

consequences on the soil carbon pool were monitored. We quantified the effects on SOC of the three main strategies in terms of carbon management: i) *carbon sequestration in forests*, based on unharvested forests, ii) *conventional harvests* of tree stems, used in most managed forests, and iii) *intensive harvests*, based on the collection of tree stems and logging residues (stumps, branches, foliage, and sometimes forest floor racking) to produce fuelwood[2,17]. Collection of trees –both in conventional and intensive harvests– can be incomplete or total. Thus, we additionally took into account if conventional, or intensive, harvests were carried out during a thinning (the felling and logging of a proportion of trees to promote the growth of the residual trees[18]) or a clear-cutting (the felling and logging of all trees, followed by seedling planting, sowing, or natural forest regeneration[19]). Because in practice most intensive harvests were done at clear-cutting, we studied possible differences between thinning and clear-cutting for conventional harvests only.

We compiled data from 284 forest sites and built two datasets related to conventional harvests and intensive harvests, respectively (see Methods). Although the majority of these forests are located in the Northern hemisphere, in North America and Europe, under temperate or cold climates (Tables S1 and S2 in Supplementary Information), they are distributed worldwide, representing all types of managed forests (Fig. 1; Fig. S1 in Supplementary Information). The consolidated datasets included a total of 2,028 values of SOC change in different soil layers up to 135 years after biomass harvesting (Figs S2 and S3). Soil layers were grouped into four classes depending on their depth (the organic layer above the mineral soil profile: forest floor "F"; top, mid and deep mineral soil layers: "T", "M", and "D"). Cumulated soil layers were also examined ("TM", "TMD", "FT", "FTM" and "FTMD").

Results

Conventional harvests. The impact assessment of conventional harvests, as compared with unharvested forests (first dataset), indicated that around 22% of SOC in the F layer was lost due to harvesting operations (Fig. 2A; Fig. S4A). This loss of carbon in forest floors appeared to be long lasting as it was still clearly apparent a decade after harvesting (Fig. 3A) and possibly required more than half a century to be fully compensated (Fig. 4 and Fig. S2). Surprisingly, there were only slight differences between thinning and clear-cutting (Fig. S4A), except during the first decade when there were higher SOC losses after clear-cutting than after thinning (Fig. 3A and Fig. S2). During the first decade, SOC losses also tended to increase with increasing thinning intensity (Fig. S5). There was, however, no thinning frequency effect or forest age effect.

The response of SOC stocks in the upper mineral layer was clearly different from that of the forest floor. In the T layer, SOC stocks often remained stable (Figs 2A and 3B). However, carbon losses did occur in some cases, especially when this topsoil layer was disturbed as a result of forest clear-cutting with heavy machinery, or *soil preparation* before seedling plantation (Fig. S6A).

Despite an overall non-significant change of carbon storage in the T layer, conventional harvests reduced the carbon stock of the "FT" upper soil by 14% on average as a result of the important loss in the forest floor (Fig. 2A; Fig. S4A). This general decrease of SOC in the upper part of the soil profile (i.e. F+T) was compensated by an accumulation beneath (Fig. 2A; Fig. S4A): when deep layers (D) and above all medium layers (M) were taken into account, the balance of SOC losses *versus* SOC gains was not significantly different from zero (the mean value for the complete FTMD soil profile = −6% SOC).

Intensive harvests. The results obtained from our second dataset indicated that intensive harvests strongly reduced SOC stocks in woody debris (WD) and in the F layer, relative to stem-only harvests

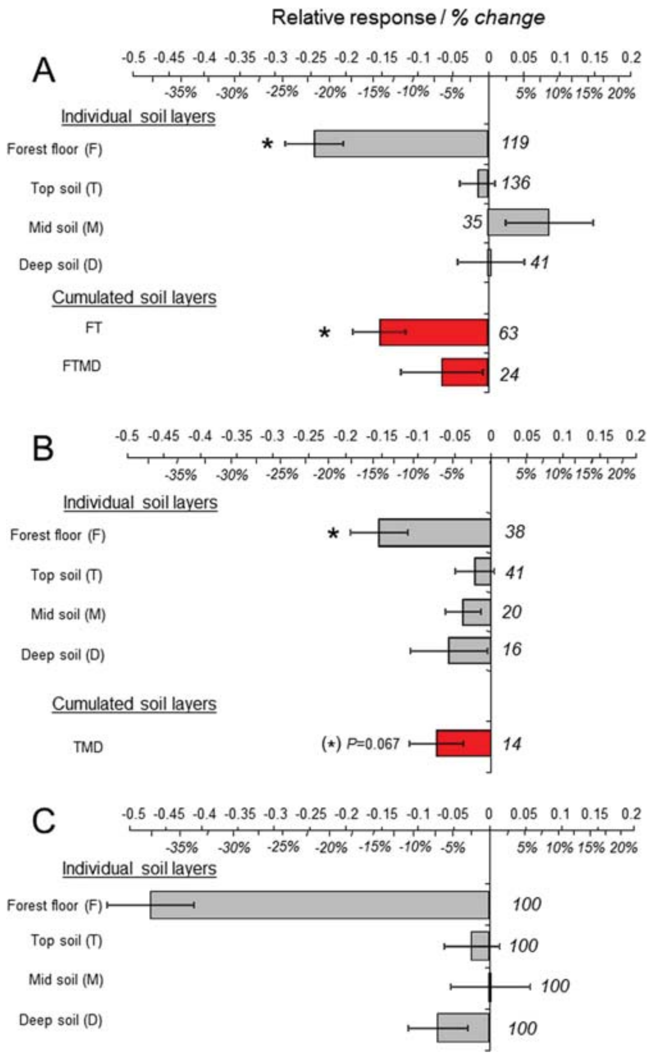

Figure 2. General effects of conventional and intensive harvests on SOC stocks as a function of soil depth (individual soil layers) and in the entire soil profile (cumulated soil layers). (**A**) Effects of conventional harvest (clear-cutting and thinning; means ± standard errors). (**B**) Effects of intensive harvest compared with stem-only harvest (means ± standard errors). (**C**) Combined effects of conventional and intensive harvests. Values are expressed as relative responses: (**A**) log(clear-cutting or thinning harvest/unharvested control) (**B**) log(whole-tree harvest/stem-only harvest) (**C**) log(whole-tree harvest/unharvested control). For the sake of clarity, comparisons between treatments and controls are also presented as the mean arithmetic difference (in italics, expressed in %). Results in (**C**) were obtained using the two datasets (data in **A**,**B**) and a bootstrap resampling method. For each panel, number of case studies (or sites) and number of bootstrap samples are shown in italics to the right of each bar. There were not enough data for FTMD in (**B**). Significant differences between relative responses and 0 are denoted by an asterisk (t test). See more results in Fig. S4.

(Fig. 2B; Fig. S4B). In addition, the entire mineral soil (TMD) was also negatively impacted (Fig. 2B), especially when the forest floor was racked and exported from the forest (Fig. S4B). Unfortunately, published studies containing information for the complete organic plus mineral soil profile (i.e. FTMD) were scarce. This gap in the literature prevented us from directly assessing the effect of intensive harvests on SOC stocks in forests. Nevertheless, because both the organic soil layers (WD and F) and the mineral soil profiles (TMD) showed a clear decrease, a general reduction of the soil carbon stock was likely to occur. After one decade, SOC losses were no longer detected in the topsoil (T), although they were still reported in the F layer (Fig. 3; Fig. S3). There were negative relationships between SOC losses in the F layer and SOC losses in mineral soils ($r^2 = 0.42$–0.61), suggesting transfers of carbon from the forest floor to mineral layers. But, these possible vertical fluxes appeared to be of small magnitude and as such, they could not compensate for SOC losses from mineral soil layers (Fig. 2B), at least during the decade following intensive harvest.

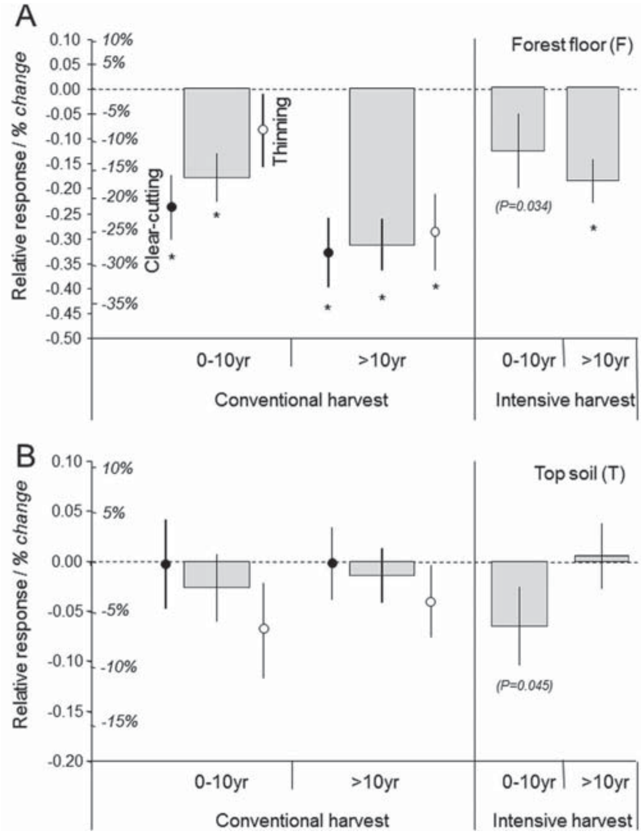

Figure 3. Effects of conventional and intensive harvests on SOC stocks in forest floor (F) and top mineral soil (T) in relation to time elapsed since harvesting. (**A**) Forest floor. (**B**) Top soil. Effects were assessed considering two periods (0–10 years and > 10 years since harvesting; Means ± standard errors). Values are expressed as relative responses: log(clear-cutting or thinning harvest/unharvested control) or log(whole-tree harvest/stem-only harvest). For the sake of clarity, comparisons between treatments and controls are also presented as the mean arithmetic difference (in italics, expressed in %). Number of case studies (or sites) ranged from 16 to 100. Significant differences between relative responses and value 0 are denoted by an asterisk (*t* test). The *P* values in brackets were calculated using all intensive harvest treatments (whole-tree harvest and whole-tree + forest floor harvest). Effects of conventional clear-cutting on C stocks in the forest floor are shown for more time classes in Fig. 4.

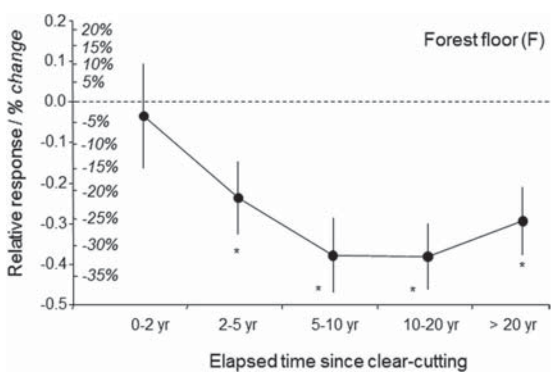

Figure 4. Effects of conventional clear-cutting harvest on SOC stocks in forest floor (F) in relation to time elapsed since harvesting. Effects were assessed considering five periods (0–2, 2–5, 5–10, 10–20 and >20 years since harvesting; Means ± standard errors). Values are expressed as relative responses: log(conventional clear-cutting harvest/unharvested control). For the sake of clarity, comparisons between treatments and controls are also presented as the mean arithmetic difference (in %). Number of case studies (or sites) ranged from 12 to 31. Significant differences between relative responses and value 0 are denoted by an asterisk (*t* test). Temporal changes associated with other harvest types are shown in Supplementary Information (conventional harvest at thinning: Fig. S2; intensive harvest: Fig. S3).

Firstly, we showed that, compared with unharvested forests, conventional harvests of forest biomass had a moderate impact on SOC stocks (Fig. 2A; see *conventional harvests* subsection, above). Here, we found that, contrary to conventional harvests, intensive harvests of forest biomass had a negative impact on SOC stocks (Fig. 2B). At this stage, we tried to test the effects of intensive harvests compared with unharvested forests. However, because published data comparing intensive harvests with unharvested forests are scarce, we were unable to perform statistical tests directly. Instead, we combined our two datasets: *i)* intensive harvests *versus* conventional harvests, and *ii)* conventional harvests *versus* no harvests, using a bootstrap resampling analysis (see Supplementary Methods). Our simulations indicated that intensive harvests were able to induce large SOC losses in comparison with untouched forests (Fig. 2C). SOC losses occurred mainly in the forest floor (−37%) and in deep soil layers (−7%).

Simulation at the European scale. Managed forests in Europe correspond to approximately 142 million hectares with 38% in boreal regions and 62% in temperate regions. Assuming mean SOC stocks in the whole organic plus mineral soil profile to be 277 and 95 Mg-C ha^{-1} respectively for boreal and temperate forests, total SOC stocks in European managed forests represent 23.5 Pg-C (15.1 and 8.4 Pg-C in boreal and temperate forests, respectively). Using the SOC distribution within the soil profile and percentage losses due to intensive harvesting under boreal and temperate climates we obtained in this study, we estimated that the implementation of management strategies based on intensive harvests would cause a loss of organic carbon in forest soils, ranging between 142 and 497 Tg-C, depending on the scenario of management conversion (see Methods). We calculated a mean annual SOC loss over three decades, because in the present study the impacts of intensive harvests have been assessed over a period of 30 years (Fig. S3). Thus, we estimated that the mean annual loss of soil organic carbon in European forests could be between 5 to 17 Tg-C $year^{-1}$.

Discussion

Conventional harvests. Our results, showing a negative effect in the F layer and little overall impact in the T layer, were in accordance with previous findings[19–21] and suggested a negative impact of conventional harvests on forest SOC stocks. Nevertheless, this conclusion is based solely on the most superficial part of soils and investigating the influence of conventional harvests on deeper soil layers led to different conclusions. Our study showed an accumulation of SOC in the M layer which resulted in a net increase of SOC storage in the combined soil layers (TMD; Fig. S4A). When considering the whole soil profile (FTMD), the SOC gain in the mineral layers compensated for the SOC loss observed in the forest floor (Fig. 2A; Fig. S4A). Overall, conventional biomass harvests had no, or only a slightly negative but statistically non-significant, impact on carbon in forest soils when deeper soil layers were also taken into account. This result brings a different perspective than the usual conclusion of decreased SOC stocks when only considering shallow horizons, as usually done in many case studies of the literature.

Our observations on the dynamics of carbon stocks in forest soils following conventional harvests can be explained by several processes. In forests, leaf and wood litterfall is, at best, quantitatively low during the first few years following the removal of standing trees. This reduced flux of organic carbon from aboveground tree biomass to the forest floor has a negative effect on the forest floor stocks[18,20,22,23]. Subsequently, as trees grow, litterfall production increases and enables the recovery of carbon stocks in the forest floor[18,20]. Besides the changes in litterfall production, an increase in organic matter decomposition is also expected to occur and to negatively impact SOC storage. Decomposition rates generally increase in the superficial part of soils immediately after harvests due to soil disturbance and changes in microclimatic conditions (increased solar radiation and, thereby, soil temperature) until canopy closure[21,23,24]. The accumulation of SOC we observed in the M layer was probably due to the inputs of carbon from dead roots immediately following harvesting[25], combined with the migration of dissolved organic carbon from the soil layers above[26]. In addition, in sites where foresters *prepare* the soil before planting (e.g. by soil ploughing), soil disturbance can mix the different soil layers; the forest floor and some logging debris being typically incorporated into the mineral soil[27]. These results demonstrated that, contrary to widely held opinion, conventional harvests have no globally negative impact on organic carbon stocks of forest soils.

Intensive harvests. Then, we investigated the extent to which intensifying biomass harvests by exporting the logging residues, to supply fuelwood chains for instance[2], can change the pattern observed with conventional harvests. Similarly to conventional harvests, intensive harvests induced large SOC losses in the F layer. Large SOC losses in the F layer seemed to reduce SOC losses in mineral soils (see negative relationships in Fig. S7D and E), possibly due to the migration of dissolved organic carbon from forest floor decomposition[26] or the mixing of soil layers due to soil preparation[27]. However, at best, this input from the F layer yielded some compensation, but it never reached the stage of SOC accumulation in the M and D layers (see negative relationships between SOC losses in F and in mineral soil layers). It implies that, contrary to conventional harvests, there was usually no complete compensation between organic and mineral soil layers under intensive harvests and the overall impact of intensive removal of forest biomass on SOC stock remained negative. This impact was even more negative when intensive harvests were compared with unharvested forests, such as those of the *old-growth* strategy.

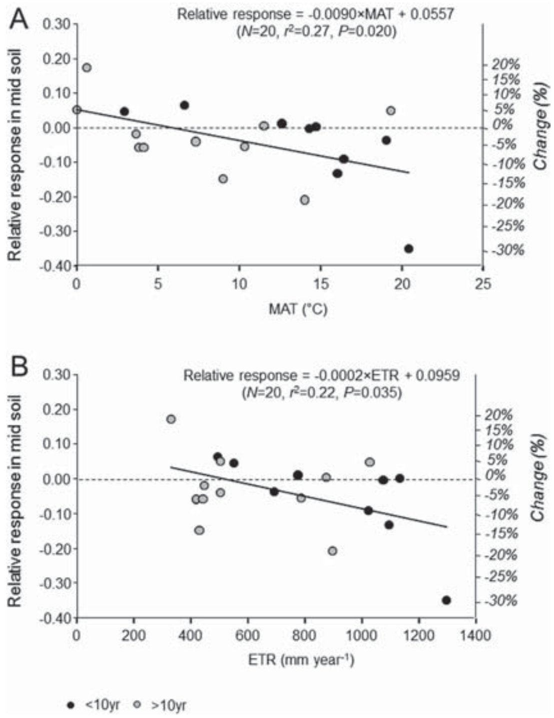

Figure 5. Effects of intensive harvest on C stocks in mid soil (M) related to mean annual temperature (MAT) and effective evapotranspiration (ETR). (**A**) MAT; (**B**) ETR. Values are expressed as relative responses [log(whole-tree harvest/stem-only harvest)]. For the sake of clarity, comparisons between treatments and controls are also presented as the mean arithmetic difference (% higher or lower). A similar trend ($P < 0.1$) was observed between ETR and SOC losses for the topsoil layer also (data not shown).

Heterogeneity of SOC response – environmental factors implied. It is worth stressing that the average effects of forestry practices reported above masked large regional and local disparities. For instance, not all forests showed an accumulation of organic matter in the mid-part of their soil after conventional harvests. This high variability was visible also in topsoils and deep soil layers, but was coherent for a given soil profile because the responses of mid and deep soil layers were influenced by topsoil layer behaviour: when SOC loss or gain was observed in the topsoil, a SOC change in the same direction was generally recorded in deeper soil (Fig. S7A-C). As for conventional harvests, high inter-site variability existed after intensive harvest, but was logical with concomitant losses or stabilities of SOC between mineral soil layers (Fig. S7F).

Such inter-site heterogeneity can be explained by climatic gradients and ecosystem characteristics. In accordance with studies reporting the impact of deforestation[5,28], SOC losses in topsoils due to conventional harvests increased with increasing initial SOC (Fig. S8A and B), the latter being itself partly controlled by climate (Fig. S8C). However, climate was a poor predictor of SOC dynamics after conventional harvests, with no significant difference when comparing tropical, temperate, and boreal forests ($P > 0.1$), perhaps due to insufficient data for tropical forests (Table S1). Climatic influence was clearer for intensive harvests, as demonstrated by the positive relationships between SOC losses and mean annual temperature and evapotranspiration (Fig. 5). Carbon losses were consequently lower under cold climates compared with temperate climates (Fig. 6; not enough data for tropical climates, see Table S1). We interpreted this pattern to be a consequence of soil microclimatic conditions induced by forest management. Indeed, less logging residues were left on site after intensive harvests, leading to microclimatic changes such as an increase in soil temperature in spring and summer due to the role of the debris in regulating temperature variations[29,30]. Sites affected by intensive harvests were probably exposed to larger increases in soil temperature in temperate regions than in cold regions, which in turn could lead to higher increases in SOC decomposition in temperate regions[5,30]. There were not enough sites in the dataset to assess the effect of intensive harvests on SOC under tropical climates (Table S1), but high temperatures in these regions were expected to favour larger increases in soil temperature and consequently higher organic matter decomposition and SOC losses, as observed in temperate regions[5]. This expectation was in line with a recent study which demonstrated a strong relationship between the carbon turnover time and climate in terrestrial ecosystems[31].

Soil type was another factor modifying SOC response to biomass harvests. For instance, highly weathered soils had an accumulation of SOC in their topsoil layer after a conventional harvest (Fig. S6B). Finally, forest composition seemed significantly influencing our results. As already reported in the literature[19], a comparison of hardwood forests with coniferous and mixed forests suggests that the

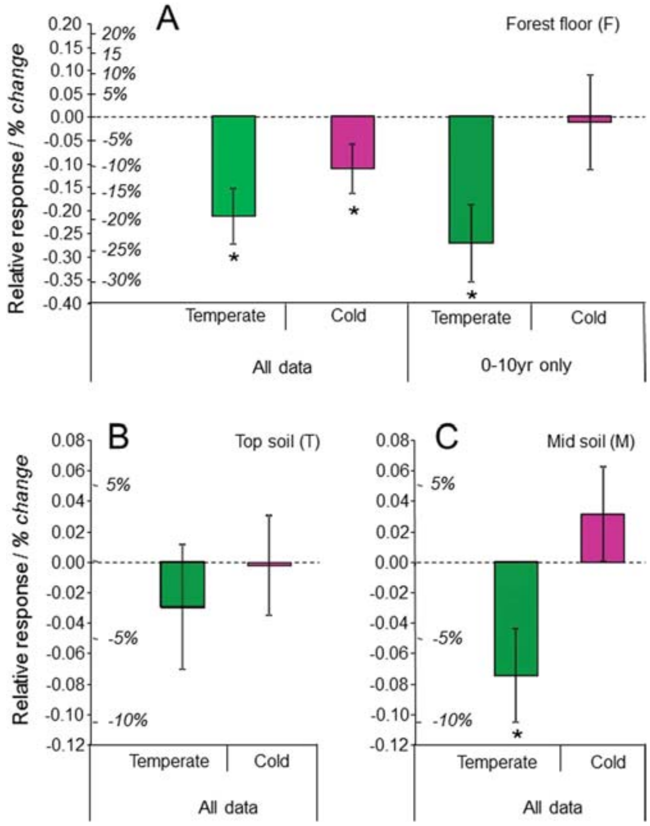

Figure 6. Effect of intensive harvest on C stocks in the forest floor (F), top soil (T) and mid soil (M) related to Köppen climate classes. (**A**) forest floor (all sites or selected sites (time elapsed since harvesting <10 years)); (**B**) top soil; (**C**) mid soil (all sites). Means ± standard errors. Values are expressed as relative responses [log(whole-tree harvest/stem-only harvest)]. For the sake of clarity, comparisons between treatments and controls are also presented as the mean arithmetic difference (% higher or lower). Number of sites ranged from 7 to 23 (insufficient data for tropical climates). Significant differences between relative responses and value 0 are denoted by an asterisk (*t* test). There were also significant differences among classes (ANOVA, $P = 0.040$ for the mid soil, $P = 0.078$ for the forest floor with selected sites (0–10 years)).

former experience higher SOC losses than the latter. However, as hardwoods and conifers are not equally distributed along global climatic gradients, we tested a possible climatic bias. In practice, we repeated the comparison between these groups of tree species, but using a subset of our data for which mean annual temperature and precipitation were in the same range of values for all forests. Under this analytical restriction, the influence of vegetation composition was not significant, which suggested that the observed effect of forest composition might be related to climate. Similarly, no significant effect of forest age on SOC change could be detected in our datasets.

Simulation at the European scale—Conclusion. The aggregation of results collected from experimental forests indicated that intensive harvests have unwarranted consequences on soil carbon stocks and, consequently, could have an impact on carbon budgets. To quantify this possible effect, we extrapolated the development of intensive harvests in the European Union under different scenarios of intensive forestry development. Our simulations indicated a total loss of 5–17 Tg-C year^{-1}, depending on the scenario. We recognize that these estimates are broad extrapolations which require further investigation, by using process-based modelling for instance. On the other hand, they provided pertinent indications in comparison with other processes involved in the carbon cycle. Indeed, Luyssaert and his colleagues[32] calculated that the carbon sink of European forest soils was around 29 Tg-C year^{-1}. In terms of magnitude this value was comparable to our estimates of annual SOC losses from the same region. In other words, changing to more intensive harvests would have detrimental consequences, because soils would fix less carbon due to the loss of part of this sink, as shown by our results. Under our most severe scenario (i.e. 17 Tg-C year^{-1}), approximately 57% of the soil carbon sink was offset by unintended losses.

Our findings clearly demonstrate that using the *intensive harvest* strategy at its maximum level decreases soil carbon storage. Besides SOC losses, the removal of logging residues has other negative effects on forest soils, such as a decrease in nutrient availability (mainly due to increased exportation of nutrients) which could lead to a reduction in site fertility[2,33–35] and tree growth[34,35], thereby reducing carbon storage in tree biomass in the long term[12]. In sites where inherent soil fertility is low, intensive harvests should consequently be discouraged, to prevent productivity decline from occurring. Otherwise, the negative effects of intensive harvests should be mitigated by reducing the removal rate of logging residues[2,17,34,35] and preserving the forest floor[35].

Because the carbon budget also depends on carbon sequestration in standing trees[8] and on the substitution of fossil carbon by biomass[2], the question of whether additional harvesting of forest biomass has a positive impact on the greenhouse gas balance remains an open debate[12]. Conversely, our study provided accurate estimates of the losses of soil organic carbon that should be taken into account when assessing the potential benefits of forest bioenergy on the global carbon budget.

Methods

Meta-analysis compilation of data at the stand scale. Our global analysis was based on observations collected from 238 peer-reviewed publications. Gathering all these studies, we built two datasets. The first included values of organic C storage in soils under the influence of *conventional harvests* (i.e. treatment = tree stem harvest *versus* control = no harvest[18,19]; N = 118 and 80 sites for forest clear-cutting and forest thinning, respectively; N = 1462 values of soil organic carbon (SOC) changes, considering all soil layers, treatments and sampling dates for each site). As clear-cutting involves more severe disturbance than thinning, we systematically searched for possible differences between these two types of biomass export. Nevertheless, because there was generally no difference, clear-cutting and thinning were often merged in the results.

The second dataset encompassed the effects of *intensive harvests* (i.e. whole-tree harvest treatment = harvest of logging residues (e.g. branches, foliage, or stumps) in addition to stem harvest *versus* control = stem-only harvest[34]; N = 86 sites; N = 566 values of SOC changes, considering all soil layers, treatments and sampling dates). Most of data about intensive harvests were at clear-cutting stage.

Sites were distributed worldwide (Fig. 1 and Fig. S1), but most of them were located in the Northern hemisphere under temperate or cold climates (Tables S1 and S2). We collected SOC data, sampling depth and explanatory variables including geographical location, altitude, time since harvesting, thinning intensity, soil disturbance (i.e. ploughing after clear-cutting and before planting), vegetation, climate, and soil type. To assess the consequences of forest management practices on SOC storage as a function of soil depth and in the entire soil profile, SOC data were classified into four soil layers (see Supplementary Information for more details): forest floor (F: organic soil layer above the mineral soil profile), top mineral soil (T: mean sampling depth ≤ 10 cm), mid soil (M: 11–20 cm) and deep soil (D: >20 cm). SOC stocks (in Mg-C ha^{-1}) were subsequently calculated in each soil layer, in the mineral soil profile (e.g. TMD = T + M + D) and in the organic plus mineral soil profile (e.g. FT = F + T, FTMD = F + T + M + D).

We assessed the magnitude of changes in SOC stocks in response to conventional harvests and intensive harvests using the concept of *effect size* and a calculation of the relative response [log(treatment/control)] in each soil layer or in the soil profile. For the sake of clarity, comparisons between treatments and controls were also presented as the mean arithmetic difference or percentage change (higher or lower). To quantify the effect of intensive harvests as compared with unharvested controls, we combined the two datasets and used a bootstrap resampling method (see Supplementary Methods).

First, we evaluated the general effects of biomass harvest on SOC storage. To test the significance of the effect of each treatment (conventional or intensive harvests) on SOC stocks, the relative response was compared to 0 using a *t* test. Then, we explored the causes which explained the results and their heterogeneity. To do this, relationships between the relative response and explanatory variables (e.g. time elapsed since harvesting, initial SOC concentration, mean annual temperature) were assessed using either linear or non-linear regressions. Differences among classes of explanatory variables (e.g. elapsed time, soil types, climate classes) in the relative response were also assessed using one-way ANOVA.

Detailed information about the methods used in this paper is presented in the Supplementary Information.

Simulation at the European scale. In a final stage, we estimated the consequences of intensive harvests in Europe. We focused on Europe because 1) a carbon budget of European forests was available[32], 2) the great majority of those forests were managed using conventional harvesting (primary unmanaged forests correspond to only ~4% of total European forested area[36]), and 3) the relative importance of intensive forestry was likely to increase in upcoming decades as a result of the commitment of European countries to increase the proportion of renewable energy in their final energy consumption[2]. Because the rate of development of intensive forestry in Europe was unpredictable[2,17], we tested two different scenarios assuming that 20% or 70% of European forests currently managed using conventional harvesting would become intensively managed in the next three decades. The surface areas of European forests and their distribution in boreal or temperate regions were calculated from published data[36]. Total SOC stocks in managed European forests were then calculated based on their surface areas and mean SOC stock values per hectare. We assumed that mean SOC stocks in the whole organic plus mineral soil profile were 277 and 95 Mg-C ha^{-1} for boreal and temperate forests, respectively[3,15]. The impact of intensive harvests was estimated by applying the mean SOC loss value found in the present study (Fig. 6). We calculated a mean annual loss of soil carbon for our two scenarios, assuming a constant rate of loss over the 30 years, because biomass harvests could have consequences over decades[37] and because, in the present study, the impacts of intensive harvests have been assessed over a period of 30 years (Fig. S3).

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Author Contributions

G.L. and M.F. initiated the project. L.A., D.L.A. and M.F. designed the study. D.L.A. collected the data and D.L.A., L.A. and B.R. analyzed the results. D.L.A. and L.A. wrote the first draft, and all authors contributed to subsequent versions.

Additional Information

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Review

Quantifying consequences of removing harvesting residues on forest soils and tree growth – A meta-analysis

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A B S T R A C T

Increasing attention is being paid to using modern fuelwood as a substitute for fossil energies to reduce CO_2 emissions. In this context, forest biomass, particularly harvesting residues (branches), and stumps and associated coarse roots, can be used to supply fuelwood chains. However, collecting harvesting residues can affect soil properties and trees, and these effects are still not fully understood. The main objective of the present study was to compile published data worldwide and to quantify the overall effects of removing harvesting residues on nutrient outputs, chemical and biological soil fertility and tree growth, through a meta-analysis. Our study showed that, compared with conventional stem-only harvest, removing the stem plus the harvesting residues generally increases nutrient outputs thereby leading to reduced amounts of total and available nutrients in soils and soil acidification, particularly when foliage is harvested along with the branches. Losses of available nutrients in soils could also be explained by reduced microbial activity and mineralization fluxes, which in turn, may be affected by changes in organic matter quality and environmental conditions (soil compaction, temperature and moisture). Soil fertility losses were shown to have consequences for the subsequent forest ecosystem: tree growth was reduced by 3–7% in the short or medium term (up to 33 years after harvest) in the most intensive harvests (e.g. when branches are exported with foliage). Combining all the results showed that, overall, whole-tree harvesting has negative impacts on soil properties and trees that may have an impact on the functioning of forest ecosystems. Practical measures that could be taken to mitigate the environmental consequences of removing harvesting residues are discussed.

Contents

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1. Introduction

In Western countries, the use of traditional fuelwood was low - or decreasing- until the end of the 1970s (Fig. 1). Then, there was an interest in using modern fuelwood, mainly due to oil crises in 1973 and 1979 (fuelwood demand increased in parallel with oil price); locally other reasons contributed to this increased demand (e.g. decision to phase out nuclear energy in 1980 in Sweden). To supply fuelwood chains, foresters developed alternative cropping systems (such as short rotation coppices, Ranger and Nys, 1986) and adapted harvest practices (Nicholls et al., 2009; Diaz-Yanez et al., 2013). One of the adaptations proposed was to remove those tree components that were conventionally left in the forest: the so-called "harvesting residues" such as branches, foliages, tree tops, small diameter trees and technically damaged trees (e.g. Nunez-Regueira et al., 2005; Diaz-Yanez et al., 2013). In Europe, the new harvest practices included the integration of a second passage for removing harvesting residues (through better planning and logistics for extraction). In North America, harvesting systems in which residues are left at roadside ("full-tree-to-roadside" systems; Morris et al., 2014) have been developed in the late 1980s for economic and safety purposes. There was therefore no new harvesting system as fuelwood is a by-product of residue piles and not a primary objective.

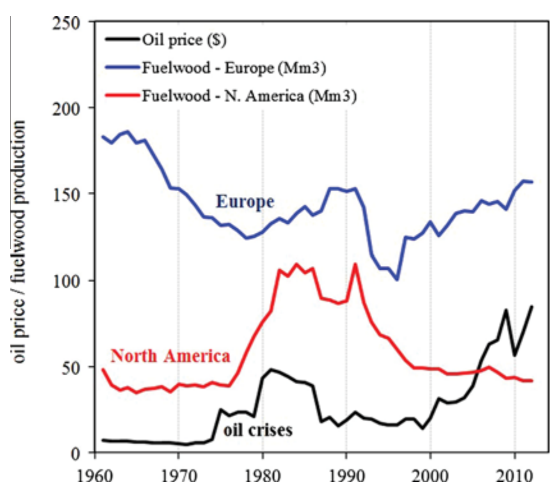

Fig. 1. Historical trends in oil price and fuelwood use in Europe and North America. Sources: oil price = World Bank Commodity Price Data (http://knoema.com); fuelwood = FAOSTAT (http://faostat.fao.org). Oil price in real 2005 US $ for crude oil. Fuelwood includes both traditional and modern fuelwoods. In Europe, the use of traditional fuelwood have decreased until the end of the 1970s. Then, there was a development of modern fuelwood and new interest in traditional fuelwood.

Early studies were carried out to assess possible environmental impacts of exporting harvesting residues (e.g. Tamm, 1969; Mann, 1984; Thompson et al., 1986; Mann et al., 1988; see also early studies in Scandinavia cited by Tveite and Hanssen (2013)). Experiment networks were also established, such as the North American long-term soil productivity study (LTSP) network (launched in 1989; Powers et al., 2005), the experiment network in Scandinavia (established in the 1970s and 1980s; Helmisaari et al., 2011; Tveite and Hanssen, 2013) or the Site Management and Productivity in Tropical Plantation Forests network (managed by the Center for International Forestry Research (CIFOR) since 1995; Nambiar et al., 2004; Nambiar, 2008). However, the demand for fuelwood decreased in the early 1990s following the collapse of the price of oil in the middle of the 1980s (Fig. 1). Interest in harvesting residues and related scientific research and funding consequently decreased. Since 2000, the emergence of developing economies (BRICS countries: Brazil, Russia, India, China and South Africa) triggered a long-term increase in the demand for energy causing a major trend toward an increase in the price of oil (Fig. 1). In the context of expensive oil and of climate change (IPCC, 2007), European countries introduced policies to promote the substitution of fossil fuel by renewable energies like fuelwood (European Commission, 2000) to enable national energy security (reduced oil dependence) and to decrease the emission of greenhouse gases (Stupak et al., 2007). One consequence of these policies was to revive interest in forest harvesting residues as a possible source of energy (Nicholls et al., 2009). Displacing fossil fuels is also the result of international competition for forest products, which led to diversification into new markets such as energy. It also should be noted that whole-tree harvesting in North America was mainly driven by the evolution of equipment for economic and safety purposes as explained before.

Already in the 1980–1990s in North America and even earlier in Scandinavia, some authors reported that collecting harvesting residues may negatively impact forest ecosystems (Tamm, 1969; Mann, 1984; Thompson et al., 1986; Mann et al., 1988; Johnson et al., 1991) because this kind of biomass (branches, foliage and tops) contains large amount of nutrients (Fahey et al., 1991; Yanai, 1991, 1998; Son and Gower, 1992) that are useful for the sustainability of ecosystem functioning and functions (Ranger and Turpault, 1999). Recently, the possible impacts of exporting harvesting residues were reviewed (Lattimore et al., 2009; Thiffault et al., 2011; Wall, 2012). Reviews and meta-analyses have also been carried out for LTSP installations in North America

(Powers et al., 2005; Fleming et al., 2006; Ponder et al., 2012). These useful studies confirmed that these practices can have negative effects on forest soils and tree growth. In the present study, we aimed to go one step further as our first objective was to quantify the overall impacts of removing harvesting residues by comparing whole tree harvesting with conventional stem-only harvesting using data published world-wide. We assessed the impacts on a large number of soil properties and tree growth variables. To this end, we compiled published data and analyzed two datasets using a meta-analysis approach. A first dataset on nutrient stocks in the different tree components and soil profiles was used to quantify the increases in nutrient outputs (exportations with harvested biomass) due to removing harvesting residues, and to compare nutrient outputs with nutrient stocks in soils (hereafter referred to as "nutrient stocks" dataset). Another dataset (referred to as "environmental impacts" dataset) was used to identify and quantify the impacts on soil fertility (e.g. soil nutrient stocks, organic matter quality, biological activity) and the growth of subsequent forest stands. Our second objective was to evaluate the effect of the intensity of residue harvest (e.g. harvest of branches vs. branches + foliage) and the potential causes of heterogeneity in the response of the soils and of the trees (e.g. soil type, inherent soil fertility, time elapsed since harvesting) with the aim of identifying practical measures that could be used to mitigate the environmental consequences of removing harvesting residues.

2. Materials and methods

2.1. Data acquisition

We used the ISI Web of Science database and holistic non-specific queries (Pullin and Stewart, 2006; Augusto et al., 2013). Then we used inclusion and exclusion criteria to select publications that reported relevant data. First, we selected publications with data on trees and related soil nutrient stocks so that we could determine nutrient removals with different intensities of biomass harvesting and compare this with soil nutrient capital. We therefore identified publications using keywords related to the amounts of nutrient in the tree components (e.g. "nutrient" or "nitrogen" or "phosphorus"; "content" or "concentration" or "stock" or "amount"; "forest" or "woodland" or "tree") and compiled a "nutrient stocks" dataset. To be included in the dataset, the studies had to satisfy the following criteria: (1) nutrient amounts in the tree components had to be quantified by destructive sampling (estimations based on published allometric relationships were excluded); (2) nutrient stocks (in kg ha^{-1}) in the tree components had to be included in the studies or calculated using nutrient concentrations (e.g. in mg g^{-1}) and biomass values (in Mg ha^{-1}; i.e. studies that reported only nutrient concentrations could not be used), (3) stem data had to be given separately from the other tree components to enable comparison among harvests (see types of harvest treatments in Fig. 2 and details in Section 2.2.1; studies that only reported nutrient stocks in total tree biomass could not be used), (4) soil nutrient data were included only when they corresponded to stocks. This selection stage led to a list of 230 primary articles representing 749 case studies (a case study was defined as the unique combination of one site and one tree species; see references list in Supplementary Information).

Secondly, we identified publications with data on the impacts of different intensities of biomass harvesting by using keywords related to the collection of harvesting residues (e.g. "whole-tree" or "slash" or "residues" or "debris"; "harvesting" or "management") and compiled an "environmental impacts" dataset. Because we wanted to assess the impacts of removing harvesting residues on a large number of physical and chemical soil properties

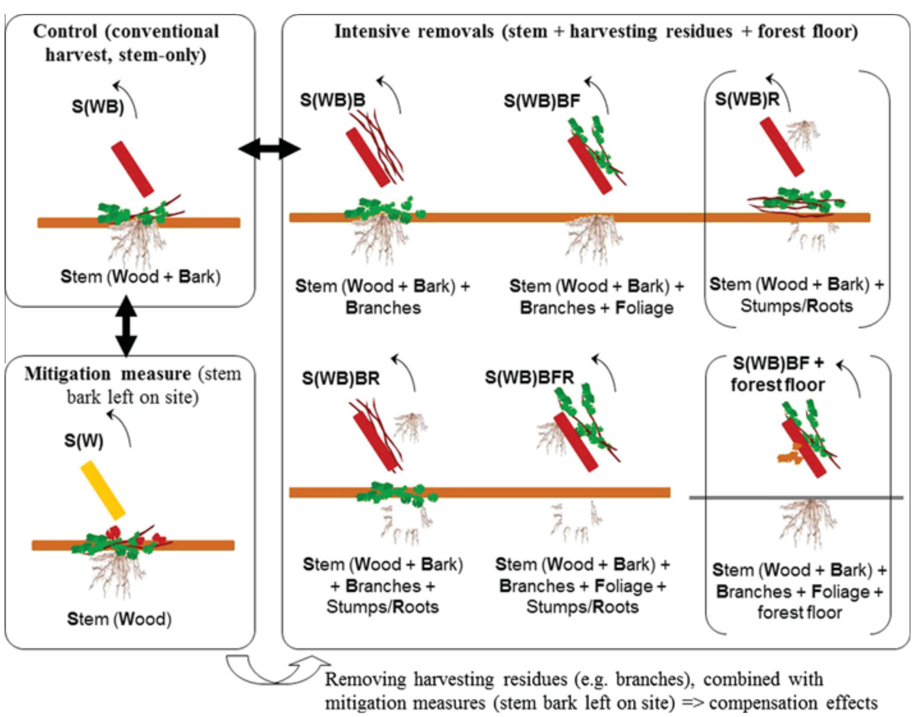

Fig. 2. Main harvest treatments considered in this study. Conventional stem-only harvest (S(WB), control) compared to different types of intensive removals or to stem wood harvest (S(W), stem bark left on site; mitigation measure). Treatments in brackets (removing stumps and associated coarse roots, with branches left on site: treatment S(WB)R; removing harvesting residues and forest floor: S(WB)BF + forest floor) are only included in the "environmental impacts" dataset. The "environmental impacts" dataset also includes case studies in which intensive removals are compared to *double slash* treatment (i.e. stem-only harvest with harvesting residues left on site and inputs of residues from an intensive removal treatment). The removal of forest floor and the *double slash* treatment were included in some experiments mainly to create large variations in treatment impacts and for theoretical reasons.

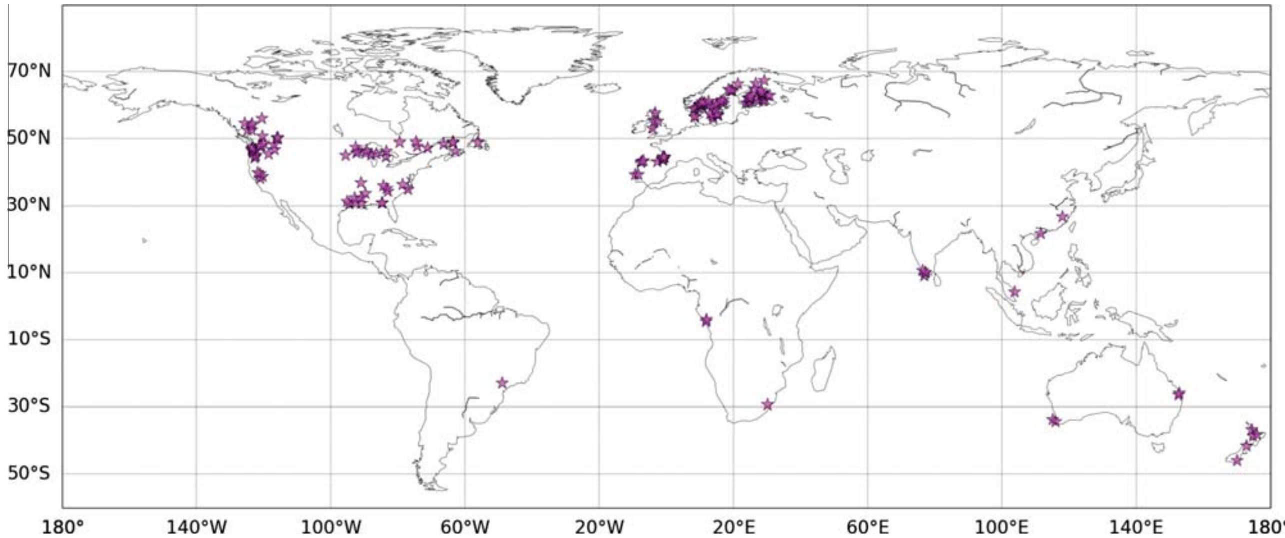

Fig. 3. Spatial distribution of the study sites ("environmental impacts" dataset). The dataset includes 168 experimental sites distributed as follow: 43% in North America (29% in USA, 14% in Canada, mainly from the "North American long-term soil productivity" study (LTSP network)), 1% in South America, 45% in Europe (35% from experiment network in Scandinavia), 4% in Asia, 2% in Africa and 5% in Oceania. Several sites in the tropics are from the "Site Management and Productivity in Tropical Plantation Forests" network (CIFOR project). Sites were mostly under temperate climate (40%) and cold climate (51%) based on the Koeppen climate classes.

(e.g. carbon (or organic matter) and nutrient concentrations (or stocks) in soils, soil pH), environmental conditions (e.g. soil temperature and moisture), biological soil properties (e.g. fauna, microbiological and enzymatic activities, decomposition processes) and tree variables (nutrient status, survival and growth), we focused our selection criteria on harvest treatments rather than on data themselves. Removing harvesting residues can also have consequences on water quality and biodiversity (e.g. Lattimore et al., 2009). These effects were however not assessed in the present study.

We selected studies that compared the conventional stem-only harvest with the removing of the stem plus harvesting residues (i.e. branches, foliage), stumps and associated coarse roots and sometimes the forest floor (Powers et al., 2005; Mariani et al., 2006; Thiffault et al., 2011; Wall, 2012). Although the forest floor can be used as fuelwood (e.g. in South Europe; Nunez-Regueira et al., 2005), its removal was generally included in experiments to create large variation in treatment impacts for theoretical reasons (e.g. in North America, LTSP network). It should also be mentioned that, contrarily to other treatments, stump removal includes soil disturbance that could also affect soil properties and tree growth (Egnell et al., 2015). To be included into the database, treatments had to be compared in experimental designs or in adjacent stands (paired sites with similar soil conditions and vegetation). This selection led to a list of 140 articles and a total of 168 experimental forest sites for the "environmental impacts" dataset. Because these studies were not all based on the same response variables, the number of case studies depended on the soil or tree variable studied and ranged from three to 57 (see lists of references in Supplementary Information).

While collecting the data from the publications, we used the DataThief III (version 1.5) software to extract the values from figures (when these were not given in tables).

The studies used for the compilation of both datasets were conducted worldwide (see Fig. 3 for the "environmental impacts" dataset and Fig. S1 in Supplementary Information for the "nutrient stocks" dataset). But, most of the sites were located in the northern hemisphere (USA, Canada and Europe) under temperate or cold climates. The "environmental impacts" dataset included soil and tree response data up to 33 years after harvesting (see as an example the complete dataset for the effects of removing harvesting residues on tree growth; Fig. S2 in Supplementary Information).

2.2. Data handling and statistics

2.2.1. Estimation of nutrient outputs ("nutrient stocks" dataset)

Nutrient outputs were estimated for the conventional harvest treatment (stem-only harvest, including wood and bark (S(WB)); i.e. control treatment) and each of the intensive harvest treatments (stem (wood + bark) + different harvesting residues; Fig. 2). The harvesting residues we considered in this study included branches and stumps (with attached coarse roots), because both can be used to produce fuelwood (e.g. Diaz-Yanez et al., 2013). We assessed the effect of exporting branches with or without foliage (nutrient rich component; Santa Regina, 2000; Ponette et al., 2001; Augusto et al., 2008a), depending on harvest conditions (e.g. with or without a delay of 1–3 months between delimbing and harvesting the branches, or more efficiently a delay between tree cutting and delimbing, which would allow the foliage to dry out and fall off the branches; Nord-Larsen, 2002; Stupak et al., 2008). Finally, the intensive harvest treatments we studied correspond to different combinations of harvested residues (Fig. 2). Some harvest treatments were not included in the "nutrient stocks" dataset. This was the case of the treatments S(WB)R (the stumps and coarse roots were harvested, while the branches were left on site) and S(WB)BF + forest floor (removal of harvesting residues and of the forest floor) which were assessed in the "environmental impacts" dataset only.

We also estimated nutrient outputs when only stem wood was harvested (stems debarked and the bark left on site); we tested this harvest treatment as a possible measure to reduce nutrient outputs because bark is known to concentrate large amounts of nutrients, particularly Ca (André et al., 2010; André and Ponette, 2003). This treatment corresponds to a real harvesting method, which is used, for instance, in Congolese commercial plantations (Laclau et al., 2010).

Nutrient outputs were estimated in two steps. In the first step, potential outputs were estimated assuming that tree components were totally removed; these estimates are defined here as *theoretical values calculated using 100% harvest rates*. However, our preliminary analysis revealed differences between these potential values and observed nutrient output data (potential values could be significantly higher than the observed values; based on data from Johnson et al. (1982), Johnson and Todd (1987), Tritton et al. (1987), Fraysse and Cotten (2008)). Indeed, several studies have shown that in practice, not all harvesting residues can be collected (see harvest rates in Eriksson (1993), Bergquist et al. (1999), Egnell and Leijon (1999), Nurmi and Hillebrand (2001), Cacot et al. (2007), Fraysse and Cotten (2008), Wall (2008), Wall and Hytönen (2011), Augusto et al. (2015); see also the recent review written by Thiffault et al. (2014)). Therefore, in the second step, we used harvest rate values that simulated incomplete harvests. Because harvest rates differed among studies (e.g. Thiffault et al., 2014), we used mean values based on all references cited above: 100% of stem wood, 20–80% of stem bark depending on the harvest method used (80% when a chainsaw was used; down to 20% as potential value when logging machines were used, as these could cause large quantities of bark to detach from the stems), 60% of stumps and associated roots, 50% of branches of coniferous tree species and 60% of branches of broadleaf tree species, and 0–40% of foliage depending on the harvest conditions (down to 0% of leaves and 10% of needles as potential values, after a delay between cutting the stem and harvesting the branches, which allowed the foliage to dry and fall off the branches; 0% of leaves when branches were harvested in fall or winter; 40% when these conditions were not met). Theoretical changes using 100% harvest rates were estimated for several macronutrients (N, P, K, Ca, Mg and S) and micronutrients (Na, Fe, Mn, Zn, Cu, Ni), while theoretical changes based on realistic rates were estimated only for macronutrients.

Finally, for each harvest treatment, theoretical N, P, K, Ca and Mg outputs were compared to the nutrient stocks in soils (stocks of total N and P, available P and exchangeable K, Ca and Mg) to assess potential impacts on chemical soil fertility (Tamminen et al., 2012). It should be noted that available/exchangeable soil nutrient data is more relevant than total soil nutrient data to assess potential impacts; there is however no data on available N in the dataset. The mean thickness of the soil profiles we analyzed was 84 ± 21 cm.

2.2.2. Data classification

For both datasets, data were primarily classified as a function of the type of harvest treatment, as shown in Fig. 2. In addition to the general effects of the harvest of each type of residues, we also assessed possible causes of heterogeneity. To this end, we collected variables as possible predictors and classified our data accordingly.

For the "nutrient stocks" dataset, data were classified based on vegetation (mainly the two classes studied: coniferous and broadleaf trees). We also assessed the effect of removing harvesting residues on nutrient outputs in relation with stand characteristics (tree age, stem biomass, tree height and diameter (DBH)).

For the "environmental impacts" dataset, because impacts on soil properties generally depend on soil depth, we split our data into four soil layers: wood debris ("WD"), forest floor ("FF") and two mineral soil layers. The mineral soil layers were classified with respect to the sampling depth: top soil ("Top" refers to soil depth <20 cm) and deep soil ("Deep" refers to a soil depth >20 cm). Data were also classified according to the methods of soil analysis used (e.g. quantification of total or only plant-available nutrients in soils, concentration or stocks of nutrients). Variations also exist among methods used to assess exchangeable/available nutrients and data are generally not directly comparable among studies. However, the metric used in the present study (response ratio) enabled avoiding any effect of methodological differences (see next section). Data were also classified based on possible predictors: time elapsed after harvesting (two classes: 0–10 years and >10 years), tree species in the subsequent forest stands (coniferous vs. broadleaf tree species), location (there were enough data only to compare Europe and North America), Koeppen climate classes (determined based on geographical coordinates; http://koeppen-geiger.vu-wien.ac.at/present.htm) and soil types (based on the FAO classification). When not specified in the publications, we determined the FAO soil type based on soil description and properties, or on correspondences among soil classification systems (e.g. Esu, 2010).

2.2.3. Calculation of the magnitude of change

To make it possible to compare publications across a wide range of experimental conditions, we calculated the magnitude of change (i.e. percent change (higher or lower); Elser et al., 2007; Nave et al., 2010; Wei et al., 2014) in nutrient outputs ("nutrient stocks" dataset) and soil and tree variables ("environmental impacts" dataset) in response to the removal of harvesting residues:

$$\text{e.g. Percent change} = \left(\frac{S(WB)BF - S(WB)}{S(WB)}\right) \times 100 \tag{1}$$

Although comparisons between treatments are presented as the arithmetic difference (see Eq. (1)), statistics were carried out using the relative response metric (Nave et al., 2010; Augusto et al., 2013), as it is generally the case in meta-analyses:

$$\text{e.g. Relative response} = \log\left(\frac{S(WB)BF}{S(WB)}\right) \tag{2}$$

Values of the relative response close to 0 are associated with a negligible effect of the intensive harvest treatments tested. Negative and positive values indicate negative and positive effects, respectively. To test the significance of the effect of each intensive harvest treatment, the relative response was compared to 0 using one sample *t*-test. Comparisons among classes of explanatory variables were also assessed using a generalized linear model and the Bonferroni *t*-test. Statistics were performed using SYSTAT (version 10, Software Inc., Chicago, IL, USA) software.

In meta-analyses, the relative response metric can be weighted by the precision of the study (i.e. using variances and sampling sizes; Gurevitch and Hedges, 1999). Because variance estimates were not available in many studies, we used an unweighted metric. According to Gurevitch and Hedges (1999), an unweighted metric can be used in meta-analysis without severely hampering test validity. Contrary to many previous meta-analyses, we avoided pseudo-replicates. Indeed, the number of values in a given publication depends on the quantity of repeated measurements (e.g. several sampling dates). Because the number of values varied greatly among the publications, assuming that each value was an independent case study would lead to different statistical weights and consequently would bias the meta-analysis (Gurevitch and Hedges, 1999; Ioannidis, 2010). When several relative responses corresponded to one case study, to avoid pseudo-replications, we calculated a single mean value. When assessing the effect of elapsed time after harvesting, sequential data of a given case study were split into different classes (0–10 and >10 year) and a mean value per class was calculated. Here, we define a case study as being based on one geographical location, the type of removal of harvesting residues, and the tree species in the forest concerned.

Another difficulty encountered in meta-analyses is publication bias (Gurevitch and Hedges, 1999; Ioannidis, 2010). Publication bias occurs when the probability of publication depends on the statistical significance, magnitude or direction of the effect and causes a bimodal distribution of the number of studies (e.g. low frequency associated with low values of relative response and high frequencies with high values of relative response). Here, we generally found unimodal distributions of the case studies in relation to the values of relative response (see examples in Fig. S3 in

Supplementary data). We consequently concluded that there was no publication bias.

3. Results

3.1. Effects of removing harvesting residues on nutrient outputs ("nutrient stocks" dataset)

3.1.1. Theoretical changes in nutrient outputs

Compared with conventional S(WB) harvest with chainsaw logging (with 80% of bark removed), S(WB) harvest with machine logging (with only 20% of bark removed) enabled reductions in nutrient outputs of up to −38% for Ca (Table 1; theoretical reductions calculating using 100% harvest rates were up to −56%; see also Tables S1 and S2).

Changes in nutrient outputs caused by removing the stem plus harvesting residues and calculated using realistic harvest rates reached +128% and were significantly lower than the theoretical values calculated using 100% harvest rates (up to 4 times lower; see comparisons in Table 1 and S2). In general, the magnitude of change increases with an increase in the number of tree components harvested (% change in S(WB)B < % change in S(WB)BF ⩽ % change in S(WB)BR < % change in S(WB)BFR). Collecting branches (e.g. treatment S(WB)B in Table 1) led to increases in nutrient outputs of +26% to 31%. Adding the removal of foliage or stumps/roots (S(WB)BF or S(WB)BR in Table 1) resulted in bigger changes in nutrient exports (+40% to 68% and +48% to 63%, respectively), but mitigation measures such as harvesting in winter or after a delay which allows the foliage to dry and fall off the branches strongly reduce these effects (increase in nutrient outputs of +28% to 38% under S(WB)BF). Adding other mitigation measures to the S(WB)BF harvest treatment to reduce the export of bark (i.e. using machines for logging), led to an increase in nutrient outputs of only +8% to 13% (no change in the case of Ca, Table 1).

Theoretical changes in macronutrient outputs due to the removal of harvesting residues are generally higher than the gain in biomass (Table S1). In particular, exporting the foliage induces a small gain in biomass harvest but huge nutrient exports/losses, because foliage mass is generally low and its nutrient concentrations are high (Fig. S4). The changes in theoretical nutrient outputs displayed high inter-site heterogeneity (e.g. Fig. S4), which appears to be correlated with the stage of development of the forest stand. Indeed, the magnitude of changes in nutrient outputs due to the collection of branches and foliage increases with decreasing tree diameter (Fig. S5A), and also tree height, tree age and stem biomass (Fig. S6). This result can be explained by the fact that the contribution to total tree biomass of foliage and thin branches, i.e. tree components with high nutrient concentrations, is larger in young stands than in old stands (Fig. S5B). The variability of nutrient outputs can also be explained by an effect of tree species. Indeed, changes were greater in coniferous tree species than in broadleaf trees as shown in Fig. S5 and other results (significant differences ($P < 0.05$) were generally found between the two classes in each harvest treatment (data not shown)).

3.1.2. Comparison between theoretical nutrient outputs and nutrient stocks in soils

Theoretical N outputs were low compared to total N stocks in the soil profiles under all types of harvest (N outputs <10% of total soil N). This was also the case when P outputs were compared with total P in soils (P outputs <2% of total soil P; data not shown). However, theoretical nutrient outputs were high compared with the stocks of available/exchangeable nutrients in soils, particularly when harvesting residues were removed. Indeed, P, K, Ca and Mg outputs generally corresponded to 20–30% of available/exchangeable soil nutrients in conventional (S(WB)) harvest, and up to 100% or more in intensive harvests (Fig. 4). Thus, in addition to results concerning percent changes in nutrient outputs, comparisons with nutrient stocks in soils also strongly suggest that intensive harvests can negatively affect chemical soil fertility.

Comparing theoretical nutrient outputs to nutrient stocks in soils also showed that harvesting wood stems without the bark (S(W) treatment) can mitigate the impacts of biomass harvest on chemical soil fertility (particularly on soil Ca, Fig. 4). This result is coherent with those on nutrient outputs (potential reduction of 56% in Ca outputs; Table 1).

3.2. Impacts of removing harvesting residues on soils and trees ("environmental impacts" dataset)

3.2.1. Soil organic matter and nutrients

In general, removing harvesting residues led to significant losses of soil organic matter (or C), particularly in the wood debris (−40%, as a median value), forest floor (−10% to −45%) and deep soil layer (−10%). Significant decreases were found for soil organic matter stocks and concentrations; Table 2). When we focused on stocks of organic matter in the forest floor, results suggest that losses increase with increasing harvest intensity (losses in S(WB)B < S(WB)BF < S(WB)BFR < S(WB)BF + forest floor).

Removing harvesting residues significantly decreased the amount of nutrients in soils (Table 2). Results for total N were similar to those for soil organic matter, with increased N losses with increasing harvest intensity (e.g. see comparisons of the effect of different harvest treatments on N stocks in the forest floor). There were also significant losses of other nutrients such as total P or Ca (−6% to −9%; no effect for total K and Mg). The consequences of removing harvesting residues for available soil N (KCl extractable NH_4 and NO_3) were generally not significant, except when the forest floor is harvested (−24% in topsoil). However, our results showed overall negative and significant effects on available soil P, cation exchange capacity and base saturation (examples for S(WB)BF treatment: changes of −8% to −12% in the forest floor, and of −10% to −17% in top soils). Concomitant with the decreases in exchangeable cations and base saturation, soil acidification can also be inferred from slight decreases in soil pH and increases in exchange acidity and exchangeable H and Al (data mainly from Northern boreal forests).

Although overall there were significant negative effects, soil responses to removing harvesting residues displayed strong heterogeneity that may be related to several explanatory variables. Classifying data based on Koeppen climate classes showed that the decrease in organic matter and total N was higher under temperate climate than cold climate (Fig. S7; there were not enough data to compare climate classes for other soil variables). Classifying data based on the elapsed time after harvesting (0–10 yrs. and >10 yrs.) showed that the decreases in total N (stock in the forest floor), and exchangeable K and Mg tended to be stronger during the first years after harvesting than later (Figs. S8 and S9). However, there were generally no significant differences among the two classes. In contrast, there was a significant effect of elapsed time after harvesting on topsoil pH, with reduced values only during the first years (Fig. S10). We did not find any significant relationship between percent changes in soil organic matter or nutrients and concentrations in control treatment (i.e. indicators of inherent soil fertility). Finally, we assessed data distribution based on geographical locations and soil types. In several cases, data distribution was unbalanced: USA and/or Sweden/Finland were generally the most frequently represented countries, and podzol the most represented soil type (acrisols, gleysols and andosols were also present). Where comparisons were possible, results showed no effects of location or of soil types.

Table 1
Percent changes in nutrient outputs due to removing harvesting residues. Theoretical values calculated using harvest rates[a] ("nutrient stocks" dataset).

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB); logging with chainsaw[b])							
	S(WB) Stem (wood + bark)	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage			S(WB)BR Stem (wood + bark) + branches + stumps/roots	S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	
	Machine[b]	Chainsaw[b]	Chainsaw	Chainsaw, in winter or after drying step[c]	Machine, in winter or after drying step	Chainsaw	Chainsaw	Chainsaw, in winter or after drying step
N								
(Number of case studies)	(117)	(112)	(109)	(109)	(109)	(35)	(35)	(35)
Mean, Median (Q1, Q3)	−24, −25 (−30, −18)	37, 30 (18, 50)	87, 68 (49, 102)	46, 37 (24, 57)	21, 13 (1, 30)	79, 59 (43, 100)	113, 100 (76, 112)	85, 65 (50, 103)
Min, Max	−62, −2	5, 216	10, 425	5, 259	−54, 222	19, 400	43, 422	26, 400
Potential Mean, Median[d]	−37, −37	77, 56	177, 150	177, 150	177, 150	140, 126	222, 192	222, 192
P								
(Number of case studies)	(109)	(105)	(102)	(102)	(102)	(34)	(34)	(34)
Mean, Median (Q1, Q3)	−24, −23 (−33, −15)	43, 31 (18, 47)	84, 62 (41, 100)	50, 38 (22, 54)	25, 11 (2, 29)	81, 52 (44, 101)	109, 96 (57, 128)	87, 59 (47, 101)
Min, Max	−56, −2	5, 541	11, 704	5, 541	−20, 525	15, 446	23, 464	15, 446
Potential Mean, Median	−36, −36	90, 62	183, 152	183, 152	183, 152	172, 122	240, 200	240, 200
K								
(Number of case studies)	(110)	(107)	(104)	(104)	(104)	(36)	(36)	(36)
Mean, Median (Q1, Q3)	−22, −21 (−28, −15)	32, 27 (18, 40)	62, 55 (34, 83)	37, 31 (19, 47)	14, 11 (0, 23)	76, 63 (45, 99)	103, 85 (69, 123)	82, 67 (52, 102)
Min, Max	−51, −2	3, 127	9, 166	5, 137	−20, 106	16, 276	28, 294	16, 276
Potential Mean, Median	−33, −32	69, 48	146, 119	146, 119	146, 119	125, 105	185, 159	185, 159
Ca								
(Number of case studies)	(102)	(101)	(98)	(98)	(98)	(36)	(36)	(36)
Mean, Median (Q1, Q3)	−37, −38 (−47, −25)	32, 26 (16, 44)	54, 40 (26, 65)	37, 28 (19, 51)	0, −3 (−18, 10)	61, 48 (38, 65)	76, 58 (41, 102)	64, 49 (38, 74)
Min, Max	−69, −10	4, 192	6, 215	5, 192	−52, 154	14, 288	38, 296	23, 288
Potential Mean, Median	−56, −56	64, 47	122, 83	122, 83	122, 83	113, 93	147, 120	147, 120
Mg								
(Number of case studies)	(101)	(100)	(97)	(97)	(97)	(36)	(36)	(36)
Mean, Median (Q1, Q3)	−25, −25 (−33, −16)	40, 27 (19, 47)	66, 53 (32, 84)	45, 34 (21, 56)	20, 8 (0, 29)	76, 59 (39, 98)	96, 80 (59, 124)	80, 59 (43, 105)
Min, Max	−57, −3	4, 273	9, 331	5, 273	−30, 261	15, 185	20, 187	15, 185
Potential Mean, Median	−38, −39	74, 51	135, 99	135, 99	135, 99	129, 109	171, 143	171, 143

[a] Harvest rates: 100% of stem wood, 20–80% of stem bark depending on harvest conditions (see below), 50–60% of branches depending on tree species, 0–40% of foliage depending on harvest conditions (see below), 60% of roots.
[b] Logging with a chainsaw causes low bark losses, 80% of the bark remains on the stem and is exported, while machine logging significantly increases bark losses, only 20% of the bark remains on the stem and is exported.
[c] Foliage exports are strongly reduced with mitigation measures (100% of foliage is left on site when broadleaf trees are removed in winter or when residues are removed after a delay that allows the foliage to dry and fall off the branches, 90% of foliage is left on site when residues of coniferous trees are removed after a delay that allows the foliage to dry and fall off the branches).
[d] Comparison with the theoretical values calculated using harvest rates of 100%.

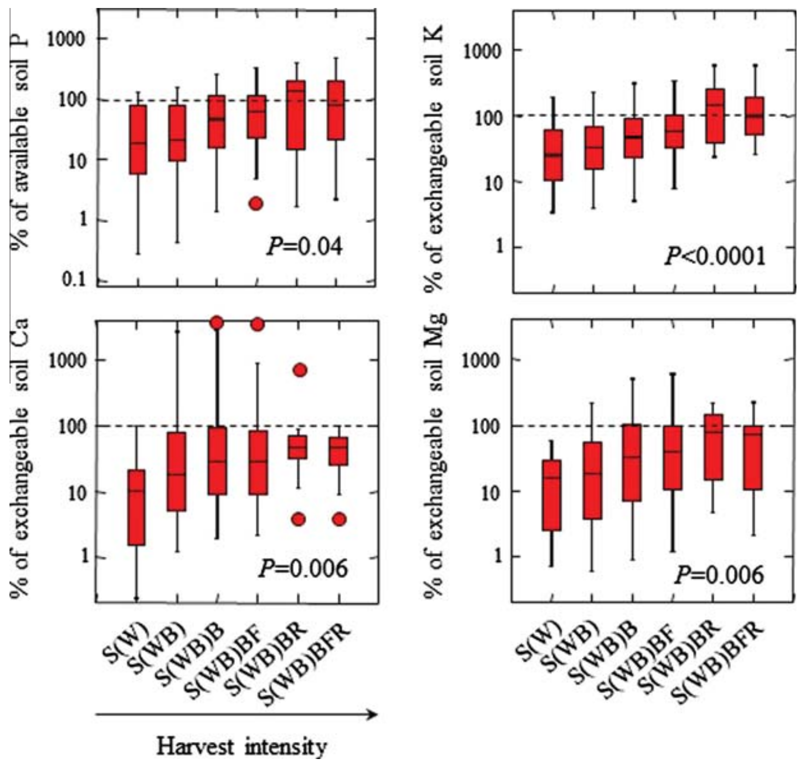

Fig. 4. Nutrient outputs at a theoretical harvesting rate of 100% as a percentage of soil nutrient stocks under different types of biomass removals ("nutrient stocks" dataset). The mean soil profile thickness is 80 cm. Number of case studies = 11–42. *P* values represent the general effects of the harvest.

3.2.2. Factors affecting soil biological activity

Effects on organic matter quality and environmental factors that may affect biological activity (compaction, soil temperature and soil moisture) are presented in Table 3.

Results suggested that removing harvesting residues may have an impact on the quality of soil organic matter, as shown by increased C:N ratios in the forest floor (+2% when all treatments are taken together; Table 3), as well as some significant changes in chemical composition of the soil organic matter. Indeed, there were reductions in the concentrations of diterpenes under S(WB)BF harvests (−15%; P = 0.063; *n* = 7), as well as reductions in the light fraction organic matter, lignin-derived phenols, cutin-derived compounds and alkyl C concentration when all treatments are taken together (−9% to −30%; *P* = 0.023–0.085; *n* = 3–7). However, there was generally no effect on other compounds, such as sesquiterpenes, triterpenes and phenolic compounds (including tannins; data not shown).

Soil compaction increased in the case of intensive harvests, as revealed by significant increases in topsoil bulk density (+4% increase under the S(WB)BF harvest treatment; Table 3).

Removing harvesting residues led to significant increases in topsoil temperature in spring and summer, while soil temperature was not affected in fall, and tended to decrease in winter (+5% to 10% increase in mean soil temperature; Table 3 and Fig. S11). In contrast, we found no significant change in topsoil moisture (Table 3) and no clear difference were observed among seasons.

3.2.3. Biological activity and decomposition processes

We found no significant effect of removing harvesting residues on soil fauna inferred from the number of individuals (mites, springtails, millipedes, nematodes, annelids, beetles, etc.) or species richness (Table 3). Microbiological activity in mineral soil layers, inferred with microbial biomass, soil respiration in incubated soils or other indicators, significantly decreased under S(WB)BF and S(WB)BF + forest floor treatments (−8% to −28% changes; no significant effect in the forest floor). Microbial C:N was however not significantly changed. An impact of removing harvesting residues on soil microbiological activity can also be suggested through reduced soil CO_2 efflux (10% decrease). In addition, there were significant decreases in enzymatic activities (particularly those involved in N mineralization (37–50% decreases), but also those involved in C decomposition and phosphate hydrolyze; Table 3).

Decomposition processes inferred from wood decomposition (*mass losses*) were not significantly affected, but net N mineralization fluxes in the forest floor and topsoil were significantly reduced. Data on other N processes (nitrification, microbial N immobilization) were also available. Their responses to the removal of harvesting residues remained unclear, although results showed an increase in nitrification when harvesting residues and forest floor are removed (Table 3).

Data on microbiological activity and decomposition processes are generally scarce and the response to removing harvesting residues was highly variable. In addition, data distribution based on explanatory factors was unbalanced (e.g. soils were mainly podzols or acrisols). Consequently, we were unable to assess the causes of heterogeneity of the effects of residue harvesting on biological soil fertility.

3.2.4. Tree growth

Our results showed that all residue harvest treatments generally had no effect on tree nutrient status, except for foliar Ca concentration, which was significantly and negatively reduced under the S(WB)BF harvest treatment (−4%; Table 4). Foliar K concentration was also reduced in some cases, i.e. when we compared residue harvest with double slash treatments (per cent change = −8%). Removing harvesting residues had generally no effect on tree survival. In contrast, our results clearly showed that tree growth was

Table 2
Percent changes in physical–chemical soil properties due to removing harvesting residues ("environmental impacts" dataset).

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB))					Compared to *double slash*
	All treatments	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage	S(WB)R or S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	S(WB)BF + forest floor Stem (wood + bark) + branches + foliage + forest floor	S(WB)BF Stem (wood + bark) + branches + foliage
Soil organic matter (or C) stock						
WD[a]	**−40.6 [−57.5, −33.2](20)*****	–[b]	**−40.0 [−47.8, −31.2](19)*****	–	–	–
FF	**−15.4 [−31.2, −2.9](39)*****	+1.7 [−6.3, 8.2](5)^{ns}	**−10.3 [−24.9, −2.3](36)****	**−24.4 [−30.0, −21.1](4)***	**−44.9 [−68.2, −34.3](8)***	–
Top	−2.1 [−6.8, 3.1](34)^{ns}	−1.1 [−3.1, 6.0](5)^{ns}	−1.1 [−5.2, 4.2](29)^{ns}	+2.8 [−0.2, 5](4)^{ns}	**−11.1 [−22.7, −7.5](10)***	**−5.5 [−6.1, −3.3](5)**$(^{*})$
Deep	**−8.0 [−14.3, 0.7](10)***	–	**−9.6 [−14.5, 1.9](9)***	–	−2.2 [−5.6, −1.5](5)^{ns}	–
Soil organic matter (or C) concentration						
FF	**−1.8 [−6.5, −0.1](27)***	–	**−3.0 [−8.2, −0.1](27)****	–	+6.6 [−7.0, 12.4](5)^{ns}	−3.7 [−3.8, −0.6](5)^{ns}
Top	**−7.3 [−19.3, 1.5](24)**$(^{*})$	–	−6.9 [−12.7, 2.4](16)^{ns}	–	**−23.5 [−38.3, −14.3](8)***	−6.8 [−25.8, −2.6](3)^{ns}
Deep	+3.0 [−2.1, 6.9](11)^{ns}	–	+2.1 [−0.9, 5.4](9)^{ns}	–	–	–
Total N stock						
FF	**−11.3 [−31.1, −1.3](24)****	+5.4 [−7.3, 17.2](6)^{ns}	**−12.1 [−31.5, −3.4](23)****	–	−51.2 [−52.4, −32.6](5)^{ns}	–
Top	−0.8 [−6.6, 1.8](27)^{ns}	−2.0 [−2.8, 6.0](5)^{ns}	−0.3 [−8.7, 2.1](25)^{ns}	–	−6.5 [−11.9, 1.7](9)^{ns}	**−8.6 [−10.2, −6.9](4)****
Deep	−5.6 [−9.1, −2.7](7)^{ns}	–	−9.0 [−11.9, −5.4](6)^{ns}	–	0.0 [−7.4, 3.5](5)^{ns}	–
Total N concentration						
FF	−2.4 [−8.1, 0.6](21)^{ns}	–	−2.4 [−7.1, 1.7](21)^{ns}	–	+9.9 [−16.3, 10.2](5)^{ns}	+0.5 [−2.2, 2.3](4)^{ns}
Top	**−10 [−18.8, −4.0](24)****	–	−7.4 [−9.7, 6.0](14)^{ns}	**−16.2 [−19.7, −15.4](5)****	**−18.8 [−27.4, −10.1](7)***	–
Deep	−1.5 [−10.7, 1.5](8)^{ns}	–	−3.0 [−12.8, 0.0](7)^{ns}	–	–	–
Total P						
FF	−4.3 [−6.8, −1.7](7)^{ns}	–	**−6.2 [−8.3, −3.6](6)**$(^{*})$	–	–	–
Top	−11.6 [−20.4, −0.4](8)^{ns}	–	0.0 [−6.0, 2.8](5)^{ns}	–	–	−6.2 [−14.2, −2.9](4)^{ns}
Total K						
FF (+ Top)	0.0 [−7.7, 5.3](9)^{ns}	–	0.0 [−5.6, 3.7](7)^{ns}	–	–	–
Total Ca						
FF	**−6.6 [−11.9, −4.6](7)***	–	**−9.1 [−15.4, −4.5](6)***	–	–	–
Total Mg						
FF	0.0 [−8.2, 7.6](7)^{ns}	–	−4.0 [−8.3, 4.2](6)^{ns}	–	–	–
Available N						
FF	+3.2 [−8.3, 9.7](13)^{ns}	–	−0.9 [−8.3, 5.8](13)^{ns}	–	–	–
Top	−15.7 [−22.2, 9.8](12)^{ns}	–	+5.7 [−11.8, 16.6](8)^{ns}	–	**−23.9 [−41.6, −11.3](6)**$(^{*})$	+2.2 [−9.6, 14.4](3)^{ns}
Available P						
FF	**−9.6 [−19.3, −5.7](13)***	–	**−12.0 [−19.3, −8.2](13)***	–	–	–
Top	**−21.4 [−39.5, −7.8](13)***	–	−12.5 [−30.8, −3.2](10)^{ns}	–	**−25.6 [−28.1, −21.4](5)***	–
Cation exchange capacity						
FF	**−12.5 [−17.0, −3.9](11)***	+0.3 [−10.1, 6.8](4)^{ns}	**−12.5 [−17.0, −3.9](11)***	–	–	–
Top	**−10.1 [−17.8, −4.5](11)****	+1.0 [−6.7, 1.2](5)^{ns}	**−9.9 [−16.7, −3.4](8)**$(^{*})$	–	–	−21.0 [−27.7, −13.5](3)^{ns}
Deep	**−23.9 [−27.1, −20.6](4)***	–	**−22.8 [−28.1, −18.4](3)**$(^{*})$	–	–	–
Base saturation						
FF	**−5.8 [−11.3, −3.7](11)****	−4.0 [−5.8, −1.7](4)^{ns}	**−8.4 [−13.1, −4.1](11)****	–	–	–
Top	**−13.2 [−23.1, −6.7](9)***	−4.8 [−8.0, −0.8](5)^{ns}	**−17.4 [−19.8, −12.2](7)***	–	–	–
Deep	+8.3 [−1.6, 17.7](4)^{ns}	–	+2.2 [−5.5, 15.0](3)^{ns}	–	–	–
Soil pH						
FF	−0.4 [−1.9, 2.0](29)^{ns}	0.0 [−0.5, 0.4](4)^{ns}	−0.5 [−1.6, 2.0](29)^{ns}	–	–	0.0 [−0.7, 1.3](3)^{ns}
Top	−0.2 [−1.3, 0.4](22)^{ns}	−0.5 [−1.1, −0.3](4)^{ns}	**−0.2 [−2.2, 0.1](19)**$(^{*})$	–	+2.4 [0.9, 3.2](5)^{ns}	–
Deep	−0.3 [−0.7, 1.2](7)^{ns}	–	−0.3 [−0.6, 1.2](7)^{ns}	–	–	–

Table 2 (*continued*)

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB))					Compared to *double slash*
	All treatments	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage	S(WB)R or S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	S(WB)BF + forest floor Stem (wood + bark) + branches + foliage + forest floor	S(WB)BF Stem (wood + bark) + branches + foliage
Exchange acidity						
FF	+46.6 [6.4, 57.0](5)^{ns}	−2.1 [−4.3, 7.8](4)^{ns}	**+78.3 [15.4, 102.5](5)**(*)	–	–	–
Top	+2.6 [−2.5, 6.2](4)^{ns}	+4.8 [−0.3, 7.5](4)^{ns}	+0.4 [−4.8, 5.0](4)^{ns}	–	–	–
Exchangeable H						
FF	+11.9 [−2.4, 31.3](7)^{ns}	**+15.3 [8.5, 24.6](4)**(*)	+20.7 [−2.4, 33.7](7)^{ns}	–	–	–
Top	+6.2 [−2.5, 18.1](4)^{ns}	+1.1 [−8.1, 17.3](4)^{ns}	+11.3 [3.2, 19.0](4)^{ns}	–	–	–
Exchangeable Al						
FF	**+13.0 [3.7, 32.1](11)***	+10.7 [2.3, 19.7](3)^{ns}	**+13.0 [6.1, 37.5](11)***	–	–	–
Top	−1.4 [−4.5, 4.8](9)^{ns}	+7.6 [1.3, 13.7](5)^{ns}	−3.0 [−10.3, 0.2](7)^{ns}	–	–	–
Deep	**−22.6 [−29.1, −10.3](5)**(*)	–	−16.4 [−24.2, −4.9](4)^{ns}	–	–	–

Median value [Quartile 1, Quartile 3] (number of case studies).
Bold values indicate statistically significant effects. Statistical significance: not significant ($P > 0.1$), ns; $P < 0.1$, (*);
* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$.
[a] WD, woody debris; FF, forest floor; Top, top mineral soil layer; Deep, deep mineral soil layer.
[b] Not determined (number of case studies < 3).

significantly decreased by removing harvesting residues. Under S(WB)BF and S(WB)BF + forest floor harvest treatments, reductions were observed on tree height (−3%), tree diameter (−4% to −7%) and tree volume, basal area and biomass (−3% to −7%; Table 4). There were also decreases in tree growth under residues harvest treatments compared to double slash treatments (per cent change = −10% to −14%), and results suggested that the effects increased with increasing harvest intensity. An exception was an increase in tree growth in some cases (up to +20% increase), when stumps and associated coarse roots are removed (S(WB)R or S(WB)BF treatment; Fig. S2).

Like for soil response, results revealed high inter-site differences in the effect on tree growth response and the data enabled an assessment of the effect of elapsed time and some comparisons among locations, soil types, and vegetation types. Differences were found among locations in tree survival, with significant increase in Scandinavia but not North America (Fig. S12). In addition, tree growth was overall negatively and significantly impacted by removing harvesting residues in European countries while only trends could be observed in North America (Fig. S12). Although there was no significant effect of elapsed time (two classes studied: 0–10 year and >10 year, Fig. S13) the data suggested stronger positive or negative impacts during the first years after harvesting (see Fig. S2). Using data from Scandinavia, no significant relationship was found between tree response to the removal of harvesting residues and the site index (Fig. S14); because there was no common method for determining inherent soil fertility, no relationship could be assessed using the whole dataset. Finally, trees growing on gleysols and podzols tended to be more impacted than trees growing on acrisols and cambisols, and growth of coniferous tree species tended to be more affected than that of broadleaf tree species. For instance, there were significant decreases in total height, diameter, volume, basal area and biomass for coniferous ($P = 0.007$–0.022; $n = 19$–41; −2.8% to −4.4% decreases in treatment S(WB)BF), but not for broadleaf trees ($P > 0.492$; $n = 6$–7).

4. Discussion

4.1. Theoretical nutrient outputs

Based on nutrient stock data, we found that removing harvesting residues can lead to theoretical increases in nutrient outputs, which are considerable especially when the foliage is removed. Under the most intensive harvests, theoretical nutrient outputs could represent up to 100% of available/exchangeable P, K, Ca and Mg stocks in soils. This suggests potential negative impacts on chemical soil fertility if processes, such as mineral weathering, are too low to compensate those large nutrient exports. Based on realistic harvest rates, we found that nutrient exports were notably lower, demonstrating the importance of using realistic harvest rates to evaluate nutrient outputs caused by harvesting tree biomass. The harvest treatments that left most foliage and/or bark on site considerably reduced the nutrient costs of removing harvesting residues (e.g. treatments with a delay between cutting the stem and harvesting the branches, thus allowing the foliage to dry and fall off the branches). The increase in nutrient exports were highest in young stands probably because the relative contribution of foliage and thin branches (i.e. tree components that are rich in nutrients; André and Ponette, 2003; Augusto et al., 2008a; André et al., 2010) to above stump biomass decreases with increasing age.

4.2. Overall consequences of removing harvesting residues on soils and trees

To meet our first objective, we combined all the results of the present study and provide an overview of the impacts of intensive

Table 3
Percent changes in soil organic matter quality, environmental conditions, soil fauna and microbiological soil properties due to removing harvesting residues ("environmental impacts" dataset).

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB))					Compared to *double slash*
	All treatments	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage	S(WB)R or S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	S(WB)BF + forest floor Stem (wood + bark) + branches + foliage + forest floor	S(WB)BF Stem (wood + bark) + branches + foliage
Organic matter quality: C/N ratio						
FF[a]	**+2.3 [0.0, 5.7](30)**([*])	+0.5 [−0.6, 2.6](5)[ns]	+1.6 [−0.7, 5.6](30)[ns]	–	−3.6 [−4.5, 22.7](5)[ns]	+1.3 [−1.9, 2.9](6)[ns]
Top	+1.1 [−4.8, 5.2](27)[ns]	+2.7 [0.4, 5.4](5)[ns]	+1.2 [−3.2, 5.0](22)[ns]	–	−7.7 [−12.2, −1.7](6)[ns]	+4.2 [−0.7, 4.9](3)[ns]
Deep	−4.4 [−16, 6.8](5)[ns]	–[b]	−4.2 [−20.7, 7.8](4)[ns]	–	–	–
Bulk density						
Top	**+4.5 [1.1, 6.5](21)**[*]	–	**+3.7 [0.8, 8.3](12)**([*])	+5.2 [3.4, 5.7](5)[ns]	+3.6 [−3.7, 10.2](6)[ns]	–
Mean soil temperature						
Top	**+4.9 [−1.8, 14.0](22)**[**]	–	**+5.4 [−2.1, 10.8](17)**[*]	–	**+9.9 [1.6, 18.7](12)**[*]	–
Mean soil moisture						
Top	−3.3 [−8.1, 8.7](11)[ns]	–	+5.4 [−1.2, 11.7](6)[ns]	–	−8.5 [−12.4, −3.4](7)[ns]	–
Fauna: number of individuals (mites, springtails, millipedes, nematodes, annelids, beetles, etc.)						
Top	+25.1 [−3.7, 53.7](9)[ns]	–	+50.4 [14.4, 101.9](6)[ns]	–	+25.1 [6.7, 42.2](3)[ns]	−3.7 [−15.4, −0.8](3)[ns]
Fauna: species richness						
Top	−4.8 [−22.5, 50.8](5)[ns]	–	−4.8 [−16.8, 50.8](5)[ns]	–	–	–
Microbial C						
FF	−1.8 [−6.5, 8.5](12)[ns]	–	−3.0 [−7.5, 8.5](12)[ns]	–	–	0.0 [−1.1, 3.4](5)[ns]
Top	**−14.3 [−20.4, −11.6](8)**[***]	–	−19.1 [−22.3, −10.7](4)[ns]	–	**−13.9 [−14.7, −12.9](5)**[**]	–
Microbial N						
FF	−0.4 [−7.6, 5.6](15)[ns]	–	0.0 [−7.6, 8.2](15)[ns]	–	+0.2 [−6.2, 5.8](4)[ns]	−6.0 [−6.3, 0.0](5)[ns]
Top	**−19.6 [−23.2, −13.4](11)**[*]	–	−10.3 [−17.0, −4.3](8)[ns]	–	**−28.3 [−39.2, −15.0](7)**[**]	–
Microbial activity (soil respiration, mainly in incubated soils)						
FF + Top	−8.8 [−12.8, 0.1](14)[ns]	–	**−8.3 [−13.0, −0.8](13)**([*])	–	–	−6.2 [−8.3, −3.1](5)[ns]
Other indicators of microbial activity (fungi ergosterol, fungi and bacterial PLFA, etc.)						
FF + Top	−4.9 [−13.9, −1.1](5)[ns]	–	–	–	–	–
All indicators of microbial activity combined						
FF	−2.1 [−7.9, 8.8](16)[ns]	–	−0.4 [−7.2, 10.1](15)[ns]	–	+2.2 [−6.2, 16.2](4)[ns]	−4.4 [−5.6, −2.3](6)[ns]
Top	**−13.9 [−20.6, −8.4](15)**[**]	–	−9.4 [−15.2, −2.7](11)[ns]	–	**−22.6 [−32.8, −12.2](9)**[**]	–
Microbial C/N						
FF	+0.8 [−1.0, 8.5](12)[ns]	–	+0.9 [−0.7, 6.8](12)[ns]	–	–	−0.4 [−1.1, 10.0](5)[ns]
Top	+3.6 [0.6, 5.3](6)[ns]	–	+0.8 [−2.5, 4.8](4)[ns]	–	+5.5 [0.0, 24.0](3)ns	–
Soil CO_2 efflux						
FF + Top	−7.0 [−21.6, 1.0](8)[ns]	–	**−9.7 [−21.5, −4.3](5)**([*])	–	–	–
Enzymatic activity (enzymes grouped into three functional groups based on their abilities to decompose C substrates and to release N and P)[c]						
FF + Top/C	**−25.2 [−28.6, −10.2](7)**[*]	–	−12.5 [−28.4, −3.7](6)[ns]	–	–	**−41.5 [−50.6, −30.9](4)**[*]
FF + Top/N	**−43.3 [−46.7, −23.5](7)**[**]	–	**−36.6 [−44.5, −24.5](6)**[**]	–	–	**−50.5 [−51.4, −44.6](4)**[*]
FF + Top/P	**−18.9 [−30.7, −6.5](7)**([*])	–	−10.1 [−17.6, 3.7](6)[ns]	–	–	**−30.3 [−36.6, −22.6](4)**[*]
FF + Top/all	**−31.8 [−36.7, −21.1](7)**[**]	–	**−22.6 [−31.0, −19.0](6)**[**]	–	–	**−41.5 [−44.5, −37.0](4)**[**]
Debris decomposition (mass losses)						
WD	−5.3 [−35.6, −1.3](3)[ns]	–	−5.3 [−46.4, 5.1](3)[ns]	–	–	–
Net N mineralization						
FF	−19.8 [−41.3, −12.0](12)[ns]	–	**−27.4 [−41.8, −13.5](12)**([*])	–	–	−14.8 [−19.6, 0.0](5)[ns]
Top	−16.2 [−23.3, −1.6](16)[ns]	–	−5.4 [−14.8, 14.5](12)[ns]	–	**−23.1 [−47.8, −5.9](11)**[*]	**−25.5 [−25.6, −21.1](3)**[*]
FF + Top	−18.2 [−24.4, −1.2](27)[ns]	–	−14.5 [−29.1, 5.2](23)[ns]	–	−22.8 [−42.2, 2.4](12)[ns]	−18.2 [−25.6, −11.1](8)[ns]

Table 3 (*continued*)

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB))					Compared to *double slash*
	All treatments	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage	S(WB)R or S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	S(WB)BF + forest floor Stem (wood + bark) + branches + foliage + forest floor	S(WB)BF Stem (wood + bark) + branches + foliage
Nitrification						
FF + Top	+14.5 [−1.7, 52.9](6)^{ns}	–	−12.9 [−25.9, −1.7](4)^{ns}	–	**+103.0 [66.0, 142.3](3)(*)**	–
N Immobilization						
Top	−4.7 [−52.2, 14.0](3)^{ns}	–	–	–	–	–

Median value [Quartile 1, Quartile 3] (number of case studies).
Bold values indicate statistically significant effects. Statistical significance: not significant ($P > 0.1$), ns; $P < 0.1$, (*);
[*] $P < 0.05$.
[**] $P < 0.01$.
[***] $P < 0.001$.
[a] WD, woody debris; FF, forest floor; Top, top mineral soil layer; Deep, deep mineral soil layer.
[b] Not determined (number of case studies < 3).
[c] Ability to decompose labile or recalcitrant C substrates (hydrolysis of cellulose, starch, glucose, lignin): cellobiohydrolase, α-1,4-glucosidase, β-1,4-glucosidase, peroxidase, phenol oxidase, dehydrogenase. Ability to hydrolyze organic N (glucosamine or amino acids): β-1,4-N-acetylglucosaminidase, protease. Ability to hydrolyze phosphate: acid and alkaline phosphatases.

harvests on forest ecosystem processes, compared with conventional stem-only harvest (Fig. 5). However, soil and tree responses varied greatly among case studies, depending on site conditions and/or on the intensity of residue harvest; the overall effects summarized in Fig. 5 rather correspond to the most intensive harvests, such as S(WB)BF. It should therefore be noted that significant impacts were not always found when harvesting residues (i.e. branches) were exported without foliage (i.e. under the S(WB)B treatment).

4.2.1. Consequences for chemical soil fertility

The results we obtained with the "nutrient stocks" dataset (changes in nutrient outputs and comparison with nutrient stocks in soils) may suggest that intensive harvests have a negative impact on chemical soil fertility. However, previous studies have shown that increased nutrient output or immobilization in trees does not systematically cause depletion of total or available soil nutrients, owing to several processes including the soil buffer capacity (e.g. ability of soils to provide base cations through mineral weathering; Kimmins, 1974; Ranger and Turpault, 1999; Bélanger et al., 2004) or the dynamic response of trees to different levels of nutrient availability (e.g. ability to promote mineral weathering). Nevertheless, the "environmental impacts" dataset enabled us to show overall reductions in total N and P stocks, as well as reductions in available soil P and 'base cation' saturation. The decrease in base saturation was the result of a concomitant decrease in non-acidic cations (Ca^{2+}, Mg^{2+}, K^{+}) and an increase in exchangeable H and Al (Olsson et al., 1996; Iwald et al., 2013). There was also a decrease in the cation exchange capacity, probably due to a reduction in the amount of soil organic matter, changes in its chemical composition (Thiffault et al., 2008), and maybe because of Al polymerization inside clay minerals caused by acidification (Augusto et al., 2001).

Soil nutrient leaching is expected to be low in the case of intensive harvests in which inputs of organic matter available for mineralization are reduced, compared with conventional stem-only harvest (Adams, 1999; Arocena, 2000). Yet, comparing the results of several experiments revealed no clear evidence for this effect because of several interactions with soil fertility and vegetation (Blumfield and Xu, 2003; Devine et al., 2012). Therefore, leaching could also exacerbate losses of several soil nutrients (with the exception of phosphorus) when forest residues are harvested. In addition to increased nutrient outputs through harvesting of tree biomass, and in some cases through leaching, the decrease in soil microbiological activity could also explain the reduced amounts of available nutrients in the soil in intensively harvested sites.

4.2.2. Decrease in biological soil activity and decomposition processes

Soil biological activity and decomposition/mineralization of soil organic matter play a crucial role in forest functioning as they make nutrients more available to the trees (Ranger et al., 2011). Relatively to stem-only harvest, microbial activity, enzymatic activities and N mineralization fluxes were reduced in intensive harvests. Because soil CO_2 efflux has two components (heterotrophic and autotrophic), the significant and negative impacts on this variable may be related to reductions not only in microbial activity but also in root respiration (decreases in both processes were found in one case study; Versini et al., 2013). The overall effects on microbial activity and mineralization fluxes may be explained by other effects on organic matter amounts and composition (Smolander et al., 2013) and environmental factors such as compaction, soil temperature, moisture and pH (see below).

In sites where harvesting residues are removed, the quantity of organic matter in soils is significantly reduced and its quality may be affected through an increase in the C:N ratio and changes in

Table 4
Percent changes in nutrient status, survival and growth of trees due to removing harvesting residues ("environmental impacts" dataset).

	Harvest treatments compared to conventional stem-only harvest (wood + bark, S(WB))					Compared to *double slash*
	All treatments	S(WB)B Stem (wood + bark) + branches	S(WB)BF Stem (wood + bark) + branches + foliage	S(WB)R or S(WB)BFR Stem (wood + bark) + branches + foliage + stumps/roots	S(WB)BF + forest floor Stem (wood + bark) + branches + foliage + forest floor	S(WB)BF Stem (wood + bark) + branches + foliage
Nutrient status of trees						
Foliar mass	−3.1 [−5.3, 3.0](19)^{ns}	–[a]	−1.1 [−3.0, 2.6](18)^{ns}	–	–	−7.3 [−7.6, 0.1](6)^{ns}
Foliar N Concentration	0.0 [−2.5, 2.0](37)^{ns}	0.0 [−1.7, 2.2](6)^{ns}	+0.8 [−2.0, 2.6](29)^{ns}	–	−0.1 [−0.7, 1.5](6)^{ns}	−1.9 [−3.5, −1.2](7)^{ns}
Foliar P Concentration	−0.2 [−3.6, 2.3](32)^{ns}	−2.6 [−4.7, 0.6](6)^{ns}	0.0 [−1.7, 3.1](28)^{ns}	–	0.0 [−2.5, 0.7](3)^{ns}	0.0 [−3.6, 3.8](7)^{ns}
Foliar K Concentration	−0.1 [−4.6, 4.1](32)^{ns}	+2.8 [−0.4, 13.6](7)^{ns}	+1.1 [−4.5, 4.6](28)^{ns}	–	−3.4 [−3.7, 0.6](3)^{ns}	**−8.1 [−14.6, −5.1](7)***
Foliar Ca Concentration	**−3.6 [−10.9, 2.9](32)***	−2.0 [−8.1, −0.3](7)^{ns}	**−3.6 [−7.0, 2.5](28)***	–	−22.5 [−36.9, −7.1](3)^{ns}	−2.7 [−10.6, 2.9](7)^{ns}
Foliar Mg Concentration	0.0 [−4.5, 4.6](32)^{ns}	−1.9 [−2.5, 2.9](7)^{ns}	0.0 [−4.6, 4.5](28)^{ns}	–	+0.8 [−6.7, 1.4](3)^{ns}	0.0 [0.0, 7.8](7)^{ns}
Foliar Mn Concentration	−3.3 [−7.9, 1.0](10)^{ns}	+1.7 [−0.3, 3.4](4)^{ns}	**−5.1 [−11.2, −2.3](9)**(*)	–	–	–
Foliar Zn Concentration	−3.3 [−4.4, 1.5](9)^{ns}	−2.2 [−4.2, 1.0](4)^{ns}	−3.3 [−4.4, 2.1](9)^{ns}	–	–	–
Foliar Cu Concentration	+0.9 [−7.6, 3.3](5)^{ns}	–	+0.9 [−7.6, 3.3](5)^{ns}	–	–	–
Foliar B Concentration	−1.2 [−6.4, 5.8](16)^{ns}	–	+0.2 [−5.8, 5.8](16)^{ns}	–	–	−6.2 [−6.4, −1.2](5)^{ns}
Tree Survival	+1.0 [−3.6, 2.3](21)^{ns}	+2.6 [0.7, 5.9](4)^{ns}	+0.3 [−6.8, 2.9](20)^{ns}	–	+5.3 [−0.2, 10.2](8)^{ns}	–
Tree growth						
Height	**−2.6 [−7.6, 0.5](49)****	−1.7 [−6.9, −0.1](6)^{ns}	**−2.8 [−6.9, 0.5](45)****	–	−3.2 [−12.2, 2.3](15)^{ns}	**−10.6 [−10.6, −9.7](3)****
Height increment	−10.9 [−18.8, 0.0](9)^{ns}	–	−1.5 [−11.9, 1.2](7)^{ns}	–	−16.3 [−45.1, −2.1](5)^{ns}	–
Diameter	**−3.5 [−9.6, 0.0](30)****	–	**−4.4 [−8.9, 0.0](25)***	–	**−7.5 [−12.8, −1.6](9)**(*)	**−10 [−13.1, −9.0](3)***
Diameter increment	−0.6 [−17.4, 1.8](8)^{ns}	–	+0.2 [−10.4, 11.2](6)^{ns}	–	−9.9 [−24.9, 1.7](5)^{ns}	–
Volume, basal area or biomass	**−5.2 [−16.1, 0.3](57)***	−6.2 [−12.1, 0.9](6)^{ns}	**−3.1 [−15.1, 2.8](48)**(*)	–	−15.0 [−30.1, −5.2](13)^{ns}	**−14.1 [−29.4, −3.8](10)***
Volume, basal area or biomass increment	**−7.0 [−12.4, −3.3](34)*****	–	**−7.5 [−12.6, −3.3](33)*****	–	–	−12.5 [−22.3, −11.9](3)^{ns}
Root biomass	−10.1 [−20.2, 0.3](6)^{ns}	–	−0.6 [−6.3, 4.4](4)^{ns}	–	–	–

Median value [Quartile 1, Quartile 3] (number of case studies).
Bold values indicate statistically significant effects. Statistical significance: not significant ($P > 0.1$), ns; $P < 0.1$, (*);
* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$;
[a] Not determined (number of case studies < 3).

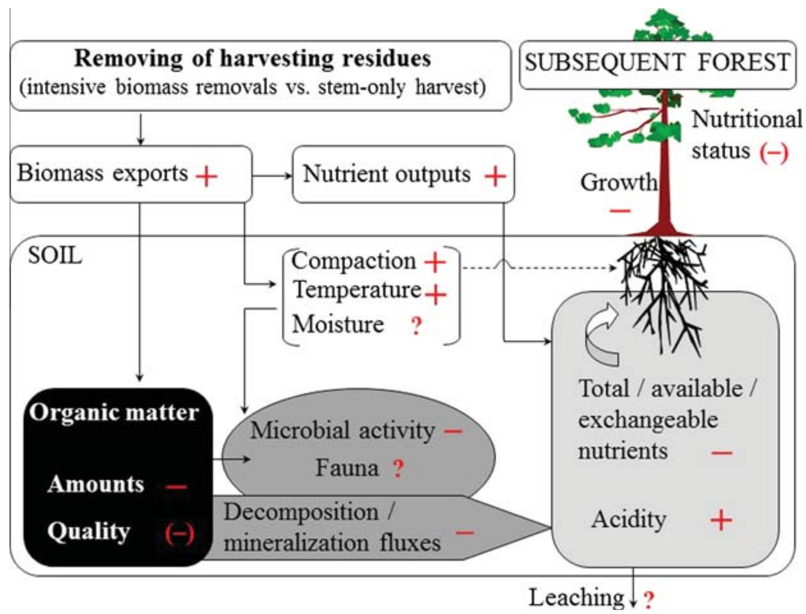

Fig. 5. Overview of the main impacts of removing harvesting residues on forest ecosystem functioning, based on the results of the meta-analysis. Negative effects are denoted by a minus sign. Positive effects are denoted by a plus sign. Signs in brackets denote trends (non-significant results or low number of sites/studies). Question marks denote unclear results. The figure mainly focuses on the effects of the most intensive removals, such as S(WB)BF or S(WB)BF + forest floor. There were far fewer impacts when the branches were exported without foliage (i.e. in S(WB)B).

chemical composition. There were reductions in the amounts of labile compounds characterized by rapid mineralization due to their biochemical nature and to the lack of protection by soil colloids (such as the light soil organic matter fraction; Huang et al., 2011a,b), but also reductions in stable compounds (alkyl C, lignin- and cutin-derived C). These changes in organic matter composition can be explained by the fact that removing harvesting residues deprives the soil of inputs of recent organic matter and certain compounds derived from plant litter (see more details on the effects of residues removal on chemical composition of light and heavy soil organic matter and other compounds and plant biomarkers in Mathers et al. (2003), Mathers and Xu (2003), Thiffault et al. (2008), Huang et al. (2011a,b) and Smolander et al. (2013)). On the other hand, effects of removing residues at harvest on soil organic matter quality may be limited because of the large amount of high quality biomass that is delivered to the soil during a rotation period. Finally, one could expect that reduced amounts of organic matter and changes in its quality could decrease mineralization fluxes and hence the supply of plant-available nutrients (O'Connell et al., 2003, 2004).

Soil micro-organisms may also be affected by the environmental changes documented in this study. Removing harvesting residues had an effect on soil pH, which is known to affect microbial activity (Fuentes et al., 2006). Decreases of soil pH were however small and micro-organisms were probably more affected by the changes in organic matter quality and the lack of fresh plant material supporting C sources. Removing harvesting residues also increases soil compaction, which was inferred from an increase in soil bulk density in the present meta-analysis or from an increase in soil strength (Carter et al., 2006; Han et al., 2009). Indeed, removing harvesting residues prevents the creation of slash mats, which distribute the weight of the harvesters, skidders and forwarders over a larger area and hence reduce direct contact between the machines and the soil surface (Han et al., 2009). Effect on soil compaction can also be explained by differences among harvest treatments in machine characteristics (e.g. machine mass and how it is distributed over the wheels, and hence pressure on the soil) as well as harvesting operations (number of machine passes; Ampoorter et al., 2012; Han et al., 2009) and may vary among countries owing to differences in harvest practices (e.g. two passes in the European system vs. one pass in the full-tree-to-roadside system in North America). Soil compaction could in turn reduce soil porosity and fluid transfer (water and gas) (Wilpert and Schäffer, 2006; Startsev and McNabb, 2009) thereby reducing microbial activity (Jordan et al., 2003) and soil CO_2 efflux (Goutal et al., 2012). Soil microbial activity may also be affected by microclimate changes. In particular, soil temperature increased significantly when residues were removed probably because soils were more exposed to solar radiation (Proe et al., 2001; O'Connell et al., 2004). Moreover, positive effects of removing harvesting residues on spring and summer temperatures and no -or negative- effects on fall and winter temperatures suggest that residues play a role in regulating seasonal variations; a role in diurnal variations has also been demonstrated (O'Connell et al., 2004). The increase in mean annual soil temperature could lead to an increase in mineralization rates (in % of soil organic matter) under intensive harvests (Dessureault-Rompré et al., 2010), thus contributing to organic matter losses. But, as shown in a previous study (O'Connell et al., 2004), mineralization fluxes in the forest floor and top mineral soil could remain at a lower level under these treatments mainly because of reduced inputs of organic matter and hence in substrates available for mineralization. In addition, an increase in soil temperature in summer and a decrease in winter could hamper mineralization processes by creating suboptimal conditions. Through the *mulching effect*, residues could also play a role in maintaining soil moisture, which in turn affects microbial activity (O'Connell et al., 2004; Roberts et al., 2005). However, in the present study, no generally and significant effect of removing harvesting residues was found on topsoil moisture, perhaps because of antagonistic effects of soil compaction (increase in soil water saturation, Goutal et al., 2012) and decreased *mulching effect*.

In addition to soil micro-organisms, soil fauna also plays a role in decomposition processes and the density and diversity of soil fauna were recently reported to be potentially affected by slash removal and the associated chemical and physical soil disturbances (Bouget et al., 2012). However, in the present study, we did not find any significant effect of removing harvesting residues on those faunal characteristics.

4.2.3. Consequences for tree growth

Results revealed negative impacts of removing harvesting residues on tree growth, which is probably due to a combination of several effects on soil, such as reduced chemical soil fertility (and hence tree nutrition). The increased soil compaction could also have negatively impacted tree growth, owing to an effect on root penetration and hence nutrient uptake; Kozlowski, 1999; Kabzems and Haeussler, 2005; Wilpert and Schäffer, 2006), but the opposite (positive effects of compaction on tree growth) has also been shown in compaction experiments (in coarse textured-soils in North America LTSP studies; Ponder et al., 2012). In addition to the vital role of residues and their decomposition for tree nutrition (particularly in soils with low fertility; Laclau et al., 2010; Ranger et al., 2011), residues could have an indirect effect on tree survival and growth through a negative effect on the density of competitive vegetation (Harrington and Schoenholtz, 2010). Although the general impacts on trees are negative (reduced growth), positive effects of residues removal were reported in some case studies (Fig. S2); these positive effects may be due to more favorable soil conditions (higher soil temperature; Roberts et al., 2005) or reduced root disease (when stumps and associated coarse roots are removed; Cleary et al., 2013). Positive effects on tree biomass have also been reported after stump harvesting and deep soil cultivation because soil disturbance stimulated N mineralization leading to increased tree nutrition (Egnell et al., 2015). The largest effects on growth are observable in the few years following residues harvesting, which corresponds to the seedling stage. Indeed, despite low nutrient requirements in absolute values, seedlings rely relatively more on topsoil supply because of lower nutrient reserves and soil exploration by their roots than older individuals, such as saplings and mature trees. Consequently, seedlings are probably more exposed to changes of growth rate. In addition, soil changes induced by intensive harvests, in comparison to stem-only harvest, tend to become insignificant with time. This trend may contribute to the convergence of treatments in terms of tree growth after one decade.

The absence of significant tree growth response in North America is in agreement with previous findings based on the LTSP installations; it was explained by the facts that most sites were established on productive sites and harvesting operations in less productive sites enabled substantial amounts of residues to be left on-site (Fleming et al., 2006; Ponder et al., 2012).

4.2.4. Recovery of forest ecosystems or long-term effects?

Although removing harvesting residues had significant and negative impacts on soil fertility and tree growth, the magnitude of changes was generally low (e.g. decrease of only 3–7% in tree growth). In addition, results suggest that chemical soil fertility may recover because stronger negative impacts on total N and exchangeable cations tend to occur during the first decade after harvesting. Some tree responses tend to display a similar pattern as the biggest changes in tree diameter and height occurred during the first years after harvesting. Soil recovery may occur through for instance tree litterfall and/or mineral weathering processes, which compensate for organic matter and nutrient losses (Ranger et al., 2011). However, in addition to the comparison with other harvest treatments, the comparison to pre-harvest conditions (e.g. such as in Nave et al., 2010) is also needed to determine if a recovery occurs.

In the "environmental impacts" dataset, the time elapsed after harvesting ranged between 0.2 and 33 years, and the slight impacts we found rather correspond to short or medium term effects and are generally the result of a single harvest (two harvests at thinning in Scandinavian studies), which may allow the forest ecosystem to recover. For the future, trials where harvesting residues have been collected once should be monitored on the long term (e.g. one complete rotation). Cumulative impacts of repeated residue removals should also be studied. Effects of repeated removal of harvesting residues in thinning and final felling have started to be studied in Finnish experiments (Tamminen and Saarsalmi, 2013; Kaarakka et al., 2014). Alternatively to long term experiments, effects due to cumulated intensive harvests have been assessed through modeling approaches, and showed a 20–40% decrease in forest productivity after 3–5 rotations (Peng et al., 2002), as well as long term effects on soil organic matter and nutrients (Peng et al., 2002; Aherne et al., 2008; Ranatunga et al., 2008; Scheller et al., 2011).

4.3. Limits of the study: causes of heterogeneity and predictors

Concerning the "nutrient stocks" dataset, it should be noted that even though soil fertility may influence tree nutrient concentrations and hence nutrient outputs, no site-specific relationship was calibrated based on available data.

Concerning the "environmental impacts" dataset, data were not equally distributed among geographic locations. As a consequence, most effects we found are representative of North America and/or Europe. Moreover, the European sub-dataset is mainly representative for Scandinavia. Data are also not equally distributed among soil types, podzol being the most represented in several cases. Data distribution and, for some soil variables, data scarcity hampered our assessment of the reasons for the heterogeneity of soil and tree responses. Only an effect of the time that elapsed after harvesting and some differences among climate classes and tree species were found. However, these effects are tricky to disentangle because they are region-dependent. For instance, for the effects on total tree height, diameter, volume, basal area and biomass, the >10 year class is largely represented by Scandinavian sites, while both North America and Scandinavia are well represented in the 0–10 year class. The effect of the climatic class is also difficult to assess because most cold sites are in Scandinavia where treatments are applied mostly at thinning and not at clear-cutting as done in other regions of the World. Finally, differences among tree species may partly reflect an effect of soil fertility, since coniferous trees are generally planted on infertile soils. Tree response to the harvesting of residues depends on several factors, among which certain were not systematically mentioned (e.g. nutrient outputs at harvest) or not quantified using the same methods among studies (e.g. initial amounts of available nutrients in the soils, or site index). As a consequence, it was not possible to assess relationships between tree growth response to intensive harvests and soil fertility indices (e.g. Scott and Dean, 2006; Fig. S14) using the whole "environmental impacts" dataset.

4.4. Prevention and mitigation measures to reduce impacts on soil fertility and tree growth

To prevent the negative effects of removing harvesting residues on site fertility (also on water quality and biodiversity) from occurring, general and/or site-specific guidelines have been developed in many countries. For instance, it is generally recommended to avoid (or limit) removing harvesting residues on *sensitive* soils such as shallow, highly acidic, highly weathered or coarse textured soils (e.g. Pinchot institute, 2010; Dickinson et al., 2012).

Besides site-specific considerations, and when harvest practices are not sustainable, measures should be taken to reduce the environmental consequences. The most impacting harvests tested in our study were clearly those including a removal of foliage [i.e. S(WB)BF]. Removing foliage strongly increases nutrient outputs, while the gain in harvested biomass is low. The strong imprint of foliage export is clearly visible in estimates of nutrient exports, reductions in soil nutrient stocks, and growth of subsequent forest stand. In comparison, harvests which avoid collecting foliage [i.e. S(WB)B] have little or no impact. Our results are in good agreement with modeling studies in Finland, which showed that S(WB)B, but not S(WB)BF harvests, had only slight effects on chemical soil fertility (Aherne et al., 2012). Besides the small effects on soil nutrients, treatment S(WB)B had also no significant effects on tree growth. In addition to the high nutrient cost of harvesting foliage, bark also contains high nutrient concentrations (Ponette et al., 2001; André et al., 2010). As a consequence, the nutrient cost of harvesting branches (or roots) could be compensated for by leaving most of the foliage and bark on site. More generally, nutrient costs and impacts on chemical soil fertility were found to increase with increasing harvest intensity. In agreement with previous conclusions (Stupak et al., 2008; Aherne et al., 2012; Augusto et al., 2015), our study thus shows that practical measures that reduce biomass exports (particularly foliage, through a removal after a drying step or the leaf-fall in winter for hardwoods, and bark) could be used to reduce nutrient costs due to removing harvesting residues.

Nutrient costs at harvest and tree growth response were both higher for coniferous tree species than for broadleaf tree species (but maybe in relation with soil fertility), and nutrient costs were higher in young stands than in old stands. Therefore, other measures of prevention may consist in removing harvesting residues preferentially in broadleaf forests and/or mature stands (e.g. at the clear-cut stage). Besides those recommendations, more studies are needed to assess the relationships between tree growth response to the removing of harvesting residues (i.e. site sensitivity) and physical and chemical soil properties (e.g. inherent soil fertility; Scott and Dean, 2006) to define prevention measures adapted to local site fertility.

If, despite the use of general and site-specific guidelines of good practices of biomass harvest, some serious consequences on forest functioning occur, some practices may help the recovery of the ecosystem. A first and easy to apply approach is based on the concept of *ecological length of rotation* (Kimmins, 1974; Ranger and Turpault, 1999). This concept states that the ecological length of a forest rotation is defined as the number of years necessary to processes, such as atmospheric deposition or weathering of soil minerals, to compensate the loss of nutrients induced by biomass export. In practice, extending the rotation could be an easy method to enable a forest to recovering from slight to moderate impacts of former intensive harvests. However, in case of severe disturbances, some forests might be not resilient enough to grow as healthy as before, even with an extended rotation length. In those cases, some mitigation measures, such as applications of fertilizers or wood ash (e.g. Augusto et al., 2008b; Helmisaari et al., 2011), may be used to compensate nutrient losses; the dose of nutrient to apply being possibly estimated using simple allometric relationships (Augusto et al., 2000; see Table S3 which is a by-product of our study). It should however be noted that fertilization, and all over wood ash application, is becoming an option also for preventing negative impacts from occurring in European countries where the development of power stations supplied with biomass consequently produces large amounts of wood ash. Conversely, this mitigation approach is currently prohibited, or discouraged, in other countries such as in North America.

5. Conclusion

We found that removing harvesting residues induces increases in nutrient outputs which can be theoretically considerable, especially when foliage is harvested. We also showed that realistic harvest rates should be taken into account as their use resulted in much lower nutrient costs. In response to our first objective, the concomitant use of our two datasets demonstrated that the most intensive harvests (e.g. of branches + foliage) often has negative impacts on chemical and biological soil fertility and tree growth, but with large disparities among harvest treatments, vegetation types, and stand development stages. Some practical measures can be taken to reduce the environmental consequences of removing harvesting residues. In particular, our results revealed low and/or non-significant negative impacts when branches are exported but the foliage is left on site. Additional mitigation measures need to be developed by establishing the link between site fertility and the intensity of the impact of removing harvesting residues.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2015.03.042.

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Quantifying consequences of removing harvesting residues on forest soils and tree growth – A meta-analysis

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Supplementary Information (14 figures, 3 tables, references used in the meta-analyses)

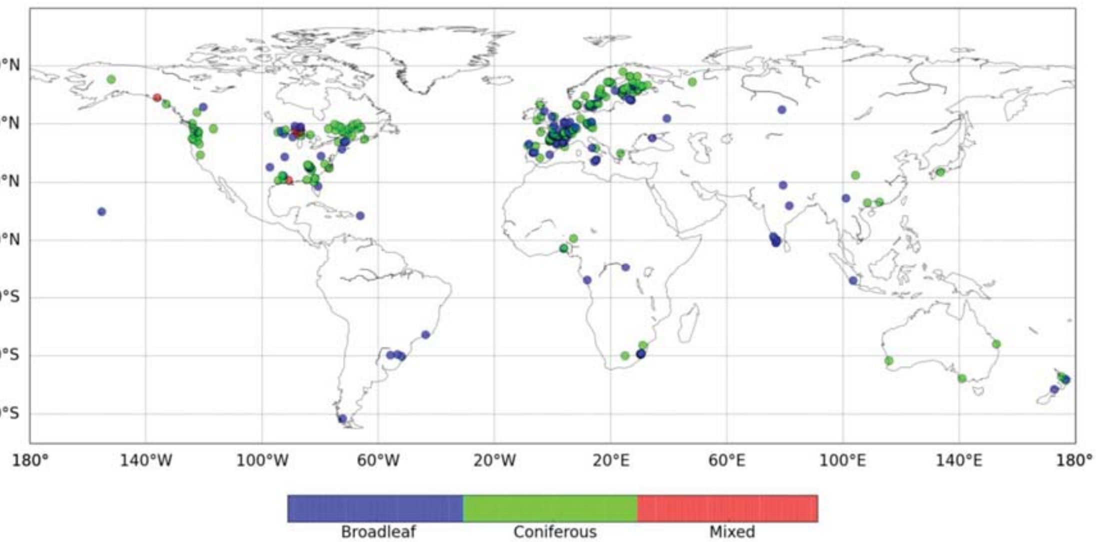

Figure S1 Spatial distribution of the study sites ("nutrient stocks" dataset). Broadleaf in blue, coniferous in green, mixed stands in red.

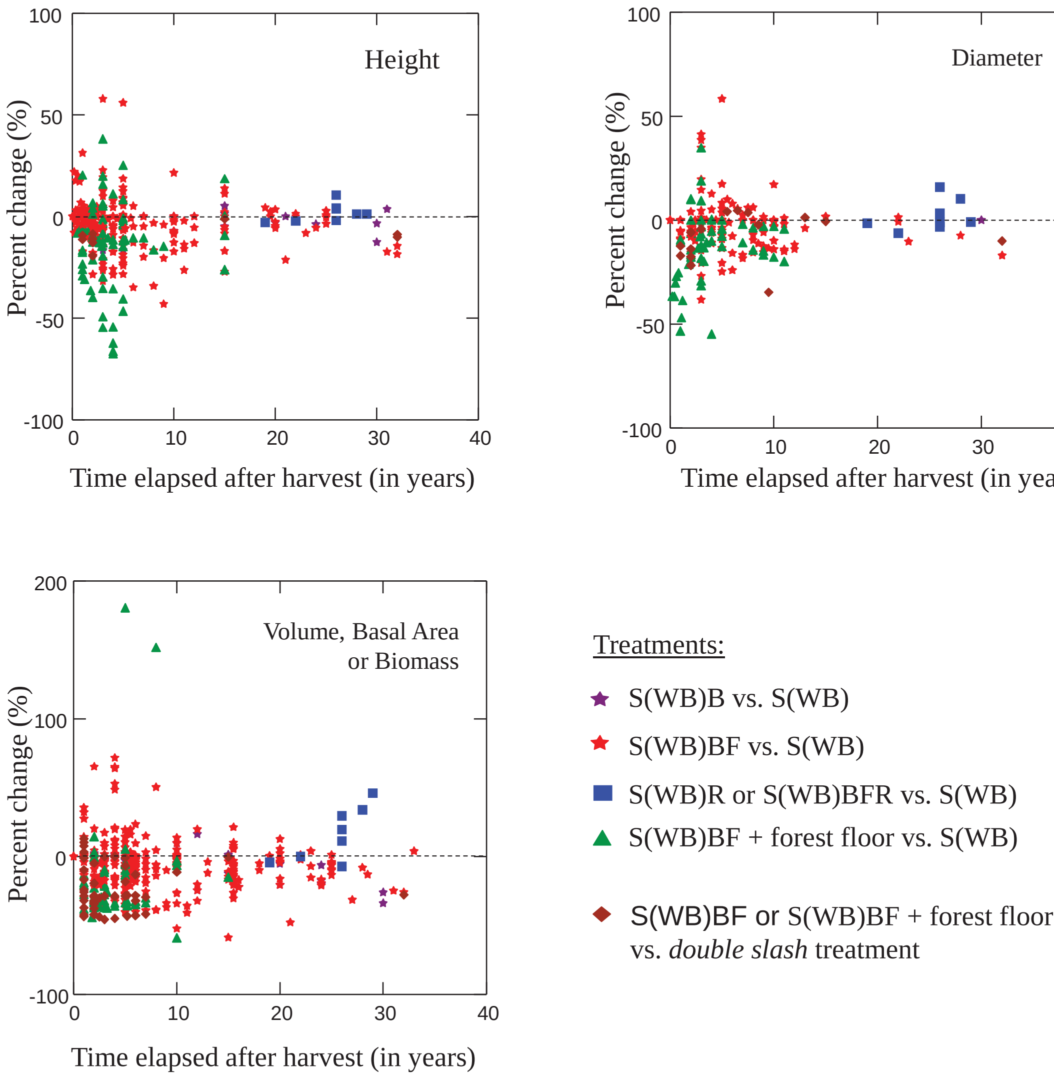

Figure S2 Effects of removing harvesting residues on tree growth as a function of the time elapsed since the removal: complete "environmental impacts" dataset. See explanation for the type of harvest in Figure 2.

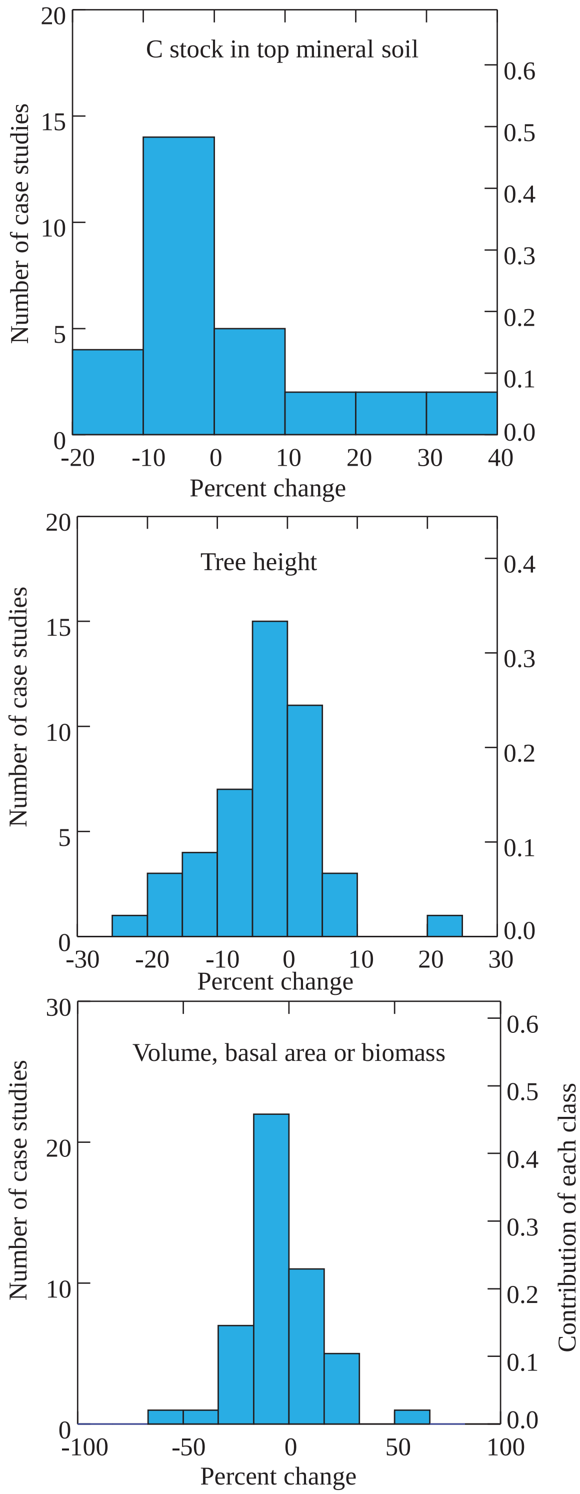

Figure S3 Statistical distribution of the case studies as a function of the percent change. "Environmental impacts" dataset, examples of C and N stocks in top mineral soils and of tree growth under treatment S(WB)BF vs. S(WB).

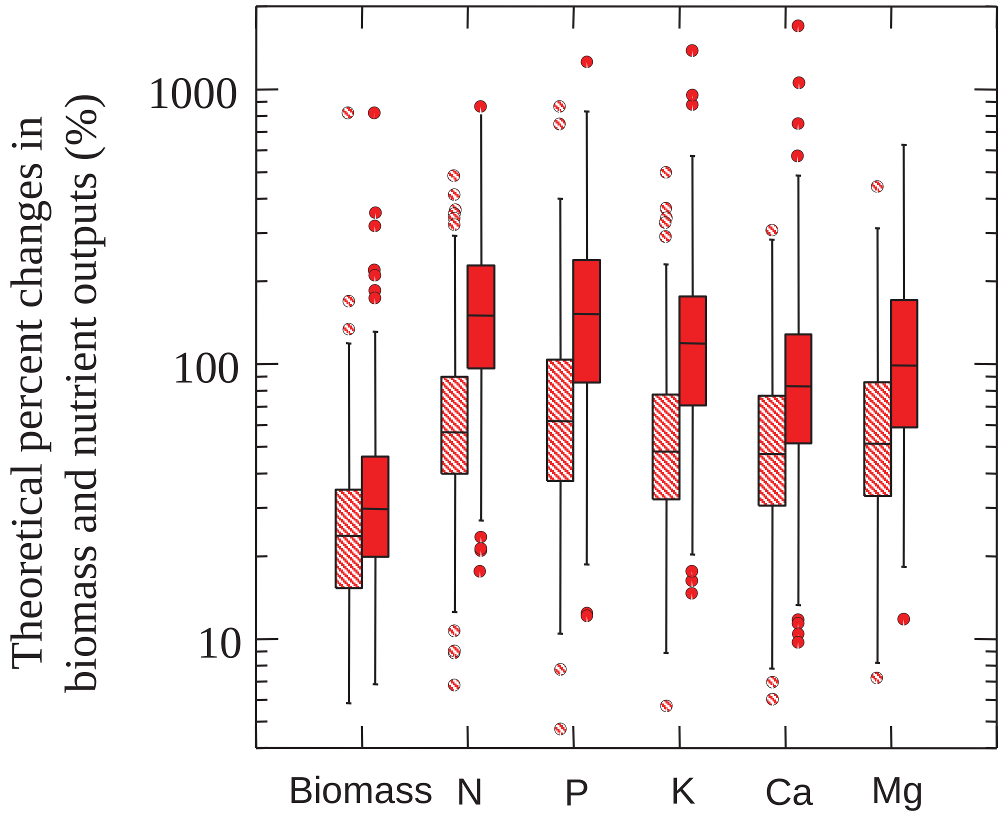

Figure S4 Theoretical percent changes in biomass and nutrient outputs as a result of removing stem + branches (treatment S(WB)B, hatched boxplots) or stem + branches + foliage (treatment S(WB)BF, solid boxplots) compared to conventional stem-only harvest (S(WB)). Theoretical values calculated using harvest rates of 100% ("nutrient stocks" dataset). Number of case studies = 214-323.

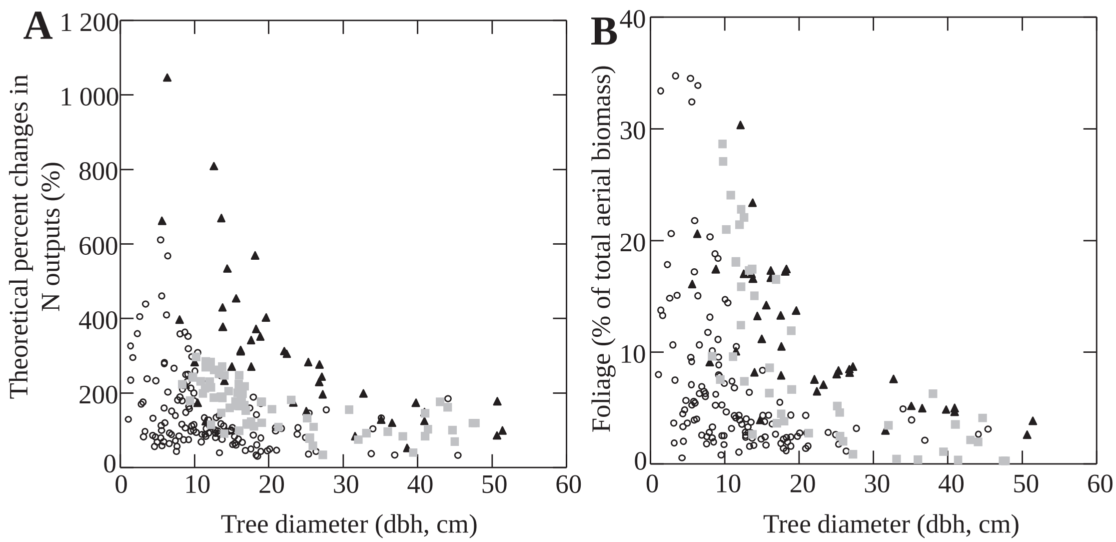

Figure S5 Theoretical percent changes in nutrient outputs as a result of removing stem + branches + foliage (treatment S(WB)BF) compared to conventional stem-only harvest (S(WB)); example for N, panel A) and contribution of foliage biomass to total aerial biomass (panel B): relationships with tree diameter. Theoretical values calculated using harvest rates of 100% (“nutrient stocks” dataset). Open circle, broadleaf trees; grey square, sparse canopy coniferous (mainly *Pinus*, also *Larix* or *Agathis*); black triangle, dense canopy coniferous (*Picea*, *Abies* and *Pseudotsuga*). For more details (relationships with tree age, tree height and stem biomass of all species) see Supplementary Figure S6.

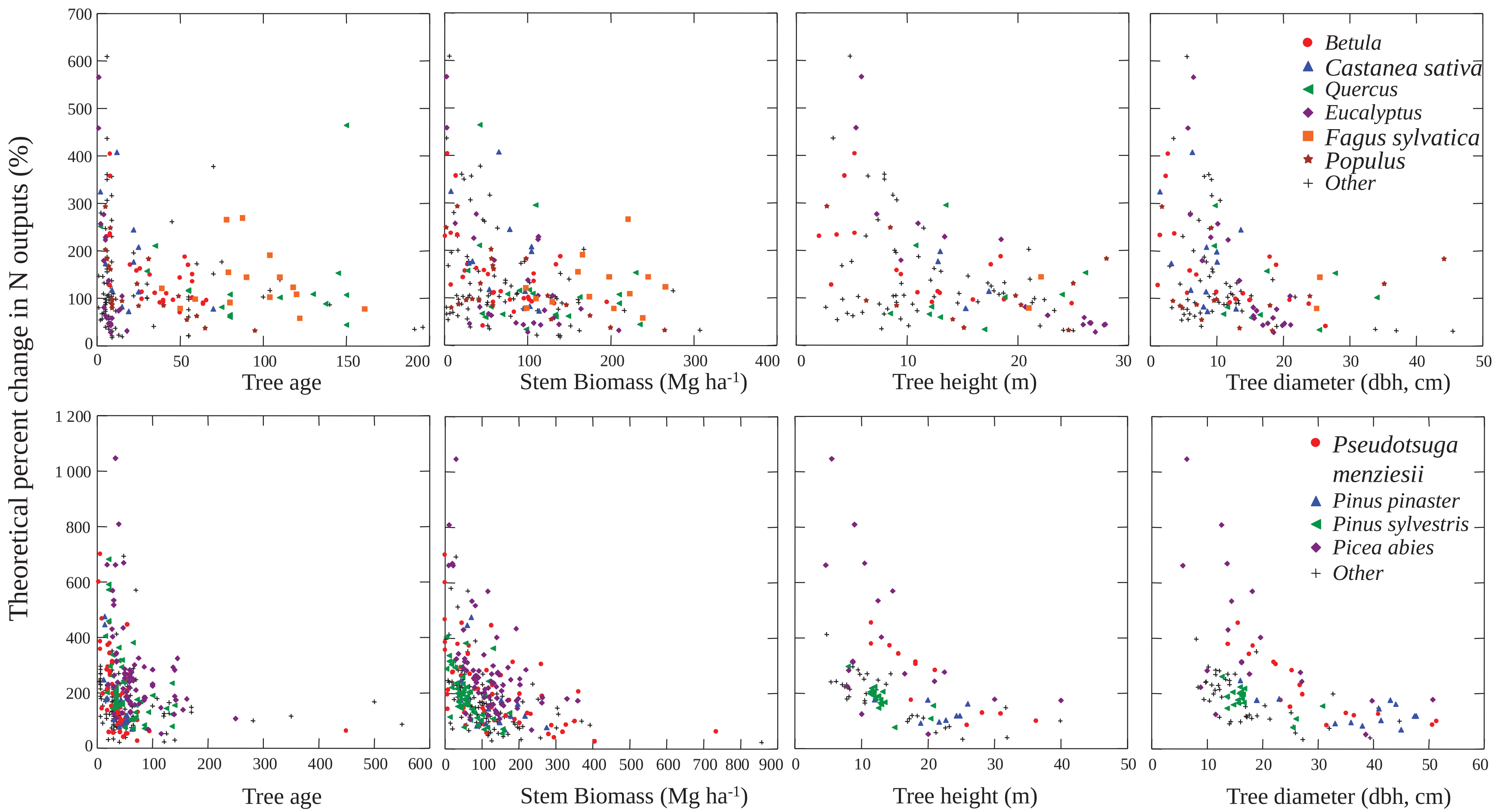

Figure S6 Theoretical percent change in nutrient outputs as a result of removing stem + branches + foliage (treatment S(WB)BF), compared to conventional stem-only harvesting (S(WB)) (example for N): relationships with tree age, stem biomass, tree height and tree diameter. Theoretical values calculated using harvest rates of 100% (“nutrient stocks” dataset). *Top*, broadleaf tree species; *Bottom*, coniferous tree species.

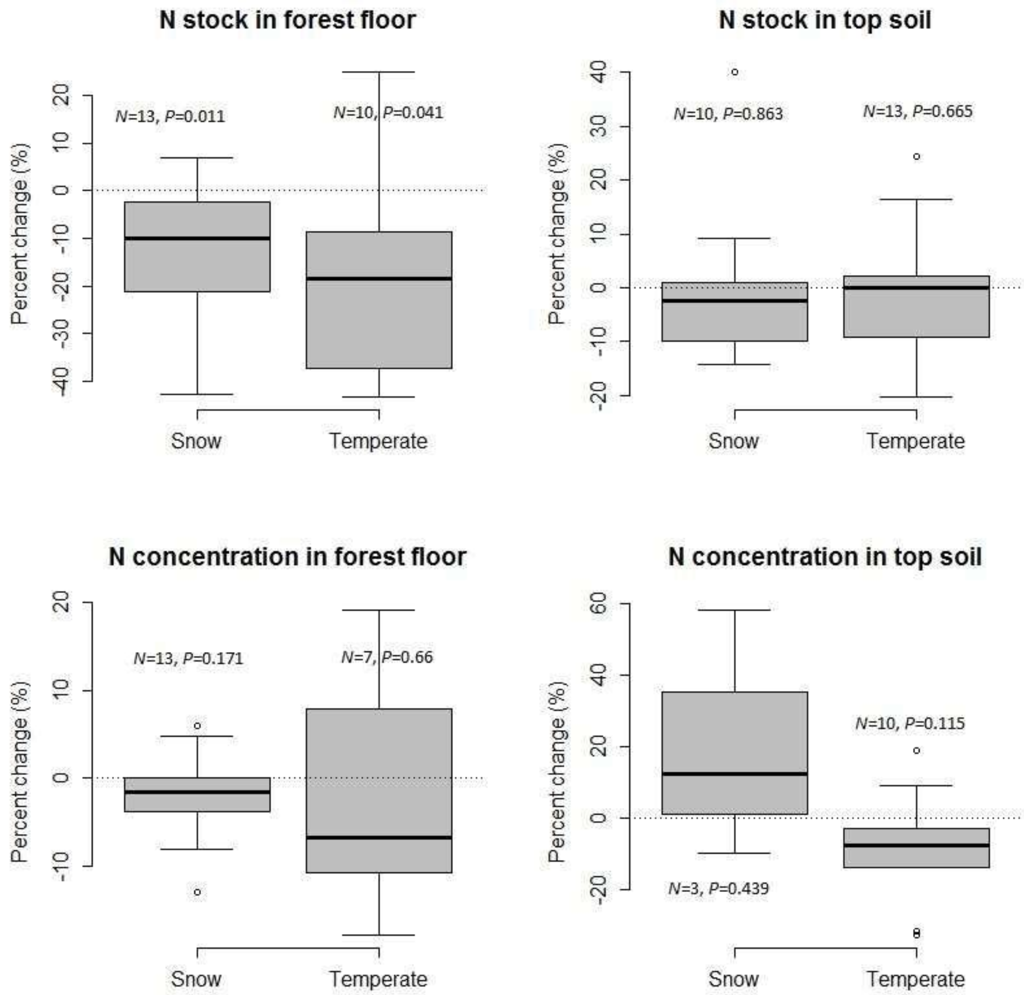

Figure S7 Effects of removing harvesting residues on total N in forest floor and top mineral soil as a function of the Koeppen climate classes ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). Number of case studies and *P* value (difference between the log ratio and value 0, one sample *t* test) are shown for each class (cold climate (snow Koeppen climate class) or temperate climate). There was generally no significant difference among classes (*P*=0.081 for N concentration in top soil, *P*=0.538-0.955 for other variables). Similar patterns were also observed for organic C.

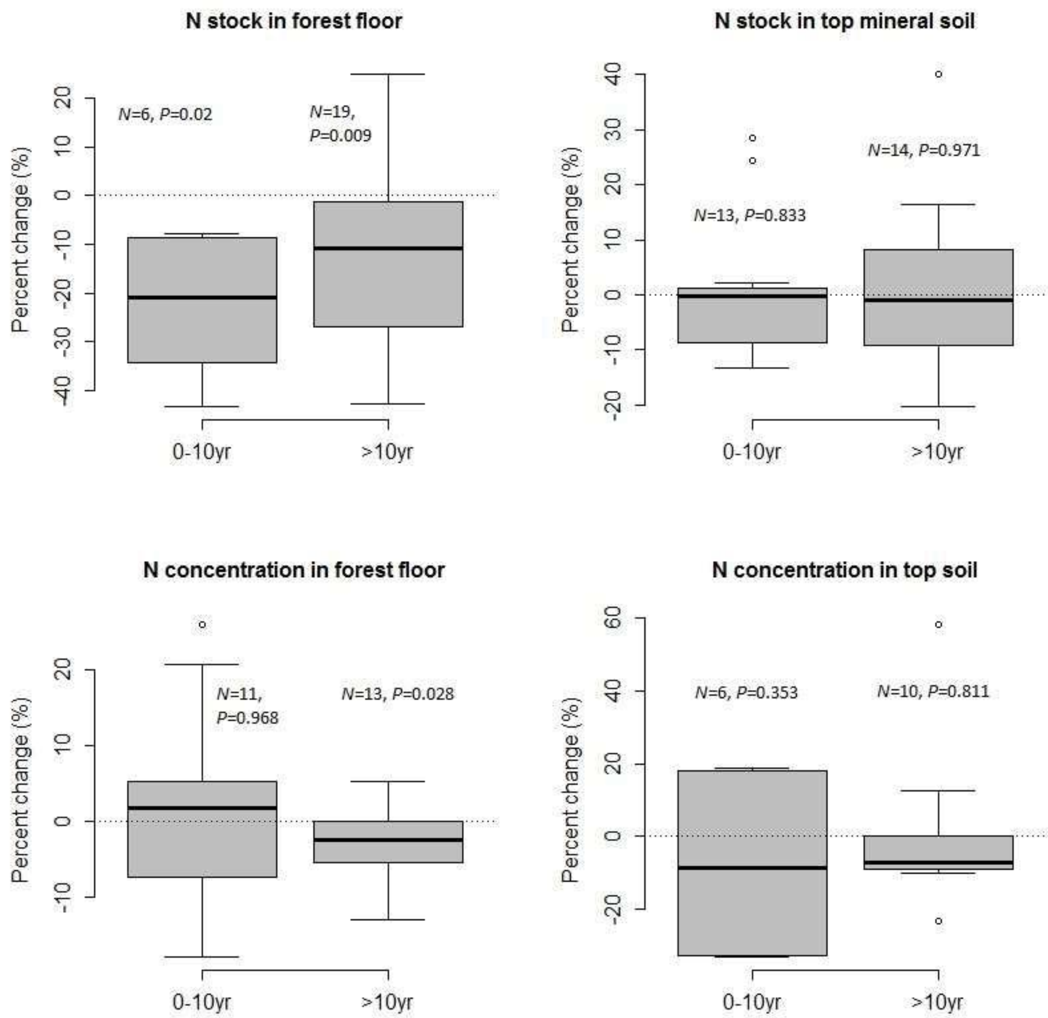

Figure S8 Effects of removing harvesting residues on total N in forest floor and top mineral soil as a function of the time elapsed since the removal ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). Number of case studies and *P* value (difference between the log ratio and value 0, one sample *t* test) are shown for each class. There was no significant difference among classes (*P*=0.290-0.909).

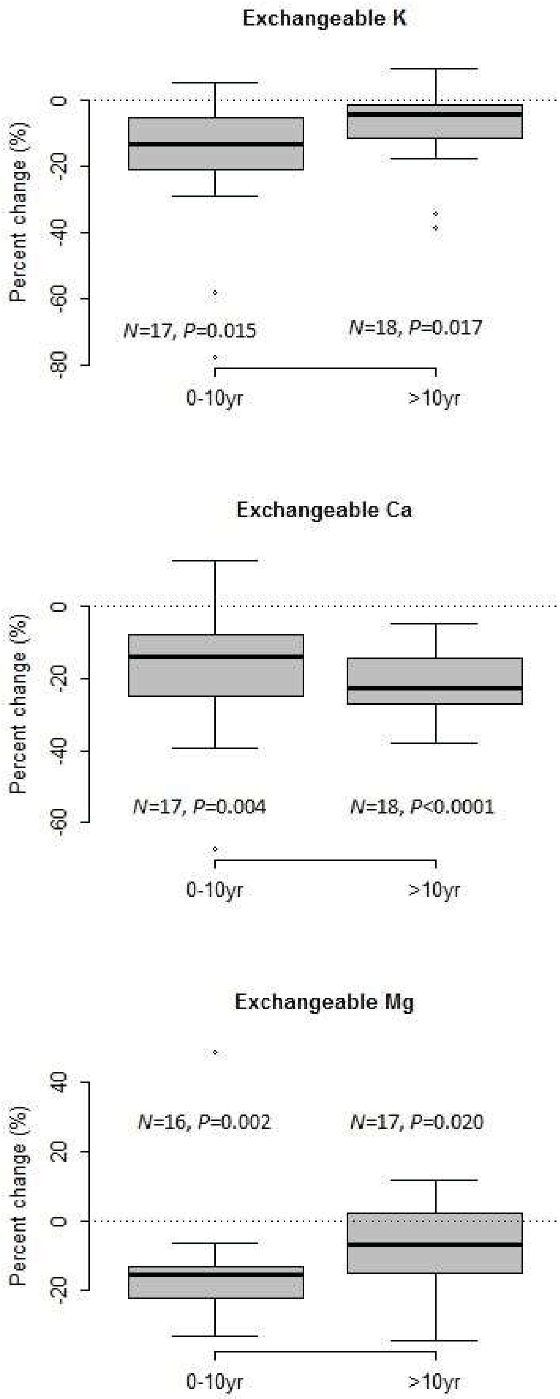

Figure S9 Effects of removing harvesting residues on exchangeable K, Ca and Mg in forest floor + top mineral soil as a function of the time elapsed since the removal ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). Number of case studies and *P* value (difference between the log ratio and value 0, one sample *t* test) are shown for each class. There was no significant difference among classes (*P*=0.115-0.648).

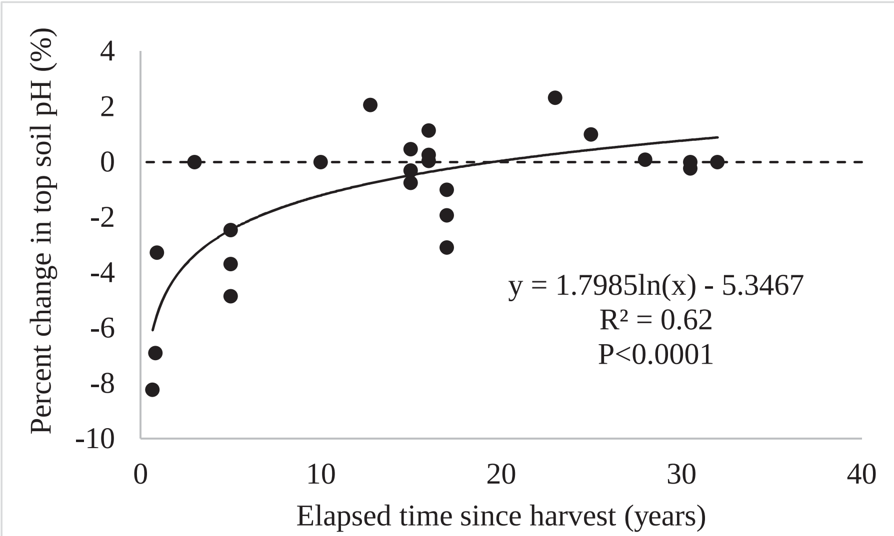

Figure S10 Effects of removing harvesting residues on pH in top mineral soil as a function of the time elapsed since the removal ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). There was significant relationships between percent change in soil pH and elapsed time (*P*<0.0001, non-linear regression).

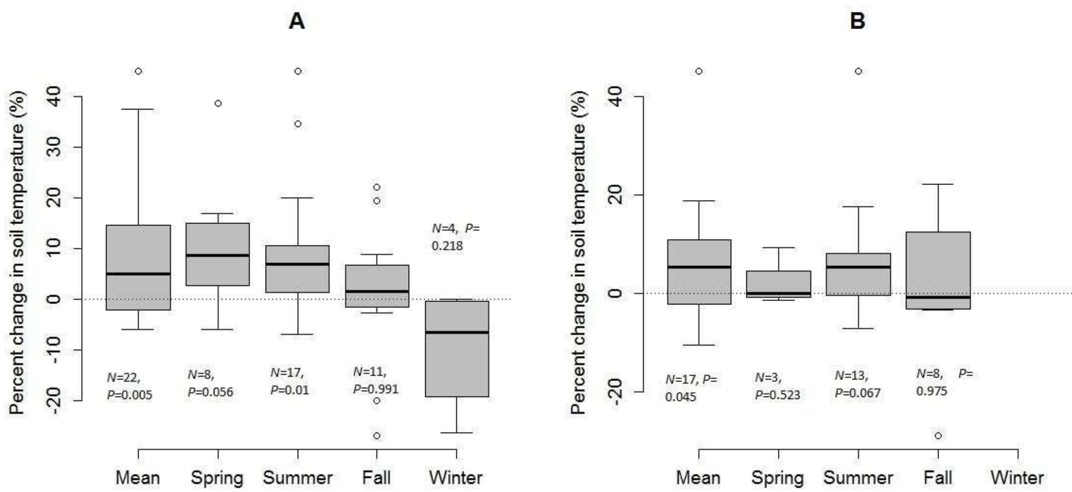

Figure S11 Effect of removing stem + harvesting residues on the temperature of the top mineral soil compared to stem-only harvest ("environmental impacts" dataset). Panel A includes all types of intensive harvests taken together. Panel B only includes removing of stem + branches + foliage (S(WB)BF)). There were significant effects on mean soil temperature and soil temperature in spring and summer (log ratio significantly higher than 0).

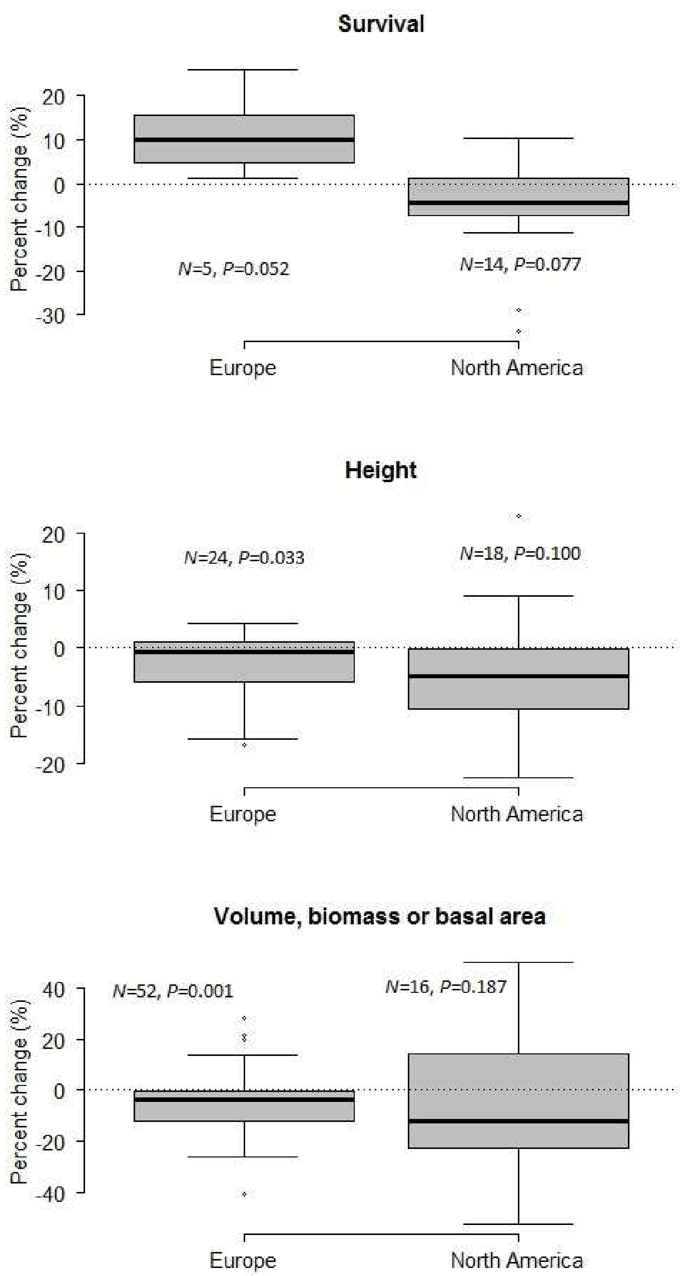

Figure S12 Effects of removing harvesting residues on tree growth: comparison between Europe and North America ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). Number of case studies and *P* value (difference between the log ratio and value 0, one sample *t* test) are shown for each class. There was significant difference among classes in tree survival (*P*=0.018) but not in tree growth (tree height, tree volume, biomass or basal area; *P*=0.466-0.605).

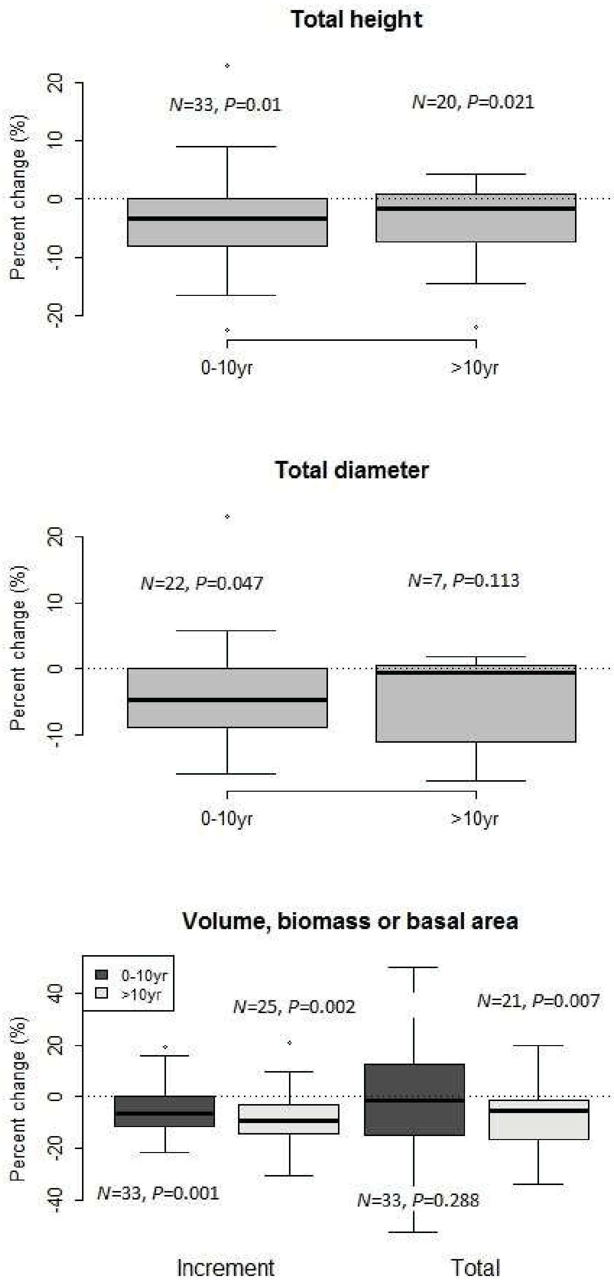

Figure S13 Effects of removing harvesting residues on tree growth as a function of the time elapsed since the removal ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). Number of case studies and *P* value (difference between the log ratio and value 0, one sample *t* test) are shown for each class. There was no significant difference among classes (*P*=0.242-0.961).

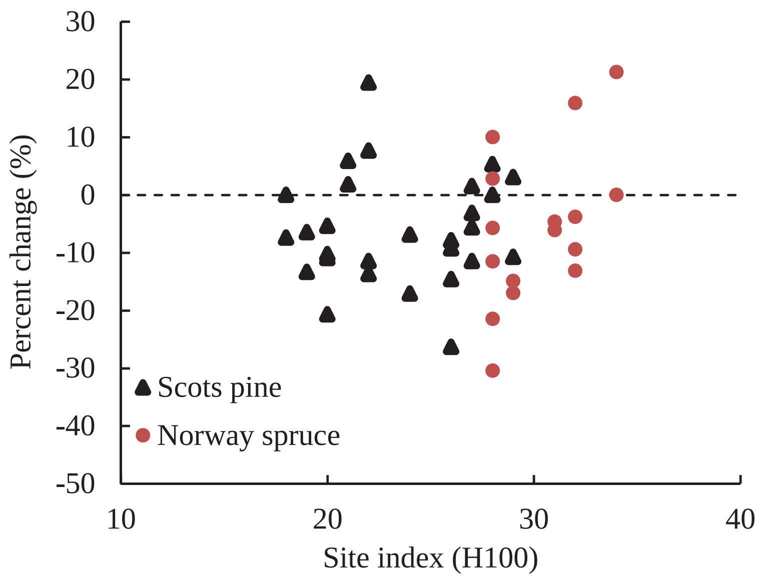

Figure S14 Effects of removing harvesting residues on tree growth (volume increment) in relation with the site index ("environmental impacts" dataset, treatment S(WB)BF compared to conventional stem-only harvest S(WB)). These data are from Helmisaari et al. 2011 (22 sites in Finland, Norway and Sweden). There are no relationship between percent change in tree growth and the site index (this figure or with data from other studies in Scandinavia (Tveite & Hanssen, 2013; Egnell & Ulvcrona, 2015)).

Table S1: Potential percent changes in biomass and nutrient outputs due to removing harvesting residues. Theoretical values calculated using harvest rates of 100% ("nutrient stocks" dataset).

	Harvest treatments compared to conventional stem-only harvest (wood + bark; S(WB))				
	S(W)	S(WB)B	S(WB)BF	S(WB)BR	S(WB)BFR
	Stem wood‡	Stem (wood + bark) + branches	Stem (wood + bark) + branches + foliage	Stem (wood + bark) + branches + stumps/roots	Stem (wood + bark) + branches + foliage + stumps/roots
Biomass					
(Number of case studies)	(113)	(240)	(314)	(98)	(101)
Mean, Median (Q1, Q3)	-14, -12 (-16, -10)	32, 24 (15, 35)	42, 30 (20, 46)	61, 54 (38, 67)	70, 62 (44, 76)
Min, Max	-73, -3	6, 813	7, 813	11, 363	23, 368
N					
(Number of case studies)	(122)	(254)	(323)	(98)	(104)
Mean, Median (Q1, Q3)	-37, -37 (-45, -28)	77, 56 (40, 90)	177, 150 (96, 228)	140, 126 (74, 168)	222, 192 (135, 281)
Min, Max	-85, -4	7, 482	18, 854	33, 692	54, 792
P					
(Number of case studies)	(113)	(231)	(290)	(88)	(92)
Mean, Median (Q1, Q3)	-36, -36 (-49, -23)	90, 62 (38, 104)	183, 152 (86, 238)	172, 122 (79, 211)	240, 200 (122, 309)
Min, Max	-78, -3	5, 856	12, 1244	23, 995	42, 1058
K					
(Number of case studies)	(116)	(235)	(293)	(93)	(96)
Mean, Median (Q1, Q3)	-33, -32 (-42, -23)	69, 48 (32, 77)	146, 119 (71, 176)	125, 105 (71, 158)	185, 159 (120, 216)
Min, Max	-72, -3	6, 495	15, 1378	24, 510	44, 566
Ca					
(Number of case studies)	(108)	(218)	(271)	(82)	(85)
Median (Q1, Q3)	-56, -56 (-69, -40)	64, 47 (31, 76)	122, 83 (51, 128)	113, 93 (62, 126)	147, 120 (73, 180)
Min, Max	-94, -16	6, 306	10, 1687	25, 770	38, 824
Mg					
(Number of case studies)	(107)	(214)	(222)	(79)	(82)
Mean, Median (Q1, Q3)	-38, -39 (-49, -26)	74, 51 (33, 86)	135, 99 (59, 169)	129, 109 (75, 150)	171, 143 (101, 227)
Min, Max	-80, -5	7, 438	12, 627	23, 456	33, 568
S					
(Number of case studies)	(17)	(42)	(35)	(15)	(16)
Mean, Median (Q1, Q3)	-37, -39 (-47, -28)	56, 44 (29, 61)	136, 75 (64, 131)	141, 99 (74, 108)	213, 124 (112, 161)
Min, Max	-58, -11	2, 230	12, 788	27, 758	33, 1317
Na					
(Number of case studies)	(7)	(17)	(17)	(9)	(9)
Mean, Median (Q1, Q3)	-24, -25 (-28, -21)	126, 37 (20, 109)	174, 87 (30, 141)	266, 248 (57, 351)	345, 258 (133, 424)
Min, Max	-33, -11	10, 811	10, 1091	25, 829	46, 1108
Fe					
(Number of case studies)	(12)	(19)	(19)	(4)	(4)
Mean, Median (Q1, Q3)	-28, -25 (-36, -20)	88, 39 (24, 75)	123, 55 (38, 140)	297, 185 (126, 355)	325, 234 (161, 399)
Min, Max	-64, -6	7, 672	10, 810	66, 752	78, 754
Mn					
(Number of case studies)	(21)	(28)	(27)	(8)	(8)
Mean, Median (Q1, Q3)	-32, -33 (-50, -19)	66, 38 (23, 55)	113, 73 (46, 114)	137, 73 (56, 171)	183, 119 (97, 210)
Min, Max	-58, -8	7, 433	19, 519	46, 431	73, 461
Zn					
(Number of case studies)	(11)	(11)	(11)	_	_
Mean, Median (Q1, Q3)	-36, -41 (-45, -29)	69, 42 (19, 119)	108, 63 (32, 190)	_	_
Min, Max	-58, -5	10, 164	12, 231	_	_
Cu					
(Number of case studies)	(10)	(10)	(10)	_	_
Mean, Median (Q1, Q3)	-18, -19 (-22, -14)	56, 45 (32, 74)	85, 72 (46, 118)	_	_
Min, Max	-27, -11	17, 137	22, 173	_	_
Ni					
(Number of case studies)	(4)	(4)	(4)	_	_
Mean, Median (Q1, Q3)	-10, -7 (-11, -6)	22, 25 (15, 32)	28, 32 (21, 40)	_	_
Min, Max	-20, -5	6, 33	8, 40	_	_
B					
(Number of case studies)	_#	(5)	(5)	(4)	(4)
Mean, Median (Q1, Q3)	_	94, 109 (78, 123)	153, 178 (105, 180)	152, 146 (139, 158)	206, 220 (190, 237)
Min, Max	_	24, 135	102, 201	121, 193	148, 237
Al					
(Number of case studies)	(3)	(6)	(6)	(3)	(3)
Mean, Median (Q1, Q3)	-44, -42 (-48, -40)	64, 66 (30, 97)	128, 147 (68, 189)	240, 290 (178, 327)	296, 360 (258, 365)
Min, Max	-54, -37	22, 101	27, 203	65, 365	156, 370

‡ Stem bark left on site (mitigation measures).

Not determined (number of case studies < 3).

Table S2: Percent changes in nutrient outputs due to removing harvesting residues. Theoretical values calculated using harvest rates# ("nutrient stocks" dataset).

	Harvest treatments compared to stem-only harvest (wood + bark, S(WB); logging with machine†)					
	S(WB)B	S(WB)BF		S(WB)BR	S(WB)BFR	
	Stem (wood + bark) + branches	Stem (wood + bark) + branches+foliage		Stem (wood + bark) + branches + stumps/roots	Stem (wood + bark) + branches + foliage + stumps/roots	
	Machine†	Machine	Machine, in winter or after drying step‡	Machine	Machine	Machine, in winter or after drying step
_N_						
(Number of case studies)	(112)	(109)	(109)	(35)	(35)	(35)
Mean, Median (Q1, Q3)	51, 41 (22, 69)	122, 95 (63, 139)	63, 52 (29, 74)	107, 73 (57, 126)	153, 127 (106, 142)	116, 93 (67, 127)
Min, Max	6, 325	11, 680	6, 414	22, 649	48, 684	29, 649
Potential Mean, Median¶	77, 56	177, 150	177, 150	140, 126	222, 192	222, 192
_P_						
(Number of case studies)	(105)	(102)	(102)	(34)	(34)	(34)
Mean, Median (Q1, Q3)	59, 43 (22, 70)	120, 90 (51, 158)	70, 52 (27, 87)	109, 72 (49, 152)	149, 128 (64, 191)	118, 86 (50, 156)
Min, Max	6, 642	12, 836	7, 642	19, 604	31, 627	20, 604
Potential Mean, Median	90, 62	183, 152	183, 152	172, 122	240, 200	240, 200
_K_						
(Number of case studies)	(107)	(104)	(104)	(36)	(36)	(36)
Mean, Median (Q1, Q3)	43, 35 (20, 54)	84, 70 (45, 107)	50, 40 (24, 62)	95, 79 (53, 120)	128, 104 (83, 144)	101, 85 (59, 121)
Min, Max	3, 183	11, 338	5, 206	19, 343	34, 365	20, 343
Potential Mean, Median	69, 48	146, 119	146, 119	125, 105	185, 159	185, 159
_Ca_						
(Number of case studies)	(101)	(98)	(98)	(36)	(36)	(36)
Median (Q1, Q3)	55, 42 (23, 73)	91, 70 (37, 121)	62, 50 (28, 87)	108, 75 (52, 129)	132, 97 (66, 158)	112, 78 (53, 140)
Min, Max	8, 310	10, 327	9, 310	18, 675	48, 693	30, 675
Potential Mean, Median	64, 47	122, 83	122, 83	113, 93	147, 120	147, 120
_Mg_						
(Number of case studies)	(100)	(97)	(97)	(36)	(36)	(36)
Mean, Median (Q1, Q3)	53, 37 (24, 67)	89, 74 (42, 118)	60, 44 (26, 77)	97, 84 (49, 137)	124, 112 (74, 177)	103, 87 (50, 145)
Min, Max	5, 310	11, 375	6, 310	22, 206	28, 249	22, 207
Potential Mean, Median	74, 51	135, 99	135, 99	129, 109	171, 143	171, 143

Harvest rates: 100% of stem wood, 20-80% of stem bark depending on harvest conditions (see below), 50-60% of branches depending on tree species, 0-40% of foliage depending on harvest conditions (see below), 60% of roots.

† Logging with a chainsaw causes low bark losses, 80% of the bark remains on the stem and is exported, while machine logging significantly increases bark losses, only 20% of the bark remains on the stem and is exported.

‡ Foliage exports are strongly reduced with mitigation measures (100% of foliage is left on site when broadleaf trees are removed in winter or when residues are removed after a delay that allows the foliage to dry and fall off the branches, 90% of foliage is left on site when residues of coniferous trees are removed after a delay that allows the foliage to dry and fall off the branches).

¶ Comparison with the theoretical values calculated using harvest rates of 100%.

Allometric relationships ("nutrient stocks" dataset)

In our meta-analysis, the effects of removing harvesting residues on nutrient outputs are expressed as percent changes relative to the conventional stem-only harvest (see results in the main text (**Table 1**) and Supplementary **Tables S1** and **S2** and **Figures S4** to **S6**).

We also used the "nutrient stocks" dataset to build allometric relationships that enable estimating theoretical nutrient outputs in kg ha^{-1} (considering 100% harvest rates) as a function of stem biomass. We built allometric relationships for macronutrients (N, P, K, Ca and Mg). These allometric relationships were fitted for several broadleaf (*Betula, Castanea sativa, Quercus, Eucalyptus, Fagus sylvatica*, and *Populus*) and needleleaf tree species (*Pseudotsuga menziesii, Picea abies, Pinus pinaster,* and *Pinus sylvestris*). These relationships make possible to quantify nutrient stocks in kg ha^{-1} in trees and hence theoretical nutrient outputs depending on stem biomass and harvest intensity. Relationships were generally robust with a high or moderate confidence index in most cases (based on R^2 values; see details in **Table S3**).

Allometric relationships are by-products of case study data compilation and were not used in the meta-analysis. We determined allometric relationships because they are useful tools to estimate for instance the amounts of fertilizers or wood ash to apply (i.e. in case of compensatory strategies).

Table S3 Relationships between nutrient stocks in trees (y, in kg ha^{-1}) and stem biomass (wood + bark; x, in Mg ha^{-1}) (theoretical values, 100% harvest rates; "nutrient stocks" dataset).

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Betula sp.									
N	S(W)	$y=ax^b$	3.8722	0.6376	9	0.86	<0.0001	10–110	High
N	S(WB)	$y=ax^b$	4.2674	0.7453	29	0.92	<0.0001	10–150	High
N	S(WB)B	$y=ax^b$	8.2710	0.6973	23	0.86	<0.0001	10–150	High
N	S(WB)BF	$y=ax^b$	17.1810	0.6134	30	0.78	<0.0001	10–150	High
N	S(WB)BFR	$y=ax^b$	10.5250	0.8228	7	0.98	<0.0001	50–140	High
P	S(W)	$y=ax^b$	0.8814	0.4476	9	0.47	0.041	10–110	Moderate
P	S(WB)	$y=ax^b$	0.6590	0.6217	28	0.72	<0.0001	10–150	High
P	S(WB)B	$y=ax^b$	1.2662	0.5686	22	0.71	<0.0001	10–150	High
P	S(WB)BF	$y=ax^b$	2.3967	0.5102	29	0.57	<0.0001	10–150	High
P	S(WB)BFR	$y=ax^b$	1.1651	0.7689	7	0.91	0.001	50–140	High
K	S(W)	$y=ax^b$	2.0339	0.6404	9	0.74	0.003	10–110	High
K	S(WB)	$y=ax^b$	2.2351	0.7061	28	0.85	<0.0001	10–150	High
K	S(WB)B	$y=ax^b$	4.2162	0.6391	22	0.85	<0.0001	10–150	High
K	S(WB)BF	$y=ax^b$	7.2588	0.6126	29	0.78	<0.0001	10–150	High
K	S(WB)BFR	$y=ax^b$	4.9815	0.7600	7	0.98	<0.0001	50–140	High
Ca	S(W)	nd	*mean = 49 kg ha^{-1}*		4	nd	nd	60–110	nd
Ca	S(WB)	$y=ax^b$	2.9600	0.8545	23	0.87	<0.0001	20–150	High
Ca	S(WB)B	$y=ax^b$	6.3915	0.7800	17	0.87	<0.0001	20–150	High
Ca	S(WB)BF	$y=ax^b$	6.6964	0.7860	24	0.88	<0.0001	20–150	High
Ca	S(WB)BFR	$y=ax^b$	10.0840	0.8093	7	0.98	<0.0001	50–140	High
Mg	S(W)	nd	*mean = 16 kg ha^{-1}*		4	nd	nd	60–110	nd
Mg	S(WB)	$y=ax^b$	0.3196	0.9152	16	0.97	<0.0001	30–140	High
Mg	S(WB)B	$y=ax^b$	0.4873	0.8996	17	0.96	<0.0001	30–140	High
Mg	S(WB)BF	$y=ax^b$	0.7957	0.8468	17	0.92	<0.0001	30–140	High
Mg	S(WB)BFR	$y=ax^b$	1.0256	0.9008	7	0.95	<0.0001	50–140	High
Castanea sativa									
N	S(W)	$y=ax^b$	4.6356	0.6232	5	1.00	<0.0001	10–120	High
N	S(WB)	$y=ax^b$	5.2603	0.6345	17	0.44	0.004	10–140	Moderate
N	S(WB)B	$y=ax^b$	9.7291	0.6441	17	0.66	<0.0001	10–140	High
N	S(WB)BF	$y=ax^b$	52.4600	0.3397	11	0.47	0.019	10–120	Moderate
N	S(WB)BFR	nd	nd	nd	0	nd	nd	nd	nd
P	S(W)	$y=ax^b$	*similar to S(WB)*		5	0.99	<0.0001	10–120	High
P	S(WB)	$y=ax^b$	0.6838	0.4729	15	0.37	0.016	10–140	Moderate
P	S(WB)B	$y=ax^b$	1.3065	0.5010	15	0.69	<0.0001	10–140	High
P	S(WB)BF	$y=ax^b$	3.4282	0.4178	9	0.87	<0.0001	10–120	High
P	S(WB)BFR	nd	nd	nd	0	nd	nd	nd	nd
K	S(W)	$y=ax^b$	*similar to S(WB)*		5	0.99	<0.0001	10–120	High
K	S(WB)	$y=ax^b$	3.3794	0.5050	17	0.29	0.025	10–140	Moderate
K	S(WB)B	$y=ax^b$	6.4518	0.5614	17	0.70	<0.0001	10–140	High
K	S(WB)BF	$y=ax^b$	20.9460	0.3746	11	0.67	0.002	10–120	High
K	S(WB)BFR	nd	nd	nd	0	nd	nd	nd	nd
Ca	S(W)	$y=ax^b$	1.7350	0.6186	5	0.99	<0.0001	10–120	High
Ca	S(WB)	$y=ax^b$	8.4702	0.6423	17	0.49	0.002	10–140	Moderate
Ca	S(WB)B	$y=ax^b$	12.5500	0.6684	17	0.68	<0.0001	10–140	High
Ca	S(WB)BF	$y=ax^b$	28.3490	0.5202	11	0.59	0.006	10–120	High
Ca	S(WB)BFR	nd	nd	nd	0	nd	nd	nd	nd
Mg	S(W)	$y=ax^b$	0.8231	0.6206	5	0.99	<0.0001	10–120	High
Mg	S(WB)	$y=ax^b$	1.3925	0.6583	17	0.71	<0.0001	10–140	High
Mg	S(WB)B	$y=ax^b$	1.9152	0.7229	17	0.73	<0.0001	10–140	High
Mg	S(WB)BF	$y=ax^b$	9.0851	0.4084	11	0.56	0.008	10–120	High
Mg	S(WB)BFR	nd	nd	nd	0	nd	nd	nd	nd

nd, not determined due to the small number of case studies.

† Range of stem biomass in which models can be used to estimate nutrient stocks.

‡ Confidence index based on R^2 values (*low* : R^2=0.00-0.25; *moderate* : R^2=0.25-0.50; *high* : R^2=0.50-1.00).

Table S3 Continued.

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Quercus sp.									
N	S(W)	$y=ax^b$	1.4867	1.0000	6	0.91	<0.0001	30–210	High
N	S(WB)	$y=ax^b$	6.7645	0.7573	27	0.85	<0.0001	10–240	High
N	S(WB)B	$y=ax^b$	4.1451	0.9689	40	0.94	<0.0001	10–300	High
N	S(WB)BF¶	$y=ax^b$	14.5020	0.7346	35	0.92	<0.0001	10–150	High
N	S(WB)BFR	$y=ax^b$	33.5290	0.6202	13	0.30	0.052	70–300	Moderate
P	S(W)	$y=ax^b$	0.4419	0.6242	7	0.35	0.162	30–470	Moderate
P	S(WB)	$y=ax^b$	0.5721	0.6974	26	0.63	<0.0001	10–470	High
P	S(WB)B	$y=ax^b$	0.9695	0.7470	30	0.78	<0.0001	10–470	High
P	S(WB)BF#	$y=ax^b$	2.4499	0.5849	30	0.86	<0.0001	10–200	High
P	S(WB)BFR	$y=ax^b$	0.5419	0.9450	11	0.35	0.054	70–300	Moderate
K	S(W)	$y=ax^b$	1.1612	0.9458	7	0.96	<0.0001	30–470	High
K	S(WB)	$y=ax^b$	3.8190	0.7659	27	0.88	<0.0001	10–470	High
K	S(WB)B	$y=ax^b$	5.6453	0.7668	30	0.89	<0.0001	10–470	High
K	S(WB)BF	$y=ax^b$	8.2330	0.7214	30	0.95	<0.0001	10–240	High
K	S(WB)BFR	$y=ax^b$	32.6460	0.4921	11	0.14	0.259	70–240	Low
Ca	S(W)	$y=ax^b$	6.8519	0.4535	7	0.23	0.275	30–470	Low
Ca	S(WB)	$y=ax^b$	12.4870	0.7606	22	0.30	0.009	10–470	Moderate
Ca	S(WB)B	$y=ax^b$	18.3890	0.7872	30	0.68	<0.0001	10–470	High
Ca	S(WB)BF	$y=ax^b$	16.4410	0.8489	30	0.88	<0.0001	10–300	High
Ca	S(WB)BFR	$y=ax^b$	38.7160	0.6977	11	0.46	0.022	70–240	Moderate
Mg	S(W)	$y=ax^b$	0.1470	0.8821	7	0.56	0.053	30–470	High
Mg	S(WB)	$y=ax^b$	0.4068	0.8785	21	0.45	0.001	30–470	Moderate
Mg	S(WB)B	$y=ax^b$	1.0088	0.8108	28	0.84	<0.0001	10–470	High
Mg	S(WB)BF	$y=ax^b$	2.2021	0.6919	28	0.92	<0.0001	10–240	High
Mg	S(WB)BFR	$y=ax^b$	1.7033	0.8106	9	0.46	0.044	70–240	Moderate

¶ Other model: $y= -0.0039x^2+4.6999x+25.538$ (R^2=0.79; range of stem biomass = 10-300 Mg ha^{-1})

Other model: $y=0.2493x+10.358$ (R^2 = 0.62; range of stem biomass = 50-300 Mg ha^{-1})

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Eucalyptus									
N	S(W)	$y=ax^b$	1.8298	0.9121	24 (22)†	0.90 (0.77)§	<0.0001 (<0.0001)§	5–170	High
N	S(WB)	$y=ax^b$	3.7891	0.7834	27 (25)	0.84 (0.65)	<0.0001 (<0.0001)	5–170	High
N	S(WB)B	$y=ax^b$	6.1370	0.7536	23 (21)	0.92 (0.83)	<0.0001 (<0.0001)	5–170	High
N	S(WB)BF	$y=ax^b$	27.8850	0.5003	28 (26)	0.77 (0.66)	<0.0001 (<0.0001)	5–170	High
N	S(WB)BFR	nd	nd	nd	2	nd	nd	nd	nd
P	S(W)	$y=ax^b$	0.5431	0.7331	24 (22)	0.63 (0.30)	<0.0001 (0.009)	5–140	Moderate
P	S(WB)	$y=ax^b$	1.3348	0.6014	26 (24)	0.56 (0.22)	<0.0001 (0.02)	5–140	Low
P	S(WB)B	$y=ax^b$	2.2210	0.5675	23 (21)	0.66 (0.37)	<0.0001 (0.003)	5–140	Moderate
P	S(WB)BF	$y=ax^b$	3.3855	0.5076	28 (26)	0.65 (0.54)	<0.0001 (<0.0001)	5–140	High
P	S(WB)BFR	nd	nd	nd	2	nd	nd	nd	nd
K	S(W)	$y=ax^b$	3.2245	0.6591	19 (18)	0.48 (0.08)	0.001 (0.271)	5–210	Low
K	S(WB)	$y=ax^b$	7.4630	0.5364	22 (21)	0.35 (0.04)	0.004 (0.292)	5–210	Low
K	S(WB)B	$y=ax^b$	10.2720	0.5566	18 (17)	0.43 (0.07)	0.003 (0.317)	5–210	Low
K	S(WB)BF	$y=ax^b$	17.8990	0.4847	23 (22)	0.42 (0.19)	0.001 (0.043)	5–210	Low
K	S(WB)BFR	nd	nd	nd	2	nd	nd	nd	nd
Ca	S(W)	$y=ax^b$	0.3725	1.0220	15 (14)	0.68 (0.21)	<0.0001 (0.096)	5–210	Low
Ca	S(WB)	$y=ax^b$	3.8803	0.7489	18 (17)	0.39 (0.11)	0.006 (0.192)	5–210	Low
Ca	S(WB)B	$y=ax^b$	6.6855	0.7293	14 (13)	0.38 (0.05)	0.018 (0.481)	5–210	Low
Ca	S(WB)BF	$y=ax^b$	55.7860	0.3635	19 (18)	0.12 (0.01)	0.150 (0.724)	5–210	Low
Ca	S(WB)BFR	nd	nd	nd	2	nd	nd	nd	nd
Mg	S(W)	$y=ax^b$	0.8946	0.6483	20 (18)	0.88 (0.72)	<0.0001 (<0.0001)	5–210	High
Mg	S(WB)	$y=ax^b$	4.0630	0.4553	23 (21)	0.51 (0.21)	<0.0001 (0.038)	5–210	Low
Mg	S(WB)B	$y=ax^b$	5.2454	0.5086	19 (17)	0.83 (0.58)	<0.0001 (<0.0001)	5–210	High
Mg	S(WB)BF	$y=ax^b$	14.8410	0.3374	24 (22)	0.62 (0.28)	<0.0001 (0.011)	5–210	Moderate
Mg	S(WB)BFR		nd	nd	2	nd	nd	nd	nd

§ In some cases (particularly for K and Ca), relationships were not significant when a case study with low stem biomass values was deleted (range of stem biomass = 2.4-2.8 Mg ha^{-1}). Results (R^2 and P) without this case study are shown in brackets.

nd, not determined due to the small number of case studies.

† Range of stem biomass in which models can be used to estimate nutrient stocks.

‡ Confidence index based on R^2 values (*low* : R^2=0.00-0.25; *moderate* : R^2=0.25-0.50; *high* : R^2=0.50-1.00).

Table S3 Continued.

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Fagus sylvatica									
N	S(W)	nd	nd	nd	1	nd	nd	nd	nd
N	S(WB)	$y=ax^b$	2.5034	0.8845	18	0.69	<0.0001	100–360	High
N	S(WB)B	$y=ax^b$	7.7513	0.7764	12	0.36	0.039	100–260	Moderate
N	S(WB)BF	$y=ax^b$	5.1332	0.9045	15	0.48	0.004	100–260	Moderate
N	S(WB)BFR	$y=ax^b$	8.8325	0.8219	9	0.34	0.097	110–260	Moderate
P	S(W)	nd	nd	nd	2	nd	nd	nd	nd
P	S(WB)	$y=ax^b$	0.5453	0.7145	17	0.49	0.002	100–650	Moderate
P	S(WB)B	$y=ax^b$	0.8057	0.7611	11	0.67	0.002	110–650	High
P	S(WB)BF	$y=ax^b$	4.5732	0.4654	13	0.23	0.096	100–260	Low
P	S(WB)BFR	$y=ax^b$	3.5641	0.5675	8	0.29	0.172	110–260	Moderate
K	S(W)	nd	nd	nd	2	nd	nd	nd	nd
K	S(WB)	$y=ax^b$	0.6568	1.0887	18	0.86	<0.0001	100–650	High
K	S(WB)B	$y=ax^b$	0.4877	1.2105	12	0.87	<0.0001	110–650	High
K	S(WB)BF	$y=ax^b$	1.0530	1.0935	14	0.77	<0.0001	100–260	High
K	S(WB)BFR	$y=ax^b$	0.5770	1.2475	9	0.71	0.005	110–260	High
Ca	S(W)	nd	nd	nd	2	nd	nd	nd	nd
Ca	S(WB)	$y=ax^b$	0.7019	1.1351	18	0.70	<0.0001	100–650	High
Ca	S(WB)B	$y=ax^b$	1.8013	1.0550	12	0.68	0.001	110–650	High
Ca	S(WB)BF	$y=ax^b$	0.6456	1.2734	14	0.61	0.001	100–260	High
Ca	S(WB)BFR	$y=ax^b$	2.3965	1.0738	9	0.38	0.077	110–260	Moderate
Mg	S(W)	nd	nd	nd	2	nd	nd	nd	nd
Mg	S(WB)	$y=ax^b$	0.2421	0.9792	18	0.75	<0.0001	100–650	High
Mg	S(WB)B	$y=ax^b$	0.0900	1.2178	12	0.83	<0.0001	110–650	High
Mg	S(WB)BF	$y=ax^b$	0.1546	1.1489	14	0.55	0.002	100–260	High
Mg	S(WB)BFR	$y=ax^b$	0.0196	1.5935	9	0.69	0.006	110–260	High
Populus sp.									
N	S(W)	$y=ax^b$	0.8383	1.0585	15	0.90	<0.0001	5–260	High
N	S(WB)	$y=ax^b$	1.8012	1.0000	33	0.86	<0.0001	5–260	High
N	S(WB)B	$y=ax^b$	5.8306	0.8019	34	0.73	<0.0001	5–260	High
N	S(WB)BF	$y=ax^b$	5.7802	0.8323	22	0.80	<0.0001	5–260	High
N	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
P	S(W)	$y=ax^b$	0.1681	0.9516	15	0.79	<0.0001	5–260	High
P	S(WB)	$y=ax^b$	0.3127	0.9945	28	0.79	<0.0001	5–260	High
P	S(WB)B	$y=ax^b$	0.6342	0.9359	30	0.74	<0.0001	40–260	High
P	S(WB)BF	$y=ax^b$	1.1822	0.8292	21	0.71	<0.0001	40–260	High
P	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
K	S(W)	$y=ax^b$	0.6219	1.1115	15	0.93	<0.0001	5–260	High
K	S(WB)	$y=ax^b$	1.2996	1.0130	28	0.85	<0.0001	5–260	High
K	S(WB)B	$y=ax^b$	2.5824	0.9215	30	0.83	<0.0001	5–260	High
K	S(WB)BF	$y=ax^b$	3.4357	0.8685	21	0.89	<0.0001	5–260	High
K	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
Ca	S(W)	$y=ax^b$	0.6182	1.2921	15	0.87	<0.0001	5–260	High
Ca	S(WB)	$y=ax^b$	2.1540	1.1643	24	0.90	<0.0001	5–260	High
Ca	S(WB)B	$y=ax^b$	3.8769	1.1089	26	0.86	<0.0001	5–260	High
Ca	S(WB)BF	$y=ax^b$	3.9373	1.0937	17	0.91	<0.0001	5–260	High
Ca	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
Mg	S(W)	$y=ax^b$	0.2114	1.1068	15	0.87	<0.0001	5–260	High
Mg	S(WB)	$y=ax^b$	0.5467	0.9445	24	0.79	<0.0001	5–260	High
Mg	S(WB)B	$y=ax^b$	1.0309	0.8857	26	0.85	<0.0001	5–260	High
Mg	S(WB)BF	$y=ax^b$	1.4646	0.8606	17	0.90	<0.0001	5–260	High
Mg	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd

nd, not determined due to the small number of case studies.

† Range of stem biomass in which models can be used to estimate nutrient stocks.

‡ Confidence index based on R^2 values (*low* : R^2=0.00-0.25; *moderate* : R^2=0.25-0.50; *high* : R^2=0.50-1.00).

Table S3 Continued.

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Pseudotsuga menziesii									
N	S(W)	$y=ax^b$	1.4397	0.7888	28	0.66	<0.0001	10–360	High
N	S(WB)	$y=ax^b$	2.2392	0.8170	54	0.84	<0.0001	10–360	High
N	S(WB)B	$y=ax^b$	4.4195	0.7496	42	0.83	<0.0001	10–360	High
N	S(WB)BF	$y=ax^b$	10.8690	0.6906	53	0.86	<0.0001	10–360	High
N	S(WB)BFR	$y=ax^b$	11.2680	0.7008	5	1.00	<0.0001	10–260	High
P	S(W)	$y=ax^b$	0.3570	0.5664	26	0.47	<0.0001	10–360	Moderate
P	S(WB)	$y=ax^b$	0.4128	0.7577	51	0.84	<0.0001	10–360	High
P	S(WB)B	$y=ax^b$	0.8010	0.6858	39	0.90	<0.0001	10–360	High
P	S(WB)BF	$y=ax^b$	2.5802	0.5932	52	0.75	<0.0001	10–360	High
P	S(WB)BFR	nd	nd	nd	4	nd	nd	nd	nd
K	S(W)	$y=ax^b$	2.0696	0.6028	28	0.45	<0.0001	10–360	Moderate
K	S(WB)	$y=ax^b$	2.1536	0.7027	55	0.57	<0.0001	10–360	High
K	S(WB)B	$y=ax^b$	4.0643	0.6623	41	0.66	<0.0001	10–360	High
K	S(WB)BF	$y=ax^b$	8.3087	0.6229	54	0.81	<0.0001	10–360	High
K	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
Ca	S(W)	$y=ax^b$	0.9292	0.7656	28	0.69	<0.0001	10–360	High
Ca	S(WB)	$y=ax^b$	1.5043	0.8102	54	0.75	<0.0001	10–360	High
Ca	S(WB)B	$y=ax^b$	3.4356	0.7773	41	0.81	<0.0001	10–360	High
Ca	S(WB)BF	$y=ax^b$	6.8160	0.7315	53	0.85	<0.0001	10–360	High
Ca	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
Mg	S(W)	$y=ax^b$	0.2854	0.6313	28	0.43	<0.0001	10–360	Moderate
Mg	S(WB)	$y=ax^b$	0.2991	0.7601	52	0.72	<0.0001	10–360	High
Mg	S(WB)B	$y=ax^b$	0.6224	0.7096	41	0.77	<0.0001	10–360	High
Mg	S(WB)BF	$y=ax^b$	1.4035	0.6696	51	0.82	<0.0001	10–360	High
Mg	S(WB)BFR	nd	nd	nd	5	nd	nd	nd	nd
Picea abies									
N	S(W)	$y=ax^b$	1.2060	0.8717	13	0.46	0.011	20–360	Moderate
N	S(WB)	$y=ax^b$	1.7360	0.9086	70	0.76	<0.0001	10–360	High
N	S(WB)B	$y=ax^b$	9.3033	0.7139	39	0.73	<0.0001	10–360	High
N	S(WB)BF	$y=ax^b$	28.1650	0.5772	70	0.62	<0.0001	10–360	High
N	S(WB)BFR	$y=ax^b$	134.1900	0.3023	18	0.25	0.035	50–200	Low
P	S(W)	$y=ax^b$	0.1732	0.8067	13	0.37	0.027	20–360	Moderate
P	S(WB)	$y=ax^b$	0.2860	0.7971	63	0.54	<0.0001	10–360	High
P	S(WB)B	$y=ax^b$	0.7614	0.7634	35	0.68	<0.0001	10–360	High
P	S(WB)BF	$y=ax^b$	3.1991	0.5609	63	0.51	<0.0001	10–360	High
P	S(WB)BFR	$y=ax^b$	12.5660	0.3388	13	0.18	0.145	50–360	Low
K	S(W)	$y=ax^b$	0.4279	1.0000	13	0.75	<0.0001	20–360	High
K	S(WB)	$y=ax^b$	0.6849	0.9959	63	0.72	<0.0001	10–360	High
K	S(WB)B	$y=ax^b$	1.3383	0.9834	37	0.85	<0.0001	10–360	High
K	S(WB)BF	$y=ax^b$	7.8444	0.6854	63	0.67	<0.0001	10–360	High
K	S(WB)BFR	$y=ax^b$	27.2920	0.4917	13	0.52	0.006	50–360	High
Ca	S(W)	$y=ax^b$	1.2558	0.8688	13	0.86	<0.0001	20–360	High
Ca	S(WB)	$y=ax^b$	2.7402	0.8520	62	0.83	<0.0001	10–360	High
Ca	S(WB)B	$y=ax^b$	9.2711	0.6798	37	0.84	<0.0001	10–360	High
Ca	S(WB)BF	$y=ax^b$	28.1520	0.5172	62	0.62	<0.0001	10–360	High
Ca	S(WB)BFR	$y=ax^b$	43.9320	0.4775	13	0.58	0.003	50–360	High
Mg	S(W)	$y=ax^b$	0.1164	1.0000	13	0.96	<0.0001	20–360	High
Mg	S(WB)	$y=ax^b$	0.2381	0.9226	44	0.90	<0.0001	10–360	High
Mg	S(WB)B	$y=ax^b$	1.1894	0.6877	37	0.83	<0.0001	10–260	High
Mg	S(WB)BF	$y=ax^b$	3.0129	0.5675	44	0.66	<0.0001	10–260	High
Mg	S(WB)BFR	$y=ax^b$	3.3890	0.5806	13	0.60	0.002	50–260	High

nd, not determined due to the small number of case studies.

† Range of stem biomass in which models can be used to estimate nutrient stocks.

‡ Confidence index based on R^2 values (*low* : R^2=0.00-0.25; *moderate* : R^2=0.25-0.50; *high* : R^2=0.50-1.00).

Table S3 Continued.

Nutrient	Components	Model	Parameter a	Parameter b	N_{obs}	R^2	P	Biomass range†	Confidence index‡
Pinus pinaster									
N	S(W)	y=ax+b	0.5419	-5.5503	12	0.87	<0.0001	40–220	High
N	S(WB)	y=ax+b	0.6162	26.9980	16	0.69	<0.0001	40–220	High
N	S(WB)B	y=ax+b	1.5237	-2.9646	11	0.96	<0.0001	40–220	High
N	S(WB)BF	y=ax+b	0.8728	137.4000	15	0.62	0.001	90–160	High
N	S(WB)BFR	y=ax+b	1.5418	113.9400	11	0.89	<0.0001	40–220	High
P	S(W)	y=ax+b	0.0841	-1.0829	11	0.73	0.001	40–220	High
P	S(WB)	y=ax+b	0.0900	0.3606	15	0.86	<0.0001	40–220	High
P	S(WB)B	y=ax+b	0.1302	0.6000	11	0.95	<0.0001	40–220	High
P	S(WB)BF	y=ax+b	0.0813	10.9030	15	0.72	<0.0001	90–160	High
P	S(WB)BFR	y=ax+b	0.1260	9.9355	11	0.88	<0.0001	40–220	High
K	S(W)	y=ax+b	0.5069	-4.2406	11	0.90	<0.0001	40–220	High
K	S(WB)	y=ax+b	0.5503	11.7300	15	0.79	<0.0001	40–220	High
K	S(WB)B	y=ax+b	0.7436	10.7420	11	0.80	<0.0001	40–220	High
K	S(WB)BF	y=ax+b	0.4401	77.2670	15	0.34	0.022	90–160	Moderate
K	S(WB)BFR	y=ax+b	1.1011	58.7790	11	0.95	<0.0001	40–220	High
Ca	S(W)	y=ax+b	0.5074	-3.8934	11	0.94	<0.0001	40–220	High
Ca	S(WB)	y=ax+b	0.6020	18.3670	15	0.94	<0.0001	40–220	High
Ca	S(WB)B	y=ax+b	0.3835	82.6030	11	0.30	0.084	40–220	Moderate
Ca	S(WB)BF	y=ax+b	0.4793	78.2380	13	0.40	0.02	90–220	Moderate
Ca	S(WB)BFR	y=ax+b	0.7715	79.1760	11	0.57	0.007	40–220	High
Mg	S(W)	y=ax+b	0.2298	-0.8903	11	0.85	<0.0001	40–220	High
Mg	S(WB)	y=ax+b	0.2547	4.7620	15	0.83	<0.0001	40–220	High
Mg	S(WB)B	y=ax+b	0.8482	-9.8245	11	0.74	0.001	40–220	High
Mg	S(WB)BF	y=ax+b	*similar to S(WB)B*		15	0.48	0.004	40–220	Moderate
Mg	S(WB)BFR	y=ax+b	1.0608	-14.2080	11	0.87	<0.0001	40–220	High
Pinus sylvestris									
N	S(W)	$y=ax^b$	0.4498	1.1743	16	0.49	0.002	40–70	Moderate
N	S(WB)	$y=ax^b$	1.9726	0.8729	57	0.81	<0.0001	10–190	High
N	S(WB)B	$y=ax^b$	6.3295	0.7618	21	0.61	<0.0001	40–150	High
N	S(WB)BF	$y=ax^b$	12.8890	0.6529	55	0.70	<0.0001	10–150	High
N	S(WB)BFR	nd	nd	nd	3	nd	nd	nd	nd
P	S(W)	$y=ax^b$	0.2004	0.7569	16	0.74	<0.0001	40–70	High
P	S(WB)	$y=ax^b$	0.4737	0.6055	52	0.36	<0.0001	10–190	Moderate
P	S(WB)B	$y=ax^b$	1.9850	0.4652	19	0.29	0.017	40–150	Moderate
P	S(WB)BF	$y=ax^b$	1.7423	0.5641	53	0.50	<0.0001	10–150	High
P	S(WB)BFR	nd	nd	nd	1	nd	nd	nd	nd
K	S(W)	$y=ax^b$	3.3876	0.4704	16	0.28	0.034	40–70	Moderate
K	S(WB)	$y=ax^b$	0.7360	0.9240	54	0.82	<0.0001	10–190	High
K	S(WB)B	$y=ax^b$	7.4907	0.5067	21	0.37	0.003	40–150	Moderate
K	S(WB)BF	$y=ax^b$	4.9738	0.6723	54	0.72	<0.0001	10–150	High
K	S(WB)BFR	nd	nd	nd	1	nd	nd	nd	nd
Ca	S(W)	$y=ax^b$	1.1869	0.8259	16	0.69	<0.0001	40–70	High
Ca	S(WB)	$y=ax^b$	1.2434	0.9316	52	0.87	<0.0001	10–190	High
Ca	S(WB)B	$y=ax^b$	1.1151	1.0488	21	0.77	<0.0001	40–80	High
Ca	S(WB)BF	$y=ax^b$	5.9176	0.6874	52	0.78	<0.0001	10–150	High
Ca	S(WB)BFR	nd	nd	nd	1	nd	nd	nd	nd
Mg	S(W)	$y=ax^b$	0.3257	0.8056	16	0.76	<0.0001	40–70	High
Mg	S(WB)	$y=ax^b$	0.1738	1.0466	29	0.88	<0.0001	30–190	High
Mg	S(WB)B	$y=ax^b$	0.3942	0.9270	21	0.69	<0.0001	30–150	High
Mg	S(WB)BF	$y=ax^b$	0.6632	0.8755	27	0.71	<0.0001	30–150	High
Mg	S(WB)BFR	nd	nd	nd	1	nd	nd	nd	nd

nd, not determined due to the small number of case studies.

† Range of stem biomass in which models can be used to estimate nutrient stocks.

‡ Confidence index based on R^2 values (*low* : R^2=0.00-0.25; *moderate* : R^2=0.25-0.50; *high* : R^2=0.50-1.00).

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Effects of removing harvesting residues on physical-chemical and biological soil properties (“environmental impacts” dataset)

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Appendix D

Air Quality and Greenhouse Gas Emissions Modeling and Calculations

Placer Biomass Construction CAP Output

Placer-Mountain Counties County, Summer

Project Characteristics

Land Usage

Land Uses	Size	Metric
Parking Lot	8	Space
Other Asphalt Surfaces	2	Acre

Land Uses	Size	Metric
Unrefrigerated Warehouse-No Rail	10.8	1000sqft
General Light Industry	21.78	1000sqft

Other Project Characteristics

Urbanization	Urban	**Wind Speed (m/s)**	2.2	**Utility Company**	Pacific Gas & Electric Company
Climate Zone	14				
		Precipitation Freq (Days)	74		

User Entered Comments

Land Use -
Total Paved Area-2 acres less the parking area: 1.93 acres
Total Parking Spaces-8
The main building is similar to a warehouse in size/shape and material
The storage area would be approximately 1 acre (43,560 SF). This SF was reduced by 50% to represent less construction intensity since the structure would be a barn-style covered storage area

Construction Phase - Construction is assumed to take place for 6 months beginning May.

Grading - Total Disturbed Area-3.7 acres

Trips and VMT - Material hauled would be moved to onsite landfill

Emissions Summary

Overall Construction-Site Prep, Grading, Paving (Maximum Daily Emission)

Unmitigated Construction

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Year	lb/day										lb/day					
2013	4.79	40.24	23.03	0.04	6.26	2.22	8.20	3.32	2.22	5.26	0.00	4,336.39	0.00	0.43	0.00	4,345.39
Total	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**

Overall Construction-Building Construction, Arch. Coatings (Maximum Daily Emission)

Unmitigated Construction

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Year	lb/day										lb/day					
2013	69.09	17.30	12.37	0.02	0.23	1.07	1.30	0.01	1.07	1.08	0.00	2,222.79	0.00	0.21	0.00	2,227.19
Total	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**	**NA**

Construction Detail

Site Preparation - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Fugitive Dust					0.14	0.00	0.14	0.01	0.00	0.01						0.00
Off-Road	4.20	34.71	18.00	0.04		1.65	1.65		1.65	1.65		3,917.77		0.37		3,925.62
Total	**4.20**	**34.71**	**18.00**	**0.04**	**0.14**	**1.65**	**1.79**	**0.01**	**1.65**	**1.66**		**3,917.77**		**0.37**		**3,925.62**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Hauling	0.49	5.45	1.99	0.00	0.22	0.06	0.28	0.00	0.06	0.06		340.55		0.02		341.03
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Worker	0.07	0.07	0.70	0.00	0.10	0.00	0.11	0.00	0.00	0.01		78.06		0.01		78.18
Total	**0.56**	**5.52**	**2.69**	**0.00**	**0.32**	**0.06**	**0.39**	**0.00**	**0.06**	**0.07**		**418.61**		**0.03**		**419.21**

Grading - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Fugitive Dust					6.13	0.00	6.13	3.31	0.00	3.31						0.00
Off-Road	4.70	37.12	22.15	0.04		1.94	1.94		1.94	1.94		3,827.58		0.42		3,836.44
Total	**4.70**	**37.12**	**22.15**	**0.04**	**6.13**	**1.94**	**8.07**	**3.31**	**1.94**	**5.25**		**3,827.58**		**0.42**		**3,836.44**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Worker	0.09	0.08	0.88	0.00	0.13	0.00	0.13	0.00	0.00	0.01		97.58		0.01		97.73
Total	**0.09**	**0.08**	**0.88**	**0.00**	**0.13**	**0.00**	**0.13**	**0.00**	**0.00**	**0.01**		**97.58**		**0.01**		**97.73**

Paving - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Off-Road	4.16	25.92	16.81	0.03		2.21	2.21		2.21	2.21		2,393.42		0.37		2,401.25
Paving	0.23					0.00	0.00		0.00	0.00						0.00
Total	**4.39**	**25.92**	**16.81**	**0.03**		**2.21**	**2.21**		**2.21**	**2.21**		**2,393.42**		**0.37**		**2,401.25**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Worker	0.13	0.12	1.32	0.00	0.20	0.01	0.20	0.01	0.01	0.01		146.37		0.01		146.59
Total	**0.13**	**0.12**	**1.32**	**0.00**	**0.20**	**0.01**	**0.20**	**0.01**	**0.01**	**0.01**		**146.37**		**0.01**		**146.59**

Building Construction - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Off-Road	2.20	16.33	10.77	0.02		1.04	1.04		1.04	1.04		1,945.40		0.20		1,949.52
Total	**2.20**	**16.33**	**10.77**	**0.02**		**1.04**	**1.04**		**1.04**	**1.04**		**1,945.40**		**0.20**		**1,949.52**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Vendor	0.07	0.86	0.37	0.00	0.05	0.03	0.07	0.00	0.03	0.03		140.79		0.00		140.85
Worker	0.12	0.12	1.23	0.00	0.18	0.01	0.19	0.01	0.01	0.01		136.61		0.01		136.82
Total	**0.19**	**0.98**	**1.60**	**0.00**	**0.23**	**0.04**	**0.26**	**0.01**	**0.04**	**0.04**		**277.40**		**0.01**		**277.67**

Architectural Coating - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Archit. Coating	68.58					0.00	0.00		0.00	0.00						0.00
Off-Road	0.49	2.96	1.94	0.00		0.27	0.27		0.27	0.27		281.19		0.04		282.10
Total	**69.07**	**2.96**	**1.94**	**0.00**		**0.27**	**0.27**		**0.27**	**0.27**		**281.19**		**0.04**		**282.10**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	lb/day										lb/day					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00		0.00		0.00		0.00
Worker	0.03	0.02	0.26	0.00	0.04	0.00	0.04	0.00	0.00	0.00		29.27		0.00		29.32
Total	**0.03**	**0.02**	**0.26**	**0.00**	**0.04**	**0.00**	**0.04**	**0.00**	**0.00**	**0.00**		**29.27**		**0.00**		**29.32**

Summary of Construction CAP Emissions

Construction Phase	NOx (lb/day)	ROG (lb/day	PM10 (lb/day)	PM2.5 (lb/day)	Source
Site Preparation					
Construction On-site	34.71	4.2	1.79	1.66	Constr CAP CalEEMod Output
Construction Off-site	5.52	0.56	0.39	0.07	Constr CAP CalEEMod Output
Phase Total	**40.23**	**4.76**	**2.18**	**1.73**	Summation
Grading					
Construction On-site	37.12	4.7	8.07	5.25	Constr CAP CalEEMod Output
Construction Off-site	0.08	0.09	0.13	0.01	Constr CAP CalEEMod Output
Phase Total	**37.2**	**4.79**	**8.2**	**5.26**	
Paving					
Construction On-site	25.92	4.39	2.21	2.21	Constr CAP CalEEMod Output
Construction Off-site	0.12	0.13	0.2	0.01	Constr CAP CalEEMod Output
Phase Total	**26.04**	**4.52**	**2.41**	**2.22**	Summation
Building Construction					
Construction On-site	16.33	2.2	1.04	1.04	Constr CAP CalEEMod Output
Construction Off-site	0.98	0.19	1.04	0.04	Constr CAP CalEEMod Output
Phase Total	**17.31**	**2.39**	**2.08**	**1.08**	Summation
Architectural Coating					
Construction On-site	2.96	69.07	0.27	0.27	Constr CAP CalEEMod Output
Construction Off-site	0.02	0.03	0.04	0	Constr CAP CalEEMod Output
Phase Total	**2.98**	**69.1**	**0.31**	**0.27**	Summation

	Maximum Emissions (lb/day)				
	NOx	**ROG**	**PM10**	**PM2.5**	
All Phases	40.2	69.1	8.2	5.3	MAX Function

Notes

Construction phases are listed in the order that they would likely occur.
There would be no substantial difference in construction emissions under the gasification alternatives and direct combustion alternative.

Placer Biomass Construction GHG Output
Placer-Mountain Counties County, Annual

2.0 Emissions Summary

Overall Construction-Site Preparation, Grading, Paving

Unmitigated Construction

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Year	tons/yr										MT/yr					
2013	0.15	1.13	0.69	0.00	0.07	0.06	0.13	0.03	0.06	0.10	0.00	106.88	106.88	0.01	0.00	107.14
Total	**0.15**	**1.13**	**0.69**	**0.00**	**0.07**	**0.06**	**0.13**	**0.03**	**0.06**	**0.10**	**0.00**	**106.88**	**106.88**	**0.01**	**0.00**	**107.14**

Overall Construction-Building Construction, Arch. Coatings

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Year	tons/yr										MT/yr					
2013	0.43	0.39	0.28	0.00	0.00	0.02	0.03	0.00	0.02	0.02	0.00	44.68	44.68	0.00	0.00	44.77
Total	**0.43**	**0.39**	**0.28**	**0.00**	**0.00**	**0.02**	**0.03**	**0.00**	**0.02**	**0.02**	**0.00**	**44.68**	**44.68**	**0.00**	**0.00**	**44.77**

3.0 Construction Detail

3.1 Mitigation Measures Construction

Site Preparation - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Fugitive Dust					0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Off-Road	0.05	0.40	0.21	0.00		0.02	0.02		0.02	0.02	0.00	40.86	40.86	0.00	0.00	40.94
Total	**0.05**	**0.40**	**0.21**	**0.00**	**0.00**	**0.02**	**0.02**	**0.00**	**0.02**	**0.02**	**0.00**	**40.86**	**40.86**	**0.00**	**0.00**	**40.94**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Hauling	0.01	0.06	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.35	3.35	0.00	0.00	3.35
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Worker	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.76	0.76	0.00	0.00	0.76
Total	**0.01**	**0.06**	**0.04**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**4.11**	**4.11**	**0.00**	**0.00**	**4.11**

Grading - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Fugitive Dust					0.06	0.00	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00
Off-Road	0.05	0.37	0.22	0.00		0.02	0.02		0.02	0.02	0.00	34.71	34.71	0.00	0.00	34.79
Total	**0.05**	**0.37**	**0.22**	**0.00**	**0.06**	**0.02**	**0.08**	**0.03**	**0.02**	**0.05**	**0.00**	**34.71**	**34.71**	**0.00**	**0.00**	**34.79**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Worker	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.82	0.82	0.00	0.00	0.82
Total	**0.00**	**0.00**	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.82**	**0.82**	**0.00**	**0.00**	**0.82**

Paving - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Off-Road	0.05	0.30	0.19	0.00		0.03	0.03		0.03	0.03	0.00	24.96	24.96	0.00	0.00	25.04
Paving	0.00					0.00	0.00		0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total	**0.05**	**0.30**	**0.19**	**0.00**		**0.03**	**0.03**		**0.03**	**0.03**	**0.00**	**24.96**	**24.96**	**0.00**	**0.00**	**25.04**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Worker	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.42	1.42	0.00	0.00	1.42
Total	**0.00**	**0.00**	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**1.42**	**1.42**	**0.00**	**0.00**	**1.42**

Building Construction - 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Off-Road	0.05	0.35	0.23	0.00		0.02	0.02		0.02	0.02	0.00	37.93	37.93	0.00	0.00	38.01
Total	**0.05**	**0.35**	**0.23**	**0.00**		**0.02**	**0.02**		**0.02**	**0.02**	**0.00**	**37.93**	**37.93**	**0.00**	**0.00**	**38.01**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vendor	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.73	2.73	0.00	0.00	2.73
Worker	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.48	2.48	0.00	0.00	2.48
Total	**0.00**	**0.02**	**0.04**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**5.21**	**5.21**	**0.00**	**0.00**	**5.21**

Architectural Coating- 2013

Unmitigated Construction On-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Archit. Coating	0.38					0.00	0.00		0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Off-Road	0.00	0.02	0.01	0.00		0.00	0.00		0.00	0.00	0.00	1.40	1.40	0.00	0.00	1.41
Total	**0.38**	**0.02**	**0.01**	**0.00**		**0.00**	**0.00**		**0.00**	**0.00**	**0.00**	**1.40**	**1.40**	**0.00**	**0.00**	**1.41**

Unmitigated Construction Off-Site

	ROG	NOx	CO	SO2	Fugitive PM10	Exhaust PM10	PM10 Total	Fugitive PM2.5	Exhaust PM2.5	PM2.5 Total	Bio- CO2	NBio- CO2	Total CO2	CH4	N2O	CO2e
Category	tons/yr										MT/yr					
Hauling	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vendor	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Worker	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.14	0.00	0.00	0.14
Total	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.00**	**0.14**	**0.14**	**0.00**	**0.00**	**0.14**

Summary of Construction GHG Emissions

Construction Phase	CO2e (MT)	Source
Site Preparation		
Construction On-site	40.94	Constr CAP CalEEMod Output
Construction Off-site	4.11	Constr CAP CalEEMod Output
Phase Subtotal	45.05	Summation
Grading		
Construction On-site	34.79	Constr CAP CalEEMod Output
Construction Off-site	0.82	Constr CAP CalEEMod Output
Phase Subtotal	35.61	Summation
Paving		
Construction On-site	25.04	Constr CAP CalEEMod Output
Construction Off-site	1.42	Constr CAP CalEEMod Output
Phase Subtotal	26.46	Summation
Building Construction		
Construction On-site	38.01	Constr CAP CalEEMod Output
Construction Off-site	5.21	Constr CAP CalEEMod Output
Phase Subtotal	43.22	Summation
Architectural Coating		
Construction On-site	1.41	Constr CAP CalEEMod Output
Construction Off-site	0.03	Constr CAP CalEEMod Output
Phase Subtotal	0.14	Summation

	value	units	source
Total CO2e from Construction	150.48	MT	summation
Expected operational life of the plant	30	years	conservative assumption
Amortized CO2e over life of the plant	5	MT/year	amortization calculation

Notes

Construction phases are listed in the order that they would likely occur.

There would be no substantial difference in construction emissions under the gasification alternatives and direct combustion alternative.

Summary of Operational Emissions of Criteria Air Pollutants and Precursors (Maximum Daily [lb/day])

	Gasification Alternatives				Direct Combustion Alternative					Jurisdiction					
	NOx	ROG	PM_{10}	$PM_{2.5}$	NOx	ROG	PM_{10}	$PM_{2.5}$	Source Worksheet	Applicable Note(s)	PCAPCD portion of MCAB	PCAPCD portion of LTAB	EDAPCD portion of LTAB	NSAQMD portion of MCAB	Nevada portion of LTAB
Construction Emissions	40.2	40.2	40.2	40.2	40.2	40.2	40.2	40.2	Construction_Emissions.xlsx		all				
Operational Emissions															
Biomass Combustion by Power Plant	15.4	72.0	14.4	14.4	53.8	5.8	13.4	13.4	Plant CAPs		all	none	none	none	none
Natural Gas Combustion by Power Plant	0.0	0.0	0.0	0.0	0.4	0.01	0.02	0.02	Natural Gas Comb	2, 3	all	none	none	none	none
Chipping Biomass	42.2	4.0	1.4	1.4	49.6	4.7	1.6	1.6	Chipping	2, 4	some	some	some	some	some
Truck Activity at the Plant	0.7	0.01	0.01	0.005	0.8	0.01	0.01	0.005	Truck@Plant		all	none	none	none	none
Loader Activity at the Plant	8.8	0.9	0.3	0.3	8.8	0.9	0.3	0.3	Loader@Plant		all	none	none	none	none
Employee Commute Trips	0.3	0.3	0.004	0.003	0.3	0.3	0.004	0.004	Employee Commute	5	some	some	none	some	none
Trucks Hauling Biomass	9.5	0.2	13.4	1.4	11.1	0.3	15.8	1.7	Trucks Hauling Biomass	6	some	some	some	some	some
Trucks Hauling Biochar or Ash	0.9	0.05	0.01	0.01	0.9	0.05	0.01	0.01	Trucks Hauling Biochar-Ash		some	none	none	some	some
Total Operational Emissions	77.7	77.5	29.5	17.5	125.7	12.1	31.2	17.1	summation		some	some	some	some	some

Air District Thresholds of Significance, Maximum Daily Emissions (lb/day)									
PCAPCD	NOx	ROG	PM_{10}	$PM_{2.5}$	NOx	ROG	PM_{10}	$PM_{2.5}$	Source
Construction-Related Emissions	82	82	82	none	82	82	82	none	PCAPCD
Operational Emissions	82	82	82	none	82	82	82	none	PCAPCD
EDCAPCD									
Operational Emissions	82	82	none	none	82	82	none	none	EDCAPCD 2002 (Source 1)
NSAQMD Level									
Level A tier	24	24	79	none	24	24	79	none	NSAQMD 2009 (Source 2)

	NOx	ROG	PM_{10}	$PM_{2.5}$	NOx	ROG	PM_{10}	$PM_{2.5}$	Source Worksheet	Applicable Note(s)	Jurisdiction: PCAPCD portion of MCAB	PCAPCD portion of LTAB	EDAPCD portion of LTAB	NSAQMD portion of MCAB	Nevada portion of LTAB
Average Daily Emissions from Open Burning of Forest Thinning Slash (lb/day)	428.2	557.5	917.2	778.3	520.0	677.0	1,113.7	945.1	Open Burn CAPs in Forest	See Note 7	some	some	some	some	some

Notes

1 Avoided emissions are not accounted for in the estimate of the net increase in maximum daily emissions.
2 This activity would occur during the summer period only when biomass is collected from the forests.
3 Natural gas would only be used during start-ups under the direct combustion alternative. On these days, a full-load of biomass fuel would not be combusted. Thus, the maximum daily operational emissions of the plant would not occur on start-up days.
4 Chipping and associated emissions would occur in the forests where the biomass is recovered. Thus, chipping would occur in all the affected air basins and jurisdictions. It is not likely that all the biomass chipped on any particular day would occur in PCAPCD's jurisdiction of the MCAB because most of the biomass would be sourced from other areas.
5 More employees would be employed by the project during the summer period when biomass is being collected from the forests. Thus, emissions associated with employee commuting would be lower during the winter period.
6 Not all truck travel associated with the hauling of biomass to the plant would occur in the jurisdiction of any single air district.
7 This analysis does not account for the reduction in emissions of CAPs and precursors that would occur due to the fact that the biomass recovered from the forests would no longer be piled and burned in the forests. This is because the analysis focuses on the maximum daily net increase in emissions associated with the project and the timing of burning is unknown.

Sources

1 El Dorado County Air Pollution Control District. 2002 (February). Guide to Air Quality Assessment. First Edition. Placerville, CA.
2 Northern Sierra Air Quality Management District. 2009. Guidelines for Assessing and Mitigating Air Quality Impacts of Land Use Projects. A draft last revised on August 18, 2009. Truckee, CA.

Summary of Annual Greenhouse Gas Emissions (MT CO2-e/year)

	Gasification Alternatives	Direct Combustion Alternative	Source Worksheet
Construction Emissions, Amortized	5	5	Construction_Emissions.xlsx
Operational Emissions			
Biomass Combustion by Power Plant	26,526	31,207	Plant GHGs
Support Emissions			
Natural Gas Combustion by Power Plant	0	1	Natural Gas Comb
Chipping Biomass	301	354	Chipping
Trucks Hauling Biomass to the Plant	84	99	Trucks Hauling Biomass
Truck Activity at the Plant	2	2	Truck@Plant
Loader Activity at the Plant	197	197	Loader@Plant
Employee Commute Trips	35	44	Employee Commute
Trucks Hauling Biochar/Ash from the Plant	10	10	Trucks Hauling Biochar-Ash
Electricity Consumption from the Grid	1,134	1,134	Electricity Consumption
Water Consumption	222	355	Water Consumption
Wastewater Treatment	156	250	Wastewater Treatment
Total Operational and Support Emissions	**28,667**	**33,654**	subtotal
Avoided Emissions			
Open Burning of Forest Thinning Slash and Hazardous Fuels	24,858	29,245	Open Burn GHGs in Forests
Net Increase in Emissions	**3,809**	**4,409**	net calculation
Electricity Generated by the Plant (MW/year)	2.0	2.0	Operational Parameters
days of operation per year	365	365	Operational Parameters
Hours of operation per day	24	24	Unit Conversions
MW-hr/year	17,520	17,520	calculation
GHG Efficiency (MT CO2e/MW-hr)	**0.22**	**0.25**	calculation
GHG Efficiency Threshold (MT CO2e/MW-hr)	**0.28**		wksht: GHG TOS
Reduction in net GHGs necessary to reduce the GHG efficiency of the direct combustion alternative to less than the GHG Efficiency Threshold (MT CO2-e/year).		155	calculation
GHG Efficiency under Direct Combustion with mitigation (MT CO2e/MW-hr)		0.24	calculation

GHG Efficiency of Electricity Production

	value	units	source
Projected CO2e from Electricity consumption in 2020 (business as usual)	121.2	MMT/year	Source 1, p. 1
Projected CO2e reductions in 2020 achieved by Renewable Portfolio Standard (20%)	12.0	MMT/year	Source 2, p. 2
Projected CO2e reductions in 2020 achieved by Renewable Electricity Standard (33%)	11.4	MMT/year	Source 2, p. 2
Projected CO2e reductions in 2020 achieved by energy efficiency programs	7.8	MMT/year	Source 2, p. 3; Source 4; See Note 1
Million Solar Roofs	1.1	MMT/year	Source 2, p. 4; See Note 2
Projected GHGs Associated with Electricity Consumption in 2020	88.9	MMT/year	calculation
energy conversion rate	1,000	MWh/GWh	onlineconversion.com
mass conversion rate	1,000,000	MT/MMT	onlineconversion.com
Projected Electricity Demand in 2020	316,280	GWh/year	Source 3, Form 1.1
Projected GHG Efficiency of Electricity Consumption in 2020	0.28	MT CO2e/MW-hr	efficiency calc.

Sources

1 California Air Resources Board (ARB). 2011 (October 26). California Greenhouse Gas Inventory 2000-2009—by Category as Defined in the Scoping Plan. Available: <http://www.arb.ca.gov/cc/inventory/data/tables/ghg_inventory_scopingplan_00-09_2011-10-26.pdf>. Accessed May 5, 2012.

2 California Air Resources Board (ARB). 2011. Status of Scoping Plan Recommended Measures. Available: <http://www.arb.ca.gov/cc/scopingplan/status_of_scoping_plan_measures.pdf>. Accessed May 5, 2012.

3 Kavalec, Chris and Tom Gorin, 2009. California Energy Demand 2010-2020, Adopted Forecast. Publication Number: CEC-200-2009-012-CMF. See Form 1.1, Total Electricity Consumption by Sector, on the spread sheets. Available: <http://www.energy.ca.gov/2009publications/CEC-200-2009-012/index.html>. Accessed May 5, 2012.

Notes

1 According to Source 2, approximately 11.9 MMT CO2e/year of GHG reductions are expected in 2020 from energy efficiency programs. This is from reduced demand of both electricity and natural gas. According to correspondence with Dave Mehle of ARB (see Source 4), 7.8 MMT CO2e/year would specifically be from associated reduction in demand for electricity and this value is used in this calculation.

2 It is uncertain at this time whether the Million Solar Roofs program will be implemented and fully achieved. Accounting for this measure results in a more conservative estimate of the needed GHG efficiency to be consistent with AB 32 goals.

Methodologies Employed to Estimate Project-Related Emissions

Category and Source	Applicability to Alternatives: Gasification Alternatives	Applicability to Alternatives: Direct Combustion	Model/Protocol/Sources of Emission Factors	Key Input Parameter(s)	Applicable Worksheet(s)
Construction Emissions	Y	Y	CalEEMod	off-road equipment use, ground disturbance	
Operational Emissions					
Syngas or Biomass Combustion by Power Plant	Y	Y	For CAPs, data provided by Technology Providers a	technology type, mass of biomass fuel consum	Plant CAPs; Plant GHGs
Support Emissions					
Natural Gas Combustion by Power Plant	NA	Y	AP 42 Emission Factors for CAPs; For GHGs, ARB's Mandatory Reporting Guidelines	volume of natural gas consumed	Natural Gas Comb
Chipping Biomass	Y	Y	ARB's OFFROAD2007 model, pilot study	diesel equipment use	Chipping
Trucks Hauling Biomass to the Plant	Y	Y	ARB's EMFAC2011 model	VMT	Trucks Hauling Biomass
Truck Activity at the Fuel Yard	Y	Y	ARB's EMFAC2011 model	idle-hours by trucks	Trucks @Fuel Yard
Loader Activity at the Plant and Fuel Yard	Y	Y	ARB's OFFROAD2007 model	time at power plant	Loader @Plant
Employee Commute Trips	Y	Y	ARB's EMFAC2011 model	VMT	Employee Commute
Trucks Hauling Biochar/Ash from the Plant	Y	Y	ARB's EMFAC2011 model	VMT	Trucks Hauling Biochar-Ash
Water Consumption	Y	Y	CEC's electricity consumption rate	water demand, well specifications	Water Consumption
Wastewater Treatment	Y	Y	CalEEMod water module	volume of wastewater	Wastewater Treatment
Avoided Emissions					
Open Burning of Forest Thinning Slash and Hazardous Fuels	Y	Y	ARB's Mandatory Reporting Guidance	mass of biomass, high heating value	Open Burn in Forests

Operational Parameters	Gasification Alternatives	Direct Combustion Alternative	Units	Source
Plant location	Cabin Creek	Cabin	none	Chapter 3, Project Alternatives
Jurisdiction	Placer Co.	Placer Co.	none	Chapter 3, Project Alternatives
Technology	gasification	direct combustion	none	Chapter 3, Project Alternatives
Plant Power Capacity				
Gross	2.0	2.0	MW	Chapter 3, Project Description, p. 3-14
parasitic load, percentage	10%	10%	%	Source 1
Net (export)	1.8	1.8	MW	calculation
Operational Life	30	30	years	conservative assumption
Operational frequency				
annual basis	365	365	days/year	Chapter 3, Project Description, p. 3-14; See Note 1
Earliest operation date	January 2015	January 2015	year on calendar	e-mail from Brett Storey to N. Hansel on 4/4/2012 and forwarded to AJK
Fuel demand of power plant, minimum	14,000	17,000	bdt/year	Project Description and Alternatives; See Note 1
Fuel demand of power plant, maximum	17,000	20,000	bdt/year	Project Description and Alternatives; See Note 1
Moisture content of forest-sourced biomass, maximum	50%	50%	%	Chapter 3, Project Description, p. 3-10
Fuel demand of power plant, annual, Green, minimum	28,000	34,000	grn ton/year	calculation
Fuel demand of power plant, annual, Green, maximum	34,000	40,000	grn ton/year	calculation
Source of biomass fuel				
hazardous fuels reduction	75%	75%	%	Chapter 3, Project Description, p. 3-15
forest thinning	25%	25%	%	Chapter 3, Project Description, p. 3-15
Biochar or ash byproduct	biochar	ash		
Mass of biochar/ash produced annually	850	1,000	tons/year	Chapter 3, Project Description, p. 3-13
High Heat Value of forest-sourced biomass, minimum	8,300	8,300	BTU/dry lb	Chapter 3, Project Description, p. 3-10
Water consumption, Maximum worst-case	14,400	23,040	gal/day	See Note 2
Water source	well	well	NA	Chapter 3, Project Description
Capacity of chip van truck, dry, by mass	12.5	12.5	bdt/load	p. 3-15 of Sec. 3, Project Description

Notes

1 Under the gasification alternatives, the two parallel 1-MW systems would operate, which allows the plant to operate at 50% capacity while one of the systems is undergoing maintenance. If, for some unforseen reason, both systems need to be shut down, then the plant would consume less biomass on an annual basis. Under the direct combustion alternative, the entire plant would be shut down on occasion for maintenance activities, in which case the annual mass of biomass fuel combusted would be lower.

2 The volume of water consumption for the gasification alternatives is stated in Chapter 3, Project Description and is the maximum, worst-case volume of water that could be consumed by a gassification system. Depending on the gasification technology used, substantially less water may be consumed by the plant. The volume consumed under the direct combustion alternative is calculated based on a maximum rate of 8 gal/min/MW * 2 MW * 60 min/hr * 24 hr/day.

Sources

1 Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 8, 2012—telephone conversation with Austin Kerr of Ascent Environmental regarding the quantity of electricity that the biomass plant would consume.

Unit Conversion Rates

Global Warming Potential (rates)

	CO_2	CH_4	N_2O	units
global warming potential	1	21	310	unitless

Source: Table A-1 of Subpart A—Global Warming Potentials (100-YearTime Horizon) from 40 CFR 98 (page 722-723), as required by ARB's Regulation for the Mandatory Reporting of GHGs (http://www.arb.ca.gov/cc/reporting/ghg-rep/regulation/mrr_2010_clean.pdf)

Mass Conversion Rates

value	units	source
1,000	kg/MT	onlineconversion.com/weight_common.htm
1,000,000	g/MT	onlineconversion.com/weight_common.htm
2,000	lb/ton	onlineconversion.com/weight_common.htm
2,204.62	lb/MT	onlineconversion.com/weight_common.htm
453.59	g/lb	onlineconversion.com/weight_common.htm
1.1023	ton/MT	onlineconversion.com/weight_common.htm
2.204622622	lb/kg	onlineconversion.com/weight_common.htm
2,204.62	lb/MT	onlineconversion.com/weight_common.htm
1,000	g/kg	onlineconversion.com/weight_common.htm
907.2	kg/ton	onlineconversion.com/weight_common.htm

Time Conversions Rates

rate	units	source
60	min/hr	definition
365	days/year	definition
7	days/wk	definition
24	hr/day	definition

Energy Conversions Rates

value	units	source
1,000,000	btu/MMbtu	definition

Electricity Conversion Rates

value	units	source
1,000	kW-hr/MW-hr	onlineconversion.com/energy.htm

Volume - Liquid

	units	source
1,000,000	gal/MG	definition
325,851	gal/acre-ft	onlineconversion.com/volume.htm

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DE-FG36-08GO88026. Auburn, CA. Prepared for the Placer County Planning Department.

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— South Coast Air Quality Management District. 2011. California Emissions Estimator Model (CalEEMod) Version 2011.1.1. Available: <http://www.caleemod.com/>.

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— Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 3, 2012—e-mail to Austin Kerr of Ascent Environmental regarding the volume of natural gas that would be consumed by the biomass plant under the gasification and direct combustion alternatives.

— Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 8, 2012—telephone conversation with Austin Kerr of Ascent Environmental regarding the quantity of electricity that the biomass plant would consume.

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Emissions of CAPs and Precursors from Power Plant Stack (Gasification and Direct Combustion)

	NOx	ROG	PM10	CO	SO2	units	Source(s)
Gasification							
Technology A: Nexterra							
hourly emissions	0.50	3.00	0.60	6.50	0.01	lb/hr	Source 1, p. 12; See Notes 1, 4, 7
daily emissions	12.00	72.00	14.40	156.00	0.24	lb/day	conversion calculation
annual emissions	2.10	11.75	2.41	26.50	0.04	tons/year	Source 1, p. 12 and Table 2-4 on p. 20; See Notes 1, 4, 7
Technology B: Phoenix							
hourly emissions	0.64	0.64	0.3	3.4	0.05	lb/hr	Source 1, p. 13-14; See Notes 2, 4, 7
daily emissions	15.36	15.36	7.20	81.60	1.20	lb/day	conversion calculation
annual emissions	2.58	2.58	1.20	13.39	0.19	tons/year	Source 1, p. 13-14; and Table 2-3 on p. 19; See Notes 2, 4, 7
Max. Annual Emissions	2.58	11.75	2.41	26.50	0.19	tons/year	max calc based on two gasification technologies
Max. Daily Emissions	15.36	72.00	14.40	156.00	1.20	lb/day	max calc based on two gasification technologies
Direct Combustion							
Technology: Envio							
hourly emissions	2.24	0.24	0.56	5.60	0	lb/hr	Source 1, p. 15; See Notes 3, 4, 7
daily emissions	53.76	5.76	13.44	134.40	0.00	lb/day	conversion calculation
annual emissions	9.05	0.96	2.30	22.60	0	tons/year	Source 1, p. 15; See Notes 3, 4, 7

	value	units	source
time conversion rate	24	hr/day	wksht: Unit Conversions

Notes

1 The estimate of NOx emissions assumes the installation of Selective Catalytic Reduction (SCR), which reduces NOx through introduction of ammonia in the presence of a precious metal catalyst at a temperature of approx. 700F, according to

2 The Phoenix unit utilizes a "rich-burn" internal combustion engine, involving minimizing (starving) the air to combust the syngas. This allows for the use of a three-way catalytic converter to reduce NOx, CO, and VOCs, according to TSS & PCAPCD 2011, p. 13. Discussion with Phoenix Energy, Authority to Construct Permit issued by the San Joaquin Air Pollution Control District for Phoenix Facility in Merced, CA.

3 Emission estimates for direct combustion assume the use of a bag house for PM control, and the use of Selective Non-Catalytic Reduction for NOx control, which is commercially available and well demonstrated for NOx control in

4 Reactive Organic Gases (ROG) are a subset of Volatile Organic Compounds (VOCs). However, in order to be conservative it is assumed that all VOCs emitted by the power plant are ROG. PCAPCD staff, including Bruce Springsteen of PCAPCD, agrees with this assumption.

5 The emission estimates for criteria air pollutants and precursors are conservatively high because they assume that the biomass fuel would have a Btu content of 8,000 Btu/dry lb and the actual Btu content of forest-sourced biomass is expected to be 8,300 Btu/dry lb. This means that less biomass would actually be needed due to its higher heat content. Bruce Springsteen of PCAPCD agrees (Springsteen, pers. comm., 2012).

6 It is conservatively assumed that PM2.5 emissions would be equal to PM10 emissions.

7 Emissions estimates shown are based on projected use of best available emissions controls, as would be required by PCAPCD Rule 502, according to Source 3, p. 3.

Source

1 TSS Consultants and Placer County Air Pollution Control District. 2011 (November). *Air/Water Emissions and Carbon Credits/Emissions Offsets.* Prepared for the Placer County Biomass Program.

2 Springsteen, Bruce. Associate Engineer. Placer County Air Pollution Control District, Auburn, CA. May 1, 2012—telephone conversation and e-mail with Austin Kerr of Ascent Environmental regarding methodologies for estimating criteria air

3 Placer County Air Pollution Control District. 2012 (April). New Source Review Permit Analysis for Small-Scale Biomass Combined Heat and Power in the Lake Tahoe Region. Task 6.0 of the U.S. Department of Energy/Placer County Biomass Utilization Pilot Project
DE-FG36-08GO88026. Auburn, CA. Prepared for the Placer County Planning Department.

Annual GHG Emissions from Biomass Power Plant

Key Note: There are typically multiple methodologies, called "tiers," for estimating GHG emissions for different sources. A Tier 1 approach relies on default emission factors and default values for other key input parameters such as the high heating value. The default values used in a Tier 1 methodology are conservative in that they result in a high estimate of GHG emissions. A Tier 2 approach uses project specific emission factors and/or input parameters (e.g., high heating value) that result in a more accurate, and lower estimate of GHG emissions. It is important to note that this analysis is conservative because a Tier 2 method is used to estimate GHG emissions from the gasification or combustion of biomass at the proposed plant and a Tier 1 approach is used to estimate the avoided level of GHG emissions that would be generated by open burning the biomass fuel in the forests.

Plant Specifications	gasification alternatives	direct combustion alternative	units	source/notes
maximum mass of biomass combusted	17,000	20,000	bdt/year	wksht: 3-Operational Parameters; See Note 5

Calculation Parameters	value	units	source/notes
high heat value of biomass, minimum	8,300	btu/dry lb	wksht: 3-Operational Parameters
mass conversion rate	2,000	lb/ton	wksht: Unit Conversions
energy conversion rate	1,000,000	btu/MMbtu	wksht: 20-Unit Conversions

Emission Factor	value	units	source/notes
CO2 emission factor	93.80	kg/MMbtu	See Note 1
CH4 emission factor	0.0032	kg/MMbtu	See Note 2
N2O emission factor	0.00042	kg/MMbtu	See Note 2
Global Warming Potential for Conversion to CO2e			
global warming potential of CH4	21	unitless	wksht: Unit Conversions
global warming potential of N2O	310	unitless	wksht: Unit Conversions
CO2-e Emission Factor	94.00	kg/MMbtu	composite calculation

Conversion Rates	value	units	source/notes
mass conversion rate	1.102	ton/MT	wksht: Unit Conversions
mass conversion rate	1,000	kg/MT	wksht: 20-Unit Conversions

GHG Emissions, maximum annual	gasification alternatives	direct combustion alternative	units	source/notes
Energy Content of Bone Dry Biomass	282,200	332,000	MMbtu/year	calculation; See Note 3
CO2-e emissions, maximum annual	26,526	31,207	MT/year	See Note 3 and Note 4
CO2-e emissions, maximum annual	29,240	34,400	ton/year	conversion calculation

Sources

1 California Air Resources Board. 2012. ARB's Regulation for the Mandatory Reporting of GHGs. Available: <http://www.arb.ca.gov/cc/reporting/ghg-rep/regulation/mrr_2010_clean.pdf>, which is hyperlinked to <http://www.arb.ca.gov/cc/reporting/ghg-rep/regulation/2010_regulation.htm>. Accessed April 25, 2012. Last updated February 29, 2012.

2 Code of Federal Regulations. Part 98-Mandatory Greenhouse Gas Reporting. Subparts A-C. Available: <http://www.epa.gov/climatechange/emissions/downloads09/RuleParts98SubpartsA-C.pdf>. Accessed April 25, 2012.

3 Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 3—e-mail to Austin Kerr of Ascent Environmental regarding the volume of natural gas that would be consumed by the biomass plant under the gasification and direct combustion alternatives.

Notes

1 Table C-1 to Subpart C - Default CO2 Emission Factors and High Heating Values for Various Types of Fuel from 40 CFR 98 (page 789-790), which is Source 2, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), which is Source 1

2 Table C-2 to Subpart C - Default CH4 and N2O Emission Factors for Various Types of Fuel from 40 CFR 98 (page 790-791), which is Source 2, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), which is Source 1

3 Equations C-2a and C-8 from 40 CFR 98 (page 728), which is Source 2, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), which is Source 1.

4 This methodology for estimating GHG emissions is considered a Tier 2 methodology, as defined on p. 47 of Source 1 (Definition number 358), because it uses the default emission factor and a measured high heat value for the biomass fuel.

5 This calculation is conservative because it assumes that the maximum range of BDT would be consumed by the biomass plant and that all the biomass consumed by the plant would be forest-sourced biomass, which has a higher HHV Btu content than WUI-sourced biomass on an average annual basis.

Combustion of Natural Gas for Start-Ups

Specifications	gasification alternatives	direct combustion alternative	units	source/notes
natural gas used per plant start-up	0	2.35	MMbtu/start-up	Source 2
start-ups per year	12	12	start-ups/year	conservative estimate, See Source 2
annual consumption of natural gas	0	28.20	MMbtu/year	calculation
volume of natural gas per start-up	0	0.0023	MMscf/start-up	conversion calculation
volume of natural gas per year	0	0.0276	MMscf/year	conversion calculation

	value	units	source/notes
energy content of natural gas per volume	1,020	MMbtu/MMscf	Source 1, p. 1.4-6

	NOx	ROG	PM_{10}	$PM_{2.5}$	units	source/notes
CAP Emission Factors	190	5.50	7.6	7.6	lb/MMscf	Source 1; See Note 1
CAP Emissions, Maximum Daily						
Gasification Alternatives	0.00	0.00	0.00	0.00	lb/day	calculation; See Note 2
Direct Combustion Alternative	0.44	0.01	0.02	0.02	lb/day	calculation; See Note 2

GHG Emission Factors	value	units	source/notes
CO2 emission factor	53.02	kg/MMbtu	See Note 3
CH4 emission factor	0.001	kg/MMbtu	See Note 4
N2O emission factor	0.0001	kg/MMbtu	See Note 4
Global Warming Potential for Conversion to CO2e			
CH4	21	unitless	wksht: Unit Conversions
N2O	310	unitless	wksht: Unit Conversions
mass conversion rate	1,000	kg/MT	wksht: 20-Unit Conversions
CO2-e Emission Factor	0.053	MT/MMbtu	composite calculation

	gasification alternatives	direct combustion alternative	units	source/notes
CO2-e Emissions, maximum annual	0.0	1.5	MT/year	calc; See Note 4 and Note 5

Sources

1 U.S. Environmental Protection Agency (EPA). 1998 (July). Emission Factors & AP 42, Compilation of Air Pollutant Emission Factors, Section 1.4, Natural Gas Combustion. From EPA's Technology Transfer Network's Clearinghouse for Inventories and Emission Factors. Available: <http://www.epa.gov/ttnchie1/ap42/ch01/final/c01s04.pdf>. Accessed May 7, 2012.

2 Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 3, 2012—e-mail to Austin Kerr of Ascent Environmental regarding the volume of natural gas that would be consumed by the biomass plant under the gasification and direct combustion alternatives.

3 California Air Resources Board. 2012. ARB's Regulation for the Mandatory Reporting of GHGs. Available: <http://www.arb.ca.gov/cc/reporting/ghg-rep/regulation/mrr_2010_clean.pdf>, which is hyperlinked to <http://www.arb.ca.gov/cc/reporting/ghg-rep/regulation/2010_regulation.htm>. Accessed April 25, 2012. Last updated February 29, 2012.

4 Code of Federal Regulations. Part 98-Mandatory Greenhouse Gas Reporting. Subparts A-C. Available: <http://www.epa.gov/climatechange/emissions/downloads09/RuleParts98SubpartsA-C.pdf>. Accessed April 25, 2012.

Notes

1 The emission factor for NOx is take from Table 1.4-1 for Large Wall-Fired Boilers, Uncontrolled (Post-NSPS) of Source 1. The emission factors for other pollutants is taken from Table 1.4-2 of the same document. The emission factor for Total PM is used for PM10 and PM2.5 and the emission factor for VOCs is used for ROG. This results in a conservatively high estimate of emissions of PM10, PM2.5, and ROG.

2 Natural gas would only be used during start-ups under the direct combustion alternative. On these days, a full-load of biomass fuel would not be combusted. Thus, the maximum daily operational emissions of the plant would not occur on start-up days.

3 Table C-1 to Subpart C - Default CO2 Emission Factors and High Heating Values for Various Types of Fuel from 40 CFR 98 (page 789-790), which is Source 4, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), which is Source 3.

4 Table C-2 to Subpart C - Default CH4 and N2O Emission Factors for Various Types of Fuel from from 40 CFR 98 (page 790-791), which is Source 4, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), whch is Source 3.

5 Equations C-2a and C-8 from 40 CFR 98 (page 728), which is Source 2, as required by ARB's Regulation for the Mandatory Reporting of GHGs (ARB 2012), which is Source 3.

GHGs Associated with Electricity Consumption by the Plant

The net amount of electricity the plant would export to the grid would be less than 2.0 MW. This is because the plant would need some electricity to power some of its own equipment—this is referred to as the parasitic load. It is conservatively assumed that the plant would purchase non-renewable power from the grid because this would be more cost effective than using its own power that sells at the premium renewable rate.

	Gasification and Direct Combustion Alternatives	units	source
size of proposed plant	2.0	MW	wksht: Operational Parameters
parasitic load, maximum, percentage	10%	%	Source 1
electricity imported from grid	0.2	MW	calculation
operation time of the plant	365	days/year	wksht: Operational Parameters
time conversion rate	24	hr/day	wksht: Unit Conversions
Electricity imported from grid, annually	1,752	MW-hr/year	calculation
GHG Emission Factors			
CO2	1,422.78	lb/MW-hr	Source 2; See Note 1
CH4	0.029	lb/MW-hr	Source 2; See Note 1
N2O	0.011	lb/MW-hr	Source 2; See Note 1
Global warming potential			
CH4	21	unitless	wksht: Unit Conversions
N2O	310	unitless	wksht: Unit Conversions
CO2-e emission factor	1,426.80	lb/MW-hr	composite calculation
Electricity imported from grid, annually	1,752.0	MW-hr/year	above calculation
CO2-e emissions, annual	2,499,752	lb/year	calculation
mass conversion rate	2,204.62	lb/MT	wksht: Unit Conversions
CO2-e emissions, annual	1,134	MT/year	conversion calculation

Notes

1 These are the default GHG emission rates associated with the consumption of electricity produced by Sierra Pacific Power Company, as provided by CalEEMod (listed below). Sierra Pacific's generation and distribution assets are now owned and operated by California Pacific Electric Company (CalPeco).

Sources

1 Tornatore, Fred. Chief Technical Officer. TSS Consultants, Rancho Cordova, CA. May 8, 2012—telephone conversation with Austin Kerr of Ascent Environmental regarding the quantity of electricity that the biomass plant would consume.

2 South Coast Air Quality Management District. 2011. California Emissions Estimator Model (CalEEMod) Version 2011.1.1. Available: <http://www.caleemod.com/>.

Emissions from Employee Commute Trips under the Gasification Alternatives

	gasification alternatives	units	source
Daily VMT by Employees working at the Biomass Plant - Summer and Winter			
Employees at Plant, daily	5	#/day	See Note 2
trip generation rate			
commute	2	trips/employee	Table 8-9 of Section 8, Traffic and Transportation
lunch time trip	2	trips/employee	Table 8-9 of Section 8, Traffic and Transportation
total	4	trips/employee	Table 8-5 of Section 8, Traffic and Transportation
trips	20	trips/day	calculation
commute	10	trips/day	calculation
lunch time trip	10	trips/day	calculation
total	20	trips/day	summation
average trip length			
commute trip, max.			
out-of-Basin	8.2	miles/trip	Table 8-9 of Section 8, Traffic and Transportation
in LTAB	3.5	miles/trip	Table 8-9 of Section 8, Traffic and Transportation
lunch time trip			
out-of-Basin	0.0	miles/trip	Table 8-9 of Section 8, Traffic and Transportation
in LTAB	4.5	miles/trip	Table 8-9 of Section 8, Traffic and Transportation
VMT, daily			
out-of-Basin	82	VMT/day	calculation
in LTAB	80	VMT/day	calculation
total	162	VMT/day	summation
Daily VMT by Employees refining biomass in the forests - Summer Only			
Additional employees	8	#/day	assumption
trip generation rate	4	trips/employee	same as Table 8-5 of Section 8, Traffic and Transportation
trips	32	trips/day	calculation
average trip length			
out-of-Basin	8.2	miles/trip	same trip length used for plant employees
in LTAB	3.5	miles/trip	same trip length used for plant employees
VMT, daily			
out-of-Basin	262	VMT/day	calculation
in LTAB	112	VMT/day	calculation
total	374	VMT/day	summation
Combined VMT by Employees, Maximum Daily			
Daily, in Summer			
out-of-Basin	344	VMT/day	summation
in LTAB	192	VMT/day	summation
total	536	VMT/day	summation; See Note 4
Daily, in Winter			
out-of-Basin	82	VMT/day	summation
in LTAB	80	VMT/day	summation
total	162	VMT/day	summation
Seasonality of Operations			
Employees at Biomass Plant - Summer and Winter	365	days/year	wksht: Operational Parameters
Employees refining biomass in the forests - Summer Only	120	days/year	See Note 1
Combined VMT by Employees, Annual			
total	104,058	VMT/year	summation; See Note 5

Mix of passenger vehicles used in employee commutes

passenger car population in Placer County portion of LTAB	value	units	source
light duty autos - gasoline	3,330	#	wksht: On-Rd Veh Emiss Rates; See Notes 3 and 6
light duty trucks 1 - gasoline	1,034	#	wksht: On-Rd Veh Emiss Rates; See Notes 3 and 6
light duty trucks 2 - gasoline	2,391	#	wksht: On-Rd Veh Emiss Rates; See Notes 3 and 6
Total, all passenger vehicle types - gasoline	6,756	#	summation
relative portion of passenger car population by vehicle type			
light duty autos - gasoline	49%	%	calculation
light duty trucks 1 - gasoline	15%	%	calculation
light duty trucks 2 - gasoline	35%	%	calculation

Emission Rates (running exhaust)	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
Expressed in grams per mile								
light duty autos	0.152	0.181	0.003	0.002	2.347	285.548	g/mile	wksht: On-Rd Veh Emiss Rates; See Note 6
light duty trucks 1	0.364	0.479	0.006	0.005	5.502	337.136	g/mile	wksht: On-Rd Veh Emiss Rates; See Note 6
light duty trucks 2	0.256	0.196	0.002	0.002	2.090	409.022	g/mile	wksht: On-Rd Veh Emiss Rates; See Note 6
* The emission rates for CO2 take into account the requirements of Pavley 1 and the Low Carbon Fuel Standard.								
					Mass Conversion Rate	453.59	g/lb	wksht: Unit Conversions
Expressed in pounds per mile								
light duty autos	0.0003	0.0004	0.00001	0.00001	0.0052	0.6295	lb/mile	conversion calculation
light duty trucks 1	0.0008	0.0011	0.00001	0.00001	0.0121	0.7433	lb/mile	conversion calculation
light duty trucks 2	0.0006	0.0004	0.00001	0.000005	0.0046	0.9017	lb/mile	conversion calculation
Composite emission rates for all passenger vehicle types	0.0005	0.0005	0.0000	0.0000	0.0060	0.7433	lb/mile	weighted average calculation

Maximum Daily Emissions from Employee Commute Trips

	NOx	ROG	PM10	PM2.5	CO	units	source
Total (both in an out of LTAB)	0.26	0.27	0.00	0.00	3.24	lb/day	summation

Annual GHG Emissions from Employee Commute Trips	CO2e	units	source
GHG emissions	77,345	lb/year	CalEEMod runs "employee trips in Placer-Nevada" and "employee trips in LTAB"
mass conversion rate	2,205	lb/MT	wksht: Unit Conversions
GHG emissions	35	MT/year	conversion calculation

Notes

1 In order to estimate the highest potential number of dialy trips, the traffic analysis assumed that most collection and delivery of biomass fuel would occur during a minimum 167-day period between May 1 and October 15 each year, which is the same as the grading season designated by TRPA (TRPA Code of Ordinances, Section 64.2.A, Grading Season), and that fuel collection and hauling would occur 5 days per week. This comes to 120 days, a value that was also used in the traffic analysis (See p. 8-13 of Section 8, Traffic and Transportation).

2 It is assumed that three employees would work during the day shift and only one employee would work during the other two 8-hour shifts.

3 It is assumed that all employee commuter trips are in light duty autos or light duty trucks (i.e., LDA, LDT1, LDT2) and that none of these vehicles are diesel, which have slightly lower emission rates. The EMFAC2011 model run indicates that only a small portion of these vehicle types are diesel-powered.

4 This maximum daily VMT value is used to estimate maximum daily emissions because forest crews are not working during the winter season.

5 This annual VMT value is used to estimate annual emissions.

6 It is assumed that proportions of various vehicle types (i.e., the fleet mix) in the portion of Placer County that is also part of the Lake Tahoe Air Basin are also representative of the fleet mix in other mountainous areas of Placer County and Nevada County, as well other areas in the Lake Tahoe Air Basin that are outside of Placer County.

Emissions from Employee Commute Trips under the Direct Combustion Alternative

Under the direct combustion alternative, more biomass would be consumed by the plant. While the number of employees working at the plant would be the same, more employees would be needed to process and haul biomass from the source. Thus, the amount of VMT associated with these employees is extrapolated based on the relative increase in biomass fuel that would be consumed by the plant.

	value	units	source
Max. mass of biomass consumed by the gassification alts	17,000	bdt/year	wksht: Operational Parameters
Max. mass of biomass consumed by the direct combustion alt	20,000	bdt/year	wksht: Operational Parameters
Ratio of biomass used by the direct combustion to gassification	1.18	ratio	calculation
Daily VMT by Employees working at the Biomass Plant - Summer and Winter			
out-of-Basin	82	VMT/day	same as for gasification above
in LTAB	80	VMT/day	same as for gasification above
total	162	VMT/day	same as for gasification above
Daily VMT by Employees refining biomass in the forests - Summer Only			
out-of-Basin	309	VMT/day	extrapolation calculation
in LTAB	132	VMT/day	extrapolation calculation
total	440	VMT/day	summation
Combined Daily VMT by Employees, Maximum Daily			
Daily, in Summer			
out-of-Basin	391	VMT/day	summation
in LTAB	212	VMT/day	summation
total	602	VMT/day	summation
Daily, in Winter			
out-of-Basin	82	VMT/day	same as for gasification above
in LTAB	80	VMT/day	same as for gasification above
total	162	VMT/day	summation
Combined VMT by Employees, Annual			
out-of-Basin	76,815	VMT/year	calculation
in LTAB	54,612	VMT/year	calculation
total	131,426	VMT/year	summation; See Note 5

The ratio of VMT associated with employee commute trips under the direct combustion alternative and the gasification alternative is used to estimate associated emissions.

Maximum Daily Emissions from Employee Commute Trips	NOx	ROG	PM10	PM2.5	CO	units	source
Total (both in an out of LTAB)	0.29	0.31	0.004	0.004	3.64	lb/day	summation

	value	units	source
Ratio of Annual VMT under direct combustion vs. gasification	1.26	ratio	calculation

	CO2e	units	source
Annual GHG Emissions from Employee Commute Trips	44	MT/year	calculation

On-Site Truck Emissions at the Plant Site

Truck Fleet Mix by Proportion of Biomass Hauled	value	units	source
chip vans	50%	%	Estimate based on project description
medium trucks	25%	%	Estimate based on project description
small trucks	25%	%	Estimate based on project description
Haul capacity	12.5	bdt/load	wksht: Operational Parameters
Work days per collection season, minimum	120	days/season	wksht: Employee Commute; See Note 1
Vehicle Category in EMFAC2011 for chip vans	T6 instate construction heavy	class	wksht: THB Exhaust
Vehicle Class in EMFAC2011 model	T6	type	Source 1
Time spent idling or maneuvering at plant	0.167	hr/delivery	assumption (10 minutes)

	gasification alternatives	direct combustion alternative	units	source
mass of biomass consumed by power plant, max.	17,000	20,000	bdt/year	wksht: Operational Parameters
Number of biomass deliveries, annual	1,360	1,600	visits/year	calculation
Max. daily biomass truck deliveries to plant	13	14	visits/day	calculation, rounded up; See Note 2
Number of biochar/ash departures, annual	213	185	visits/day	wksht: Trucks Hauling Biochar-Ash
Max. daily biochar/ash departures	1	1	visits/day	wksht: Trucks Hauling Biochar-Ash
Total Annual truck visits	1,573	1,785	visits/year	summation
Total Max. Daily truck visits	14	15	visits/day	summation

	NOx	ROG	PM10	PM2.5	CO	CO2**	units	source
Idling Emission Factors								
chip vans hauling biomass	138.2	2.3	1.0	0.9	35.0	6836.5	g/veh-hr	row 4846 of Idle_ER_Other_Area of Source 1

* It is assumed that idling emission rates for the smaller trucks that would haul away biochar/ash would be the same as, if not less than, the idling emission rates for the chip vans that would deliver biomass.

** The emission rates for CO2 take into account the requirements of Pavley 1 and the Low Carbon Fuel Standard.

	value	units	source
mass conversion rate	453.59	g/lb	wksht: Unit Conversions
mass conversion rate	1,000,000	g/MT	wksht: Unit Conversions

Emissions	NOx	ROG	PM10	PM2.5	CO	CO2	units	source
Gasification Alternatives								
Maximum Daily Emissions	0.7	0.01	0.01	0.005	0.2	32.7	lb/day	calculation w/conversions
Annual Emissions						1.8	MT/year	calculation w/conversions
Direct Combustion Alternative								
Maximum Daily Emissions	0.8	0.01	0.01	0.005	0.2	35.2	lb/day	calculation w/conversions
Annual Emissions						2.0	MT/year	summation

Notes

1 In order to estimate the highest potential number of dialy trips, the traffic analysis assumed that most collection and delivery of biomass fuel would occur during a minimum 167-day period between May 1 and October 15 each year, which is the same as the grading season designated by TRPA (TRPA Code of Ordinances, Section 64.2.A, Grading Season), and that fuel collection and hauling would occur 5 days per week. This comes to 120 days, a value that was also used in the traffic analysis (See p. 8-13 of Section 8, Traffic and Transportation).

Sources

1 California Air Resources Board. 2012 (February 8) (last updated). EMFAC2011 Idling Emission Rates. Available: <http://www.arb.ca.gov/msei/modeling.htm>. Accessed May 3, 2012.

Truck Hauling Biomass

Maximum Daily Emissions

	NOx	ROG	PM10	PM2.5	CO	CO2	units	source
Gasification Alternatives								
Exhaust Emissions	9.5	0.2	0.1	0.1	0.8	1,549	lb/day	wksht: THB Exhaust
Fugitive Dust Emissions	—	—	13.3	1.3	—	—	lb/day	wksht: THB Dust
Total	9.5	0.2	13.4	1.4	0.8	1,549	lb/day	summation
Direct Combustion Alternative								
Exhaust Emissions	11.1	0.3	0.1	0.1	0.9	1,823	lb/day	wksht: THB Exhaust
Fugitive Dust Emissions	—	—	15.6	1.6	—	—	lb/day	wksht: THB Dust
Total	11.1	0.3	15.8	1.7	0.9	1,823	lb/day	summation

Annual GHG Emissions

	CO2	units	source
Gasification Alternatives	84	MT/year	wksht: THB Exhaust
Direct Combustion Alternative	99	MT/year	extrapolation calc

Emissions from Trucks Hauling Biomass under Gasification Alternatives

Vehicle Miles Traveled by Trucks Hauling Biomass under the Gasification Alternatives

	gasification alternatives	units	source
Max. fuel demand of power plant, annual range	17,000	BDT	wksht: Operational Parameters
If all Biomass would be hauled in Chip Vans			
Daily Fuel Deliveries			
Capacity of chip van truck, dry, by mass	12.5	bdt/load	wksht: Operational Parameters
Loads of biomass to be hauled, annually	1,360	loads/year	calculation, and Table 6-7 of Sec 6, Traffic and Transportation
Number of trips, max. daily - Summer Only	22	trips/day	Table 8-8 of Section 8, Traffic and Transportation
Moisture content of biomass	50%	%	wksht: Operational Parameters
Capacity of chip van truck, green	18.75	grn tons/load	calculation
mass conversion rate	2,000	lb/ton	wksht: Unit Conversions
Capacity of chip van truck, green	37,500	grn lb/load	conversion calculation
Vehicle Category in EMFAC2011 model	T6 instate construction heavy	class	EMFAC2011; See Note 1
VMT, daily			
out-of-LTAB	309	VMT/day	Table 8-9 of Section 6, Traffic and Transportation
in LTAB	292	VMT/day	Table 8-9 of Section 6, Traffic and Transportation
VMT, annual			
out-of-LTAB	37,080	VMT/year	Table 8-9 of Section 6, Traffic and Transportation
in LTAB	35,040	VMT/year	Table 8-9 of Section 6, Traffic and Transportation

EMISSIONS SUMMARY

Emission Rates (running exhaust)	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
Chip Vans, Expressed in grams per mile	7.145	0.187	0.088	0.081	0.587	1,169	g/mile	wksht: On-Rd Veh Emiss Rates
Chip Vans, Expressed in pounds per mile	0.0158	0.0004	0.0002	0.0002	0.0013	2.5777	lb/mile	conversion calculation

* The emission rates for CO2 take into account the requirements of Pavley 1 and the Low Carbon Fuel Standard.

Emissions, Maximum Daily	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
out-of-LTAB	4.9	0.1	0.1	0.1	0.4	796.5	lb/day	calculation
in LTAB	4.6	0.1	0.1	0.1	0.4	752.7	lb/day	calculation
Total	9.5	0.2	0.1	0.1	0.8	1,549.2	lb/day	summation
Emissions, Annual								
out-of-LTAB	584.1	15.3	7.2	6.6	48.0	95,581	lb/year	calculation
in LTAB	551.9	14.4	6.8	6.2	45.3	90,323	lb/year	calculation
Total	1,136.0	29.7	13.9	12.8	93.3	185,904	lb/year	summation
					Annual Total - expressed in MT/year	84.3	MT/year	conversion calculation

	value	units	source
Mass Conversion Rate	453.59	g/lb	wksht: Unit Conversions
Mass Conversion Rate	2,204.62	lb/MT	wksht: Unit Conversions

Emissions from Trucks Hauling Biomass under Direct Combustion Alternatives

Emissions generated by trucks hauling biomass under the direct combustion alternatives are estimated based proportionally according to the ratio of the annual mass biomass consumed by the two different technologies.

Ratio of Biomass Consumption by Direct Combustion vs. Gasification	value	units	source
Max. fuel demand of power plant, annually, under the direct combustion alternative	20,000	BDT/year	wksht: 3-Operational Parameters
Max. fuel demand of power plant, annually, under the gasification alternatives	17,000	BDT/year	wksht: 3-Operational Parameters
Ratio of biomass fuel consumption	1.18	ratio	calculation

	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
Emissions, Maximum Daily								
out-of-LTAB	5.7	0.1	0.1	0.1	0.5	937	lb/day	calculation
in LTAB	5.4	0.1	0.1	0.1	0.4	886	lb/day	calculation
Total	11.1	0.3	0.1	0.1	0.9	1,822.6	lb/day	summation
Emissions, Annual								
out-of-LTAB	687.1	18.0	8.4	7.8	56.5	112,449	lb/year	calculation
in LTAB	649.3	17.0	8.0	7.3	53.3	106,262	lb/year	calculation
Total	1,336.4	35.0	16.4	15.1	109.8	218,711	lb/year	summation
					Annual Total - expressed in MT/year	99.2	MT/year	conversion calculation

Notes

1 The vehicle class in EMFAC2011 is based on the mass of the haul load.

Dust Emissions from Trucks Hauling Biomass on Unpaved Roadways under the Gasification Alternatives

Emission Factor (EF) Calculation for Truck Travel on Unpaved Roads

	value	units	source
Truck Type	T6 heavy	NA	EMFAC2011
green biomass capacity	37,500	lb/load	wksht: THB Exhaust
truck curb weight	10,000	lb	EMFAC2011
truck total weight	47,500	lb	summation
mass conversion rate	2,000	lb/ton	wksht: Unit Conversions
truck total weight	23.75	tons/truck	conversion calculation

Emission Factor Calculation (Based on formula 1a in AP-42 Section 13.2.2., EPA 2006)

Variables	PM10 EF Calc	PM2.5 EF Calc	Unit	Source
a	0.9	0.9	constant	Source 1, Table 13.2.2-2 Constants for Equations 1a and 1b AP-42 Section 13.2.2
b	0.45	0.45	constant	
k	1.5	0.15	constant (lbs/VMT)	
s	8.5%	8.5%	surface material silt content (%)	CalEEMod
W	23.75	23.75	mean vehicle weight (tons)	Calc'ed above based on truck size anticipated for project
Emission Factor	0.044	0.0044	lbs/VMT	

Maximum Daily VMT by Surface Type

	in LTAB	out of LTAB	Total	units	source
Max Daily VMT	309	292	601	VMT/day	wksht: Truck Hauling Biomass
Portion of truck travel on unpaved roads			50%	%	assumption
Truck VMT on unpaved roads, max. daily			301	VMT/day	calculation

	PM10	PM2.5	units	source
Fugitive Dust Emissions	13.3	1.3	lb/day	calc using emission factor

Dust Emissions from Trucks Hauling Biomass under Direct Combustion Alternatives

Emissions generated by trucks hauling biomass under the direct combustion alternatives are estimated based proportionally according to the ratio of the annual mass biomass consumed by the two different technologies.

Ratio of Biomass Consumption by Direct Combustion vs. Gasification

	value	units	source
Max. fuel demand of power plant, annually, under the direct combustion alternative	20,000	BDT/year	wksht: 3-Operational Parameters
Max. fuel demand of power plant, annually, under the gasification alternatives	17,000	BDT/year	wksht: 3-Operational Parameters
Ratio of biomass fuel consumption	1.18	ratio	calculation

	PM10	PM2.5	units	source
Fugitive Dust Emissions	15.6	1.6	lb/day	calc using emission factor

Sources

1 U.S. Environmental Protection Agency 2006 (November). Emission Factors & AP 42, Compilation of Air Pollutant Emission Factors-Section 13.2.2 Unpaved Roads. Available http://www.epa.gov/ttnchie1/ap42/. Accessed May 5, 2012

Emissions from Trucks Hauling Biochar/Ash

Vehicle Miles Traveled

	gasification alternatives	direct combustion alternative	units	source
type of waste product produced	biochar	ash	none	wksht: 3-Operational Parameters
maximum mass of biochar/ash per year	850	1,000	tons/year	Project Description, pg. 3-13
mass conversion rate	2,000	2,000	lb/ton	wksht: Unit Conversions
maximum mass of biochar/ash per year	1,700,000	2,000,000	lb/year	conversion calculation
density of biochar/ash	800	1,080	lb/cu. Yd.	Project Description, pg. 3-13
minimum truck capacity, by volume	10	10	cu. Yd/load	Project Description, pg. 3-13
minimum truck capacity, by mass	8,000	10,800	lb/load	calculation
Truck Class in EMFAC2011 model	LHD2	LHD2	class	EMFAC2011; See Note 1
truck loads hauled annually	213	185	trips/year	calculation
maximum loads hauled daily	1	1	trips/day	See Note 2
Trip length in	53.4	53.4	miles/trip	See Note 3
Trip length out	53.5	53.5	miles/trip	See Note 3
Combined round trip length	106.9	106.9	miles/trip	summation
Maximum daily VMT by biochar haul truck	106.9	106.9	VMT/day	calculation
Maximum Annual VMT	22,716	19,796	trips/year	calculation; See Note 4.

Emission Rates (running exhaust)	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
Expressed in grams per mile (LHD2)	3.893	0.202	0.046	0.043	1.060	510	g/mile	wksht: On-Rd Veh Emiss Rates
* The emission rates for CO2 take into account the requirements of Pavley 1 and the Low Carbon Fuel Standard.								
			Mass Conversion Rate			453.59	g/lb	wksht: Unit Conversions
Expressed in pounds per mile	0.0086	0.0004	0.0001	0.0001	0.0023	1.13	lb/mile	conversion calculation

Emissions	NOx	ROG	PM10	PM2.5	CO	CO2*	units	source
Gasification Alternatives								
Maximum Daily Emissions	0.92	0.05	0.01	0.01	0.25	120	lb/day	calculation
Annual Emissions	195.0	10.1	2.3	2.1	53.1	25,565	lb/year	calculation
			Annual Total - expressed in MT/year			11.6	MT/year	
Direct Combustion Alternative								
Maximum Daily Emissions	0.92	0.05	0.01	0.01	0.25	120	lb/day	calculation
Annual Emissions	169.9	8.8	2.0	1.9	46.2	22,279	lb/year	calculation
			Annual Total - expressed in MT/year			10.1	MT/year	

	rate	units	source
Mass Conversion Rate	2,204.62	lb/MT	wksht: Unit Conversions

Notes

1 The vehicle class in EMFAC2011 is based on the mass of the haul load.

2 It is assumed that no more than one load of biochar or ash would be hauled per day given that the project description states that biochar/ash would be hauled away once or twice each week.

3 Trip lengths for the hauling of biochar are provided in Table 6-9 of Section 6, Traffic and Transportation and assume that biochar would be hauled to the Lockwood Regional Landfill southeast of Sparks, NV via Interstate 80. Thus, all related VMT would be outside of the Lake Tahoe Air Basin. Information about the landfill is available at http://ndep.nv.gov/bwm/landfill_lockwood.htm. However, the biochar and ash may be may also be used as a soil amendment in landscaping or agriculture.

4 A conservative (i.e., higher) estimate for annual VMT associated with the hauling of biochar under the gasification alternatives is provided in Table 6-9 of Section 6, Traffic and Transportation. The traffic analysis estimated 12,840 VMT during the summer period and 12,305 during the winter period, which comes to a total of 25,145 VMT/year. An estimate of VMT associated with the hauling of ash under the direct combustion alternative could be estimated based on the ration of truck loads needed for ash vs. biochar, resulting in 21,913 VMT/year. The estimates for annual VMT provided above are used because they are directly based on the maximum volume of biochar or ash that would be generated under the respective alternatives, as provided on p. 3-13 of the project description.

5 Because biochar is highly resistant to decomposition (See p. 2.42 of IPCC 2006), therefore the biochar sent to Lockwood Regional Landfill is not anticipated to generate subsequent GHGs.

Running Exhaust Emission Rates for On-Road Vehicles

Source: These emission rates were provided by the California Air Resources Board's Mobile Source Emissions Inventory (EMFAC2011), which is available at http://www.arb.ca.gov/msei/modeling.htm.

It is assumed that emission rates for vehicles in the portion of Placer County that is also part of the Lake Tahoe Air Basin are also representative of emission rates in other mountainous areas of Placer County and Nevada County, as well other areas in the Lake Tahoe Air Basin that are outside of Placer County.

EMFAC 2011
2015 Estimated Annual Emission Rates
EMFAC 2011 Vehicle Categories
Placer COUNTY
Lake Tahoe AIR BASIN
Placer County APCD

Area	CalYr	Season	Veh	Fuel	Population	Speed	ROG_RUNEX	ROG_IDLEX	ROG_RUNLS	CO_RUNEX	CO_IDLEX	NOX_RUNEX	NOX_IDLEX	CO2_RUNEX (Pavley I+LCFS)	CO2_IDLEX(Pavley I+LCFS)	PM10_RUNEX	PM10_IDLEX	PM2_5_RUNEX	PM2_5_IDLEX	SOX_RUNEX	SOX_IDLEX
						(Miles/hr)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)
Placer (LT)	2015	Annual	LDA	GAS	3,330	AllSpeeds	0.081	0	0.100	2.347	0	0.152	0	285.548	0	0.003	0	0.002	0	0.003	0
Placer (LT)	2015	Annual	LDA	DSL	24	AllSpeeds	0.055	0	0	0.239	0	0.489	0	259.166	0	0.042	0	0.038	0	0.003	0
Placer (LT)	2015	Annual	LDT1	GAS	1,034	AllSpeeds	0.175	0	0.304	5.502	0	0.364	0	337.136	0	0.006	0	0.005	0	0.004	0
Placer (LT)	2015	Annual	LDT1	DSL	0	AllSpeeds	0.075	0	0	0.267	0	0.576	0	265.727	0	0.062	0	0.057	0	0.003	0
Placer (LT)	2015	Annual	LDT2	GAS	2,391	AllSpeeds	0.053	0	0.143	2.090	0	0.256	0	409.022	0	0.002	0	0.002	0	0.005	0
Placer (LT)	2015	Annual	LDT2	DSL	0	AllSpeeds	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Placer (LT)	2015	Annual	T6 instate construction heavy	DSL	36	AllSpeeds	0.18690843	0.146263997	0	0.587	1.535	7.145	9.017	1169.227	701.333	0.088	0.027	0.081	0.025	0.011	0.007
Placer (LT)	2015	Annual	LHD2	DSL	12	AllSpeeds	0.20164507	0.109759108	0	1.060	0.910	3.893	2.596	510.485	138.210	0.046	0.029	0.043	0.027	0.005	0.001

EMFAC2011 only provides idling exhaust emissions for large trucks, but no passenger vehicles.

EMFAC 2011
2015 Estimated Annual Emission Rates
EMFAC 2011 Vehicle Categories
Placer COUNTY
Lake Tahoe AIR BASIN
Placer County APCD

Area	CalYr	Season	Veh	Fuel	MdlYr	Speed	Pop	VMT	Trips	ROG_RUNEX	ROG_IDLEX	ROG_STREX	ROG_DIURN	ROG_HTSK	ROG_RUNLS	ROG_RESTL
						(Miles/hr)	(Vehicles)	(Miles/day)	(Trips/day)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)
Placer (LT)	2015	Annual	LDA	GAS	AllMYr	AllSpeeds	3330.483	110294.5	20755.42	0.080560553	0	2.539139699	0.226505178	1.150414816	0.100044872	0.124522259
Placer (LT)	2015	Annual	LDA	DSL	AllMYr	AllSpeeds	24.47696	758.0702	142.3505	0.054870937	0	0	0	0	0	0
Placer (LT)	2015	Annual	LDT1	GAS	AllMYr	AllSpeeds	1034.443	32579.83	6164.704	0.174985174	0	5.219016393	0.541994862	2.264322031	0.303724386	0.278301764
Placer (LT)	2015	Annual	LDT1	DSL	AllMYr	AllSpeeds	7.73E-04	0.022656	0.004092	0.075126906	0	0	0	0	0	0
Placer (LT)	2015	Annual	LDT2	GAS	AllMYr	AllSpeeds	2391.427	83204.37	14975.22	0.053370698	0	3.066381419	0.230477669	1.166504241	0.1426122	0.124225209
Placer (LT)	2015	Annual	LDT2	DSL	AllMYr	AllSpeeds	0	0	0	0	0	0	0	0	0	0
Placer (LT)	2015	Annual	LHD2	GAS	AllMYr	AllSpeeds	36.12349	1308.196	538.1861	0.263630098	0.538890089	13.67830471	0.026193781	1.747550291	0.378340063	0.010969868
Placer (LT)	2015	Annual	LHD2	DSL	AllMYr	AllSpeeds	95.30488	3459.012	1198.815	0.201645067	0.109759108	0	0	0	0	0
Placer (LT)	2015	Annual	T6 instate c	DSL	AllMYr	AllSpeeds	12.27172	617.6325	0	0.186908426	0.146263997	0	0	0	0	0
Placer (LT)	2015	Annual	T6 instate c	DSL	AllMYr	AllSpeeds	23.57266	1516.642	0	0.256977925	0.230147582	0	0	0	0	0

Area	TOG_RUNE	TOG_IDLEX	TOG_STRE)	TOG_DIURI	TOG_HTSK	TOG_RUNL	TOG_RESTI	CO_RUNEX	CO_IDLEX	CO_STREX	NOX_RUNEX	NOX_IDLEX	NOX_STREX
	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)
Placer (LT)	0.098643	0	2.716162	0.226505	1.150415	0.100045	0.124522	2.346525	0	29.40047	0.15222997	0	1.407328955
Placer (LT)	0.062467	0	0	0	0	0	0	0.238695	0	0	0.489383034	0	0
Placer (LT)	0.209604	0	5.58044	0.541995	2.264322	0.303724	0.278302	5.501677	0	66.56665	0.364069887	0	2.406068381
Placer (LT)	0.085527	0	0	0	0	0	0	0.267248	0	0	0.576433004	0	0
Placer (LT)	0.073776	0	3.275674	0.230478	1.166504	0.142612	0.124225	2.089983	0	38.78615	0.255690542	0	2.658779704
Placer (LT)	0	0	0	0	0	0	0	0	0	0	0	0	0
Placer (LT)	0.300084	0.570003	14.63195	0.026194	1.74755	0.37834	0.01097	5.760854	3.29732	199.6931	0.466928213	0.034377786	26.55947231
Placer (LT)	0.229559	0.124953	0	0	0	0	0	1.059623	0.90974	0	3.893080766	2.596080142	0
Placer (LT)	0.212781	0.16651	0	0	0	0	0	0.586963	1.535372	0	7.144611728	9.017227945	0
Placer (LT)	0.29255	0.262006	0	0	0	0	0	0.839255	2.598325	0	3.9038983	7.051017965	0

Area	CO2_RUNE	CO2_IDLEX	CO2_STREX	CO2_RUNE	CO2_IDLEX	CO2_STREX	PM10_RUN	PM10_IDLE	PM10_STRI	PM10_PMT	PM10_PMI	PM2_5_RU	PM2_5_IDI	PM2_5_STI	PM2_5_PM	PM2_5_PM	SOX_RUNE	SOX_IDLEX	SOX_STREX
	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/mile)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)	(gms/mile)	(gms/mile)	(gms/mile)	(gms/vehicle/day)	(gms/vehicle/day)
Placer (LT)	339.7213	0	463.5827	285.5477	0	402.9852	0.002716	0	0.026371	0.008	0.03675	0.002463	0	0.023908	0.002	0.01575	0.003431	0	0.005169
Placer (LT)	308.6688	0	0	259.1663	0	0	0.041712	0	0	0.008	0.03675	0.038375	0	0	0.002	0.01575	0.002947	0	0
Placer (LT)	392.8097	0	511.6937	337.1357	0	456.0379	0.00552	0	0.045136	0.008	0.03675	0.005025	0	0.040983	0.002	0.01575	0.004013	0	0.006318
Placer (LT)	319.8629	0	0	265.7267	0	0	0.062146	0	0	0.008	0.03675	0.057174	0	0	0.002	0.01575	0.003054	0	0
Placer (LT)	462.1309	0	627.7749	409.0215	0	568.9758	0.002331	0	0.021371	0.008	0.03675	0.002145	0	0.019647	0.002	0.01575	0.004648	0	0.006972
Placer (LT)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Placer (LT)	969.216	116.3645	884.2904	944.9856	113.4553	862.1832	0.003487	0	0.035344	0.008	0.03675	0.003162	0	0.031131	0.002	0.01575	0.009774	0.00123	0.012394
Placer (LT)	523.5747	141.7534	0	510.4853	138.2096	0	0.04627	0.028982	0	0.012	0.08918	0.042568	0.026663	0	0.003	0.03822	0.004998	0.001353	0
Placer (LT)	1199.207	719.3155	0	1169.227	701.3326	0	0.087722	0.026971	0	0.012	0.130339	0.080704	0.024813	0	0.003	0.05586	0.011441	0.006863	0
Placer (LT)	1188.223	734.7692	0	1158.517	716.4	0	0.139343	0.040651	0	0.012	0.130339	0.128196	0.037399	0	0.003	0.05586	0.011336	0.00701	0

Exhaust Emissions of Loader at Plant and Fuel Storage Area

Equipment Operation Parameters	gasification and direct combustion alternatives	units	source
number of loaders	1	#	assumption
hours of operation	8	hours/day	conservative assumption; See Note 3
days per week	7	days/week	conservative assumption
operational frequency, annual basis	365	days/year	wksht: Operational Parameters
year of fleet	January 2015	year on calendar	wksht: Operational Parameters
hours operated annually	2,920	hours/year	calculation

	NO_x	ROG	PM_{10}	$PM_{2.5}$	CO	CO_2	CH_4	units	sources/notes
Emission factors for Rubber Tired Loader	1.10	0.12	0.04	0.04	0.35	148.84	0.01	lb/hr	wksht: Offroad emfacs
Maximum Daily Emissions	8.79	0.95	0.30	0.30	2.84	1190.74	0.09	lb/day	calculation

	CO_2	CH_4	units	sources/notes
Annual GHG Emissions	434,621	31	lb/year	calculation; See Note 2
global warming potential	1	21	unitless	wksht: Unit Conversions

	value	units	source
Annual CO2-e emissions	435,278	lb/year	calculation
mass conversion rate	2,204.62	lb/MT	wksht: Unit Conversions
Annual CO2-e emissions	197	MT/year	conversion calculation

Notes

1. Estimates do not include any fugitive PM dust emissions generated by operation of the loader. This is because the loader would not engage in earth disturbance and would operate in areas that consists of asphalt, compacted soils, and/or gravel.
2. According to the model run in OFFROAD2007, the loader would not generate any emissions of N2O.
3. Emissions from loaders in future years would be lower due to anticipated improvements in fuel efficiency and emissions controls.
4. It is conservatively assumed that the loader would be operated 8 hours per day, 7 days per year, all year. However, it is more likely that it would be operated no more than 4 hours per day, 5 days per week.

Off-Road Equipment Use for Chipping of Forest-Source Biomass

Data in the following shaded table was provided in Appendix C to the following study: Sierra Nevada Conservancy. 2008 (November 17). *Forest Biomass Removal on National Forest Lands: First Progress Report* . Prepared by Placer County Chief Executive Office and TSS Consultants.

Date	Chip Van Loads	Chip Production		Equipment Operation hours	Total Diesel Fuel Usage (gal)	Diesel Fuel Usage by Equipment Type (gal)			Green Tons/ operating hour
		BDT	green tons			Grinder	Loader	Excavator	
4/14/2008	2	23.5	43.3	2.5	93.5	75	6.5	12	17.3
4/15/2008	5	57.6	116.5	11.8	0				9.9
4/16/2008	6	80.1	148.7	2.8	105	84	7	14	53.1
4/17/2008	7	84.9	162.9	5.8	218	174	15	29	28.1
4/18/2008	1	12.9	23						
4/21/2008	1	13.7	24.5						
4/22/2008	5	73.2	124	1.8	68	54	5	9	68.9
4/23/2008	1	13	22.9	2.3	86	69	6	11	9.9
4/24/2008				3.3	124	99	9	16	0
4/25/2008	3	35.6	74.5						
4/28/2008	1	13.4	25.4						
5/5/2008	4	54.5	120.2	3.1	116	93	8	15	38.8
5/6/2008	8	123.6	195.8	2.2	83	66	6	11	89
5/7/2008	10	135.9	253.9	3.1	116	93	8	15	81.9
5/8/2008	8	103.4	211.6	2.6	98	78	7	13	81.4
5/9/2008	9	130	217.8	2.8	105	84	7	14	77.8
5/12/2008	7	107.7	166.5	3.5	132	105	9	18	47.6
5/13/2008	6	83.7	149.5	1.9	71	57	5	9	78.7
5/27/2008	4	71.1	102.4	1.7	63	51	4	8	60.2
5/28/2008	3	30.9	69.2	0.7	26	21	2	3	98.9
5/29/2008	8	118	213.4	4.8	180	144	12	24	44.4
5/30/2008	9	119.4	225.2	2	75	60	5	10	112.6
6/2/2008	8	102.9	185.3	3.1	116	93	8	15	59.8
6/3/2008	7	79	157.4	4.8	180	144	12	24	32.8
6/4/2008	1	11.7	21.3						
6/5/2008	8	112.5	192.4	4.5	169	135	12	22	42.8
6/6/2008	7	91.4	183.3	5.1	191	153	13	25	35.9
6/9/2008	11	131.8	244.5	5.5	206	165	14	27	44.4
6/10/2008	6	83.8	138	4.3	161	129	11	21	32.1
6/11/2008	10	152.6	234.4	5.2	196	156	14	26	45.1
6/12/2008	8	108.9	198.9	5.5	206	165	14	27	36.2
6/13/2008	7	110	170.4	4.7	176	141	12	23	36.2
6/16/2008	7	95.4	158.6	3.5	131	105	9	17	45.3
6/17/2008	7	96.3	156.8	5.4	203	162	14	27	29
6/18/2008	8	109.6	178.8	3.3	124	99	9	16	54.2
6/20/2008	5	73.7	104.1	6.3	236	189	16	31	16.5
6/23/2008	8	107.2	187.6	6.1	229	183	16	30	30.7
6/24/2008	2	29	47.2	3.4	128	102	9	17	13.9
6/25/2008	10	144.9	253.9	4.3	161	129	11	21	59.1
6/26/2008	8	116.1	196.4	5.6	211	168	15	28	35.1
6/27/2008	9	127.8	227.8	3.7	139	111	10	18	61.6
7/9/2008	5	62.9	91.4	5	188	150	13	25	18.3
7/10/2008	8	114.8	169.6	4.9	184	147	13	24	34.6
7/11/2008	9	124.6	208.8	4.9	184	147	13	24	42.6
7/16/2008	7	116	152	5.5	206	165	14	27	27.6
7/17/2008	8	133.8	180.7	5.8	218	174	15	29	31.2
7/18/2008	5	78	107.2	4.5	169	135	12	22	23.8
7/21/2008	2	29.8	45.4						
7/22/2008	4	76.9	99.6	2.1	78	63	5	10	47.4
7/23/2008	4	84.2	106.2	2.5	93	75	6	12	42.5

The following metrics provide the amount of commitment use per green ton of biomass recovered.

	Chip Van Loads		Equipment Operation		Diesel Fuel Usage byEquipment Type (gal)		
	BDT	green tons	hours	Diesel Fuel Usag	Grinder	Loader	Excavator
Total	3,833	6,423	148	5,548	4,437	383	728
Rate			0.02	0.86	0.69	0.06	0.11
(units)			hr/green ton	gal/grn ton	gal/grn ton	gal/grn ton	gal/grn ton

	gasification alternatives	direct combustion alternative	units	source
mass of biomass recovered	34,000	40,000	grn ton/year	wksht: Operational Parameters
equipment-hours per season that chipping would occur	783	921	hr/season	calculation using above operation rate
work days during chipping season, minimum	120	120	days/season	See Note 2
max. equipment-hours per day operated	6.5	7.7	hr/day	See Note 2
days per season that chipping would occur	120	120	days/season	calculation for crosscheck, See Note 2

Emission factors from offroad equip. used in chipping	NO_x	ROG	PM_{10}	$PM_{2.5}$	CO	CO_2	units	sources/notes
Excavators	1.01	0.12	0.03	0.03	0.35	158.54	lb/hr	wksht: Offroad emfacs; See Note 4
Rubber Tired Loaders	1.10	0.12	0.04	0.04	0.35	148.84	lb/hr	
Chippers/Stump Grinders	3.78	0.32	0.12	0.12	1.23	594.49	lb/hr	
Shredders	0.58	0.06	0.02	0.02	0.35	70.16	lb/hr	
composite emission factor, all equip. combined	6.47	0.61	0.21	0.21	2.28	972.04	lb/hr	summation

Maximum daily emissions from chipping equipment	NO_x	ROG	PM_{10}	$PM_{2.5}$	CO	CO_2	units	sources/notes
gasification alternatives	42.2	4.0	1.4	1.4	14.9	6,341	lb/day	calculation
direct combustion alternative	49.6	4.7	1.6	1.6	17.5	7,460	lb/day	calculation

GHG emissions from chipping equipment were estimated based on fuel consumption measured during the pilot project presented in he following study: Sierra Nevada Conservancy. 2008 (November 17). Forest Biomass Removal on National Forest Lands: First Progress Report. *Prepared by Placer County Chief Executive Office and TSS Consultants. This method was also chosen instead of using exhaust emission factors for offroad equipment from ARB because the GHG emission factors in OFFROAD2007 are not consistent with ARB's official GHG inventory, as noted at http://www.arb.ca.gov/msei/offroad/offroad.htm.*

CO2-e Emission Factor for Diesel Fuel Consumption	value	units	source
GHG emission factors for diesel fuel consumption			
CO_2	10.15	kg/gal	Ref. 3, Table C.3 on p. 96
CH_4	0.74	g/gal	Ref. 3, Table C.6 on p. 100
N_2O	0.26	g/gal	Ref. 3, Table C.6 on p. 100
mass conversion rate	1,000	g/kg	wksht: Unit Conversions
global warming potential of GHGs			
CH_4	21	ratio	wksht: Unit Conversions
N_2O	310	ratio	wksht: Unit Conversions
CO_2-e emissions emission factor	10.25	kg/gal	calculation
mass conversion rate	1,000	kg/MT	wksht: Unit Conversions
CO_2-e emissions emission factor	0.01025	MT/gal	conversion calculation

	gasification alternatives	direct combustion alternative	units	source
diesel fuel consumed by chipping	29,366	34,549	gal/year	calculation
CO_2-e emissions	301	354	MT/year	calculation
Crosscheck using Emission Rates based on OFFROAD model run				
	760,967	895,256	lb/season	calculation based on emfac
	2,204.62	2,204.62	lb/MT	wksht: Unit Conversions
	345	406	MT/year	conversion calculation; See Note 3

Notes

1 It is assumed that all offroad equipment used for chipping biomass is powered by diesel fuel.

2 The maximum number of equipment-hours per day is based on the fact that biomass would be recovered during the 167-day period between May 1 and October 15 each year, which is the same as the grading season designated by TRPA (TRPA Code of Ordinances, Section 64.2.A, Grading Season), and that fuel collection and hauling would occur 5 days per week. This comes to 120 days, a value that was also used in the traffic analysis (See Table 6-9 of Section 6, Traffic and Transportation). Thus, the maximum number of equipment-hours per day is estimated based on the amount of biomass fuel needed by the plant and the 120-days when the fuel would be collected.

3 As a crosscheck, annual CO_2 emissions were also estimated using the lb/hr emission factors based on the OFFROAD2007 model run. This crosscheck does not account for emissions of other GHGs (i.e., CH_4 and N_2O). The annual estimate of CO_2 is not substantially different from the estimate based on fuel consumption.

4 The emission rate for $PM_{2.5}$ exhaust is assumed to be the same as for PM_{10} exhaust because diesel PM is less than 2.5 microns in aerodynamic diameter.

5 Emission rates for PM_{10} and $PM_{2.5}$ are exhaust only and do not include fugitive PM emissions associated with operation of chipping-related equipment.

Emission Factors for Off Road Equipment

Select Output Data from OFFROAD2007 Model Run

Equipment	MaxHP	Population	Activity	Consumption	NOX Exhaust	ROG Exhaust	PM Exhaust	CO Exhaust	SO2 Exhaust	CO2 Exhaust	N2O Exhaust	CH4 Exhaust
Units	hp	#	equip-hr/day	gal/day (diesel)	ton/day	ton/day	ton/day	ton/day	ton/day	ton/day	ton/day	ton/day
Excavators	250	2.32E-01	1.03E+00	7.39E+00	5.20E-04	6.07E-05	1.72E-05	1.79E-04	9.17E-07	8.15E-02	0.00E+00	5.47E-06
Rubber Tired Loaders	250	3.24E-01	9.85E-01	6.65E+00	5.41E-04	5.84E-05	1.85E-05	1.75E-04	8.25E-07	7.33E-02	0.00E+00	5.27E-06
Chippers/Stump Grinders	750	3.04E-02	4.24E-02	1.14E+00	8.01E-05	6.76E-06	2.46E-06	2.61E-05	1.27E-07	1.26E-02	0.00E+00	6.10E-07
Shredders	175	1.23E-03	4.59E-04	1.47E-03	1.33E-07	1.30E-08	5.63E-09	7.97E-08	1.81E-10	1.61E-05	0.00E+00	1.17E-09

	value	units	source
mass conversion rate	2,000	lb/ton	wksht: Unit Conversions

Exhaust Emission Factors

	NOx	ROG	PM_{10}	$PM_{2.5}$	CO	CO2	CH4	units	source
Excavators	1.01	0.12	0.03	0.03	0.35	158.54	0.01	lb/hr	calculation and conversion
Rubber Tired Loaders	1.10	0.12	0.04	0.04	0.35	148.84	0.01	lb/hr	calculation and conversion
Chippers/Stump Grinders	3.78	0.32	0.12	0.12	1.23	594.49	0.03	lb/hr	calculation and conversion
Shredders	0.58	0.06	0.02	0.02	0.35	70.16	0.01	lb/hr	calculation and conversion

Output from OFFROAD2007 Model

CY	Season	AvgDays	Code	Equipment	Fuel	MaxHP	Class	C/R	Pre	Hand	Port	County	Air Basin	Air Dist.	Population	Activity	Consumpti	ROG Exhau	CO Exhaus	NOX Exhau	CO2 Exhau	SO2 Exhau	PM Exhaus	N2O Exhau	CH4 Exhau
2014	Summer	Mon-Fri	2.27E+09	Pavers	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	5.00E-02	1.30E-01	7.60E-01	1.10E-05	5.02E-05	8.50E-05	8.32E-03	9.36E-08	4.69E-06	0.00E+00	9.95E-07
2014	Summer	Mon-Fri	2.27E+09	Pavers	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.02E-03	1.56E-02	1.38E-01	1.53E-06	4.54E-06	1.42E-05	1.52E-03	1.71E-08	5.44E-07	0.00E+00	1.38E-07
2014	Summer	Mon-Fri	2.27E+09	Pavers	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.18E-03	1.60E-02	1.70E-01	1.73E-06	6.91E-06	1.57E-05	1.87E-03	1.84E-08	6.05E-07	0.00E+00	1.57E-07
2014	Summer	Mon-Fri	2.27E+09	Rollers	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.33E-01	2.91E-01	1.43E+00	1.72E-05	8.98E-05	1.39E-04	1.57E-02	1.77E-07	7.43E-06	0.00E+00	1.55E-06
2014	Summer	Mon-Fri	2.27E+09	Rollers	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.88E-02	4.12E-02	2.86E-01	2.43E-06	7.65E-06	2.48E-05	3.15E-03	3.55E-08	8.40E-07	0.00E+00	2.20E-07
2014	Summer	Mon-Fri	2.27E+09	Rollers	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.32E-02	2.89E-02	2.87E-01	2.25E-06	8.55E-06	2.22E-05	3.16E-03	3.11E-08	7.76E-07	0.00E+00	2.03E-07
2014	Summer	Mon-Fri	2.27E+09	Scrapers	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	2.79E-02	9.68E-02	6.54E-01	9.57E-06	4.38E-05	7.15E-05	7.16E-03	8.05E-08	4.00E-06	0.00E+00	8.63E-07
2014	Summer	Mon-Fri	2.27E+09	Scrapers	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.72E-02	9.43E-02	8.97E-01	1.01E-05	2.89E-05	8.94E-05	9.87E-03	1.11E-07	3.42E-06	0.00E+00	9.08E-07
2014	Summer	Mon-Fri	2.27E+09	Scrapers	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	7.49E-02	2.60E-01	3.79E+00	3.93E-05	1.47E-04	3.39E-04	4.17E-02	4.09E-07	1.31E-05	0.00E+00	3.55E-06
2014	Summer	Mon-Fri	2.27E+09	Scrapers	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.38E-01	1.17E+00	2.95E+01	3.08E-04	1.14E-03	2.70E-03	3.25E-01	3.27E-06	1.03E-04	0.00E+00	2.78E-05
2014	Summer	Mon-Fri	2.27E+09	Paving Equ	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.16E-02	3.04E-02	1.40E-01	2.01E-06	9.18E-06	1.56E-05	1.53E-03	1.73E-08	8.60E-07	0.00E+00	1.82E-07
2014	Summer	Mon-Fri	2.27E+09	Paving Equ	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.28E-03	8.56E-03	4.75E-02	5.16E-07	1.54E-06	4.86E-06	5.23E-04	5.89E-09	1.84E-07	0.00E+00	4.66E-08
2014	Summer	Mon-Fri	2.27E+09	Surfacing E	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	2.35E-04	3.31E-04	1.29E-03	1.40E-08	7.80E-08	1.21E-07	1.42E-05	1.60E-10	6.02E-09	0.00E+00	1.26E-09
2014	Summer	Mon-Fri	2.27E+09	Surfacing E	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	4.69E-04	6.62E-04	4.04E-03	3.16E-08	1.07E-07	3.39E-07	4.46E-05	5.02E-10	1.13E-08	0.00E+00	2.85E-09
2014	Summer	Mon-Fri	2.27E+09	Surfacing E	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.91E-03	5.52E-03	5.53E-02	3.95E-07	1.67E-06	4.18E-06	6.10E-04	5.99E-09	1.42E-07	0.00E+00	3.56E-08
2014	Summer	Mon-Fri	2.27E+09	Surfacing E	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.46E-02	9.11E-02	1.43E+00	1.04E-05	4.33E-05	1.11E-04	1.58E-02	1.59E-07	3.73E-06	0.00E+00	9.38E-07
2014	Summer	Mon-Fri	2.27E+09	Signal Boar	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.11E-02	1.85E-02	1.30E-01	1.32E-06	7.70E-06	1.16E-05	1.43E-03	1.61E-08	5.78E-07	0.00E+00	1.19E-07
2014	Summer	Mon-Fri	2.27E+09	Signal Boar	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.35E-03	3.92E-03	4.52E-02	2.98E-07	1.02E-06	3.54E-06	4.99E-04	5.62E-09	1.03E-07	0.00E+00	2.69E-08
2014	Summer	Mon-Fri	2.27E+09	Trenchers	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	3.49E-02	6.86E-02	4.50E-01	6.41E-06	2.94E-05	5.07E-05	4.93E-03	5.55E-08	2.74E-06	0.00E+00	5.78E-07
2014	Summer	Mon-Fri	2.27E+09	Trenchers	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.13E-03	6.15E-03	6.22E-02	6.84E-07	2.08E-06	6.44E-06	6.85E-04	7.71E-09	2.50E-07	0.00E+00	6.17E-08
2014	Summer	Mon-Fri	2.27E+09	Trenchers	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.99E-03	7.84E-03	1.11E-01	1.11E-06	4.73E-06	1.04E-05	1.22E-03	1.20E-08	4.00E-07	0.00E+00	1.00E-07
2014	Summer	Mon-Fri	2.27E+09	Trenchers	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.29E-02	2.54E-02	6.77E-01	6.82E-06	2.89E-05	6.46E-05	7.44E-03	7.49E-08	2.47E-06	0.00E+00	6.15E-07
2014	Summer	Mon-Fri	2.27E+09	Bore/Drill I	D	175	Constructi	U	P	NHH	P	Placer	LT	PLA	7.27E-03	1.91E-02	1.23E-01	6.41E-07	7.20E-06	6.23E-06	1.35E-03	1.52E-08	2.35E-07	0.00E+00	5.78E-08
2014	Summer	Mon-Fri	2.27E+09	Bore/Drill I	D	250	Constructi	U	N	NHH	P	Placer	LT	PLA	6.25E-03	1.64E-02	1.40E-01	6.05E-07	2.81E-06	5.04E-06	1.54E-03	1.74E-08	1.47E-07	0.00E+00	5.46E-08
2014	Summer	Mon-Fri	2.27E+09	Bore/Drill I	D	500	Constructi	U	N	NHH	P	Placer	LT	PLA	1.39E-02	3.66E-02	5.14E-01	2.20E-06	1.01E-05	1.74E-05	5.69E-03	5.58E-08	5.36E-07	0.00E+00	1.99E-07
2014	Summer	Mon-Fri	2.27E+09	Bore/Drill I	D	750	Constructi	U	N	NHH	P	Placer	LT	PLA	2.00E-01	5.26E-01	1.46E+01	6.27E-05	2.86E-04	4.99E-04	1.62E-01	1.63E-06	1.53E-05	0.00E+00	5.66E-06
2014	Summer	Mon-Fri	2.27E+09	Bore/Drill I	D	1000	Constructi	U	N	NHH	P	Placer	LT	PLA	3.36E-01	5.97E-01	2.51E+01	1.16E-04	4.95E-04	1.61E-03	2.77E-01	2.78E-06	4.21E-05	0.00E+00	1.05E-05
2014	Summer	Mon-Fri	2.27E+09	Excavators	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	5.71E-01	2.53E+00	1.29E+01	1.44E-04	8.41E-04	1.05E-03	1.42E-01	1.59E-06	5.80E-05	0.00E+00	1.30E-05
2014	Summer	Mon-Fri	2.27E+09	Excavators	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.32E-01	1.03E+00	7.39E+00	6.07E-05	1.79E-04	5.20E-04	8.15E-02	9.17E-07	1.72E-05	0.00E+00	5.47E-06
2014	Summer	Mon-Fri	2.27E+09	Excavators	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.67E-01	7.42E-01	7.85E+00	6.14E-05	1.89E-04	4.87E-04	8.66E-02	8.50E-07	1.72E-05	0.00E+00	5.54E-06
2014	Summer	Mon-Fri	2.27E+09	Excavators	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.01E-01	4.48E-01	7.86E+00	6.19E-05	1.89E-04	5.04E-04	8.67E-02	8.72E-07	1.75E-05	0.00E+00	5.58E-06
2014	Summer	Mon-Fri	2.27E+09	Concrete/I	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.56E-04	2.83E-04	2.06E-03	2.07E-08	1.23E-07	1.81E-07	2.26E-05	2.55E-10	9.16E-09	0.00E+00	1.87E-09
2014	Summer	Mon-Fri	2.27E+09	Cranes	D	175	Constructi	U	P	NHH	P	Placer	LT	PLA	2.92E-02	1.16E-01	4.27E-01	5.70E-06	2.80E-05	4.26E-05	4.67E-03	5.26E-08	2.40E-06	0.00E+00	5.15E-07
2014	Summer	Mon-Fri	2.27E+09	Cranes	D	250	Constructi	U	N	NHH	P	Placer	LT	PLA	5.65E-02	2.26E-01	1.15E+00	1.11E-05	3.17E-05	1.03E-04	1.26E-02	1.42E-07	3.57E-06	0.00E+00	9.97E-07
2014	Summer	Mon-Fri	2.27E+09	Cranes	D	500	Constructi	U	N	NHH	P	Placer	LT	PLA	2.07E-02	8.27E-02	6.76E-01	6.07E-06	2.04E-05	5.37E-05	7.44E-03	7.31E-08	1.94E-06	0.00E+00	5.47E-07
2014	Summer	Mon-Fri	2.27E+09	Cranes	D	750	Constructi	U	N	NHH	P	Placer	LT	PLA	4.20E-01	1.68E+00	2.30E+01	2.08E-04	6.96E-04	1.88E-03	2.54E-01	2.55E-06	6.72E-05	0.00E+00	1.88E-05
2014	Summer	Mon-Fri	2.27E+09	Cranes	D	9999	Constructi	U	N	NHH	P	Placer	LT	PLA	5.27E-01	1.43E+00	6.28E+01	6.49E-04	2.21E-03	6.98E-03	6.91E-01	6.95E-06	2.14E-04	0.00E+00	5.86E-05
2014	Summer	Mon-Fri	2.27E+09	Graders	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	2.49E-01	7.39E-01	4.18E+00	5.14E-05	2.71E-04	3.89E-04	4.58E-02	5.15E-07	2.14E-05	0.00E+00	4.64E-06
2014	Summer	Mon-Fri	2.27E+09	Graders	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.55E-01	4.59E-01	3.58E+00	3.23E-05	9.57E-05	2.95E-04	3.95E-02	4.44E-07	1.02E-05	0.00E+00	2.91E-06
2014	Summer	Mon-Fri	2.27E+09	Graders	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	4.38E-03	1.30E-02	1.35E-01	1.14E-06	3.88E-06	9.90E-06	1.49E-03	1.46E-08	3.56E-07	0.00E+00	1.03E-07
2014	Summer	Mon-Fri	2.27E+09	Graders	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.46E-03	1.91E-02	4.21E-01	3.58E-06	1.21E-05	3.18E-05	4.64E-03	4.67E-08	1.13E-06	0.00E+00	3.23E-07
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	5.08E-03	3.15E-02	1.80E-01	2.14E-06	1.19E-05	1.52E-05	1.97E-03	2.21E-08	8.51E-07	0.00E+00	1.93E-07
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.75E-02	2.33E-01	1.75E+00	1.54E-05	4.37E-05	1.29E-04	1.93E-02	2.18E-07	4.29E-06	0.00E+00	1.39E-06
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	5.28E-02	3.28E-01	4.04E+00	3.38E-05	1.00E-04	2.61E-04	4.46E-02	4.37E-07	9.29E-06	0.00E+00	3.05E-06
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.35E+00	8.39E+00	1.68E+02	1.41E-03	4.17E-03	1.12E-02	1.85E+00	1.86E-05	3.93E-04	0.00E+00	1.27E-04
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	1000	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.35E-01	2.66E+00	7.54E+01	6.90E-04	2.08E-03	7.43E-03	8.31E-01	8.36E-06	2.21E-04	0.00E+00	6.23E-05
2014	Summer	Mon-Fri	2.27E+09	Crushing/P	D	175	Constructi	U	P	NHH	P	Placer	LT	PLA	1.49E-02	4.45E-02	3.39E-01	4.02E-06	2.13E-05	3.16E-05	3.72E-03	4.18E-08	1.75E-06	0.00E+00	3.63E-07
2014	Summer	Mon-Fri	2.27E+09	Crushing/P	D	250	Constructi	U	N	NHH	P	Placer	LT	PLA	1.49E-03	4.43E-03	4.90E-02	3.86E-07	1.17E-06	4.05E-06	5.41E-04	6.08E-09	1.24E-07	0.00E+00	3.49E-08
2014	Summer	Mon-Fri	2.27E+09	Crushing/P	D	500	Constructi	U	N	NHH	P	Placer	LT	PLA	8.36E-03	2.49E-02	4.22E-01	3.09E-06	1.01E-05	3.04E-05	4.65E-03	4.57E-08	9.98E-07	0.00E+00	2.79E-07
2014	Summer	Mon-Fri	2.27E+09	Crushing/P	D	750	Constructi	U	N	NHH	P	Placer	LT	PLA	1.08E-02	3.21E-02	8.54E-01	6.29E-06	2.02E-05	6.39E-05	9.43E-03	9.48E-08	2.05E-06	0.00E+00	5.68E-07

2014	Summer	Mon-Fri	2.27E+09	Crushing/P	D	9999	Constructi	U	N	NHH	P	Placer	LT	PLA	1.08E-02	3.21E-02	1.90E+00	1.68E-05	5.38E-05	1.99E-04	2.09E-02	2.11E-07	5.72E-06	0.00E+00	1.52E-06
2014	Summer	Mon-Fri	2.27E+09	Rough Terr	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	5.32E-02	1.88E-01	1.07E+00	1.19E-05	6.81E-05	9.19E-05	1.17E-02	1.32E-07	5.04E-06	0.00E+00	1.08E-06
2014	Summer	Mon-Fri	2.27E+09	Rough Terr	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.97E-03	1.05E-02	8.11E-02	6.46E-07	1.95E-06	6.12E-06	8.95E-04	1.01E-08	1.97E-07	0.00E+00	5.83E-08
2014	Summer	Mon-Fri	2.27E+09	Rough Terr	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.95E-03	6.90E-03	8.01E-02	6.02E-07	1.90E-06	5.29E-06	8.85E-04	8.68E-09	1.83E-07	0.00E+00	5.43E-08
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	3.26E-01	9.91E-01	4.80E+00	5.81E-05	3.10E-04	4.43E-04	5.26E-02	5.92E-07	2.43E-05	0.00E+00	5.24E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.24E-01	9.85E-01	6.65E+00	5.84E-05	1.75E-04	5.41E-04	7.33E-02	8.25E-07	1.85E-05	0.00E+00	5.27E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.35E-01	4.10E-01	4.40E+00	3.63E-05	1.24E-04	3.18E-04	4.85E-02	4.77E-07	1.13E-05	0.00E+00	3.27E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.60E-01	7.92E-01	1.74E+01	1.44E-04	4.92E-04	1.29E-03	1.92E-01	1.93E-06	4.56E-05	0.00E+00	1.30E-05
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	1000	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.80E-02	5.76E-02	1.55E+00	1.42E-05	4.99E-05	1.62E-04	1.71E-02	1.72E-07	4.85E-06	0.00E+00	1.28E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	7.82E-04	3.97E-03	2.35E-02	4.04E-07	1.66E-06	2.95E-06	2.57E-04	2.89E-09	1.67E-07	0.00E+00	3.65E-08
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.92E-02	9.72E-02	8.11E-01	1.13E-05	3.18E-05	9.49E-05	8.91E-03	1.00E-07	3.98E-06	0.00E+00	1.02E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.95E-02	1.50E-01	1.81E+00	2.29E-05	9.91E-05	1.91E-04	1.98E-02	1.94E-07	7.89E-06	0.00E+00	2.07E-06
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	2.86E-01	1.45E+00	2.64E+01	3.36E-04	1.45E-03	2.84E-03	2.89E-01	2.91E-06	1.16E-04	0.00E+00	3.03E-05
2014	Summer	Mon-Fri	2.27E+09	Rubber Tir	D	1000	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.94E-02	6.65E-02	1.80E+00	2.39E-05	1.07E-04	2.37E-04	1.97E-02	1.98E-07	8.17E-06	0.00E+00	2.16E-06
2014	Summer	Mon-Fri	2.27E+09	Tractors/Lo	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.31E-01	3.96E-01	1.83E+00	1.84E-05	1.16E-04	1.42E-04	2.00E-02	2.26E-07	7.55E-06	0.00E+00	1.66E-06
2014	Summer	Mon-Fri	2.27E+09	Tractors/Lo	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	4.25E-02	1.28E-01	9.95E-01	7.31E-06	2.31E-05	6.61E-05	1.10E-02	1.24E-07	2.12E-06	0.00E+00	6.60E-07
2014	Summer	Mon-Fri	2.27E+09	Tractors/Lo	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	6.86E-02	2.07E-01	3.22E+00	2.26E-05	7.48E-05	1.89E-04	3.56E-02	4.01E-07	6.48E-06	0.00E+00	2.04E-06
2014	Summer	Mon-Fri	2.27E+09	Tractors/Lo	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.30E+00	3.92E+00	9.17E+01	6.47E-04	2.13E-03	5.54E-03	1.01E+00	1.14E-05	1.88E-04	0.00E+00	5.84E-05
2014	Summer	Mon-Fri	2.27E+09	Crawler Tra	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	2.10E-01	6.85E-01	3.79E+00	5.48E-05	2.54E-04	4.07E-04	4.15E-02	4.67E-07	2.28E-05	0.00E+00	4.94E-06
2014	Summer	Mon-Fri	2.27E+09	Crawler Tra	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.81E-01	5.89E-01	4.44E+00	4.92E-05	1.41E-04	4.34E-04	4.89E-02	5.50E-07	1.65E-05	0.00E+00	4.44E-06
2014	Summer	Mon-Fri	2.27E+09	Crawler Tra	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.24E-01	4.04E-01	4.75E+00	4.88E-05	1.79E-04	4.17E-04	5.23E-02	5.13E-07	1.61E-05	0.00E+00	4.40E-06
2014	Summer	Mon-Fri	2.27E+09	Crawler Tra	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.72E-01	5.62E-01	1.18E+01	1.22E-04	4.46E-04	1.06E-03	1.30E-01	1.31E-06	4.06E-05	0.00E+00	1.10E-05
2014	Summer	Mon-Fri	2.27E+09	Crawler Tra	D	1000	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.72E-01	3.80E-01	1.14E+01	1.25E-04	4.78E-04	1.33E-03	1.25E-01	1.26E-06	4.23E-05	0.00E+00	1.13E-05
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	9.56E-02	3.33E-01	1.98E+00	3.27E-05	1.38E-04	2.43E-04	2.17E-02	2.44E-07	1.37E-05	0.00E+00	2.95E-06
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	250	Constructi	U	N	NHH	NP	Placer	LT	PLA	9.04E-02	3.15E-01	1.86E+00	2.46E-05	7.05E-05	2.13E-04	2.05E-02	2.31E-07	8.78E-06	0.00E+00	2.22E-06
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	750	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.08E+00	3.76E+00	9.73E+01	1.17E-03	5.05E-03	1.02E-02	1.07E+00	1.07E-05	4.13E-04	0.00E+00	1.06E-04
2014	Summer	Mon-Fri	2.27E+09	Off-Highwa	D	1000	Constructi	U	N	NHH	NP	Placer	LT	PLA	1.14E-01	2.69E-01	9.97E+00	1.26E-04	5.65E-04	1.29E-03	1.09E-01	1.10E-06	4.37E-05	0.00E+00	1.14E-05
2014	Summer	Mon-Fri	2.27E+09	Other Cons	D	175	Constructi	U	P	NHH	NP	Placer	LT	PLA	1.53E-02	3.35E-02	1.63E-01	1.46E-06	9.83E-06	1.26E-05	1.78E-03	2.01E-08	6.30E-07	0.00E+00	1.32E-07
2014	Summer	Mon-Fri	2.27E+09	Other Cons	D	500	Constructi	U	N	NHH	NP	Placer	LT	PLA	3.56E-02	7.78E-02	8.95E-01	5.37E-06	1.97E-05	5.24E-05	9.88E-03	9.70E-08	1.71E-06	0.00E+00	4.84E-07
2014	Summer	Mon-Fri	2.27E+09	Aerial Lifts	D	500	Industrial E	U	N	NHH	NP	Placer	LT	PLA	2.81E-01	3.36E-01	3.24E+00	1.86E-05	7.37E-05	2.35E-04	3.58E-02	3.51E-07	7.13E-06	0.00E+00	1.68E-06
2014	Summer	Mon-Fri	2.27E+09	Aerial Lifts	D	750	Industrial E	U	N	NHH	NP	Placer	LT	PLA	2.26E-02	2.71E-02	4.71E-01	2.80E-06	1.07E-05	3.54E-05	5.20E-03	5.23E-08	1.06E-06	0.00E+00	2.52E-07
2014	Summer	Mon-Fri	2.27E+09	Forklifts	D	175	Industrial E	U	P	NHH	NP	Placer	LT	PLA	1.18E+00	6.63E+00	1.70E+01	1.78E-04	1.11E-03	1.30E-03	1.86E-01	2.09E-06	7.38E-05	0.00E+00	1.60E-05
2014	Summer	Mon-Fri	2.27E+09	Forklifts	D	250	Industrial E	U	N	NHH	NP	Placer	LT	PLA	1.17E+00	6.58E+00	2.30E+01	1.86E-04	5.28E-04	1.57E-03	2.54E-01	2.85E-06	5.26E-05	0.00E+00	1.68E-05
2014	Summer	Mon-Fri	2.27E+09	Forklifts	D	500	Industrial E	U	N	NHH	NP	Placer	LT	PLA	5.02E-01	2.82E+00	1.42E+01	1.11E-04	3.09E-04	8.47E-04	1.56E-01	1.53E-06	3.12E-05	0.00E+00	9.98E-06
2014	Summer	Mon-Fri	2.27E+09	Sweepers/S	D	175	Industrial E	U	N	NHH	NP	Placer	LT	PLA	8.16E-01	3.11E+00	1.97E+01	2.09E-04	1.25E-03	1.61E-03	2.16E-01	2.43E-06	9.04E-05	0.00E+00	1.88E-05
2014	Summer	Mon-Fri	2.27E+09	Sweepers/S	D	250	Industrial E	U	N	NHH	NP	Placer	LT	PLA	1.31E-01	4.97E-01	3.65E+00	2.81E-05	8.30E-05	2.67E-04	4.02E-02	4.53E-07	8.60E-06	0.00E+00	2.53E-06
2014	Summer	Mon-Fri	2.27E+09	Other Gen	D	175	Industrial E	U	N	NHH	NP	Placer	LT	PLA	9.61E-01	4.27E+00	1.87E+01	2.53E-04	1.23E-03	1.86E-03	2.05E-01	2.30E-06	1.09E-04	0.00E+00	2.29E-05
2014	Summer	Mon-Fri	2.27E+09	Other Gen	D	250	Industrial E	U	N	NHH	NP	Placer	LT	PLA	9.57E-01	4.26E+00	2.61E+01	2.44E-04	6.56E-04	2.37E-03	2.88E-01	3.24E-06	7.69E-05	0.00E+00	2.20E-05
2014	Summer	Mon-Fri	2.27E+09	Other Gen	D	500	Industrial E	U	N	NHH	NP	Placer	LT	PLA	9.55E-01	4.25E+00	5.11E+01	4.46E-04	1.28E-03	4.01E-03	5.63E-01	5.53E-06	1.40E-04	0.00E+00	4.02E-05
2014	Summer	Mon-Fri	2.27E+09	Other Gen	D	750	Industrial E	U	N	NHH	NP	Placer	LT	PLA	2.39E-01	1.06E+00	2.10E+01	1.85E-04	5.26E-04	1.71E-03	2.32E-01	2.33E-06	5.90E-05	0.00E+00	1.67E-05
2014	Summer	Mon-Fri	2.27E+09	Other Gen	D	1000	Industrial E	U	N	NHH	NP	Placer	LT	PLA	1.45E-01	4.37E-01	1.11E+01	1.10E-04	3.27E-04	1.26E-03	1.22E-01	1.23E-06	3.72E-05	0.00E+00	9.88E-06
2014	Summer	Mon-Fri	2.27E+09	Other Mat	D	175	Industrial E	U	N	NHH	NP	Placer	LT	PLA	4.42E-02	1.82E-01	1.01E+00	1.36E-05	6.62E-05	1.00E-04	1.11E-02	1.25E-07	5.87E-06	0.00E+00	1.22E-06
2014	Summer	Mon-Fri	2.27E+09	Other Mat	D	250	Industrial E	U	N	NHH	NP	Placer	LT	PLA	1.05E-01	4.32E-01	2.84E+00	2.61E-05	7.09E-05	2.57E-04	3.13E-02	3.52E-07	8.30E-06	0.00E+00	2.35E-06
2014	Summer	Mon-Fri	2.27E+09	Other Mat	D	500	Industrial E	U	N	NHH	NP	Placer	LT	PLA	1.96E-02	8.08E-02	7.01E-01	6.03E-06	1.75E-05	5.49E-05	7.73E-03	7.59E-08	1.91E-06	0.00E+00	5.44E-07
2014	Summer	Mon-Fri	2.27E+09	Other Mat	D	9999	Industrial E	U	N	NHH	NP	Placer	LT	PLA	5.89E-03	2.42E-02	8.14E-01	8.12E-06	2.39E-05	9.22E-05	8.97E-03	8.81E-08	2.71E-06	0.00E+00	7.32E-07
2014	Summer	Mon-Fri	2.27E+09	Leaf Blowe	D	250	Lawn and (	U	N	NHH	P	Placer	LT	PLA	1.45E-03	5.21E-04	2.36E-03	1.13E-08	4.90E-08	1.67E-07	2.61E-05	2.94E-10	4.31E-09	0.00E+00	1.02E-09
2014	Summer	Mon-Fri	2.27E+09	Snowblow	D	175	Lawn and (	U	P	NHH	P	Placer	LT	PLA	5.61E-03	2.38E-04	1.44E-03	1.29E-08	8.24E-08	1.21E-07	1.58E-05	1.78E-10	5.71E-09	0.00E+00	1.17E-09
2014	Summer	Mon-Fri	2.27E+09	Snowblow	D	250	Lawn and (	U	N	NHH	P	Placer	LT	PLA	9.54E-02	4.04E-03	3.70E-02	2.24E-07	8.25E-07	2.76E-06	4.08E-04	4.59E-09	8.04E-08	0.00E+00	2.02E-08
2014	Summer	Mon-Fri	2.27E+09	Snowblow	D	500	Lawn and (	U	N	NHH	P	Placer	LT	PLA	2.86E-01	1.21E-02	1.64E-01	9.09E-07	3.66E-06	1.10E-05	1.81E-03	1.78E-08	3.36E-07	0.00E+00	8.20E-08
2014	Summer	Mon-Fri	2.27E+09	Chippers/S	D	175	Lawn and (	U	P	NHH	P	Placer	LT	PLA	1.23E-02	1.72E-02	1.03E-01	9.67E-07	6.00E-06	8.86E-06	1.13E-03	1.27E-08	4.29E-07	0.00E+00	8.73E-08
2014	Summer	Mon-Fri	2.27E+09	Chippers/S	D	250	Lawn and (	U	N	NHH	P	Placer	LT	PLA	2.89E-03	4.04E-03	4.07E-02	2.58E-07	9.19E-07	3.11E-06	4.49E-04	5.05E-09	9.21E-08	0.00E+00	2.33E-08
2014	Summer	Mon-Fri	2.27E+09	Chippers/S	D	500	Lawn and (	U	N	NHH	P	Placer	LT	PLA	2.67E-02	3.74E-02	4.18E-01	2.41E-06	9.57E-06	2.83E-05	4.62E-03	4.53E-08	8.83E-07	0.00E+00	2.17E-07
2014	Summer	Mon-Fri	2.27E+09	Chippers/S	D	750	Lawn and (	U	N	NHH	P	Placer	LT	PLA	3.04E-02	4.24E-02	1.14E+00	6.76E-06	2.61E-05	8.01E-05	1.26E-02	1.27E-07	2.46E-06	0.00E+00	6.10E-07

2014	Summer	Mon-Fri	2.27E+09	Chippers/S	D	1000	Lawn and (	U	N	NHH	P	Placer	LT	PLA	5.78E-02	6.37E-02	2.44E+00	1.79E-05	6.42E-05	2.43E-04	2.70E-02	2.71E-07	6.54E-06	0.00E+00	1.62E-06
2014	Summer	Mon-Fri	2.27E+09	Generator	D	175	Light Comr	U	N	NHH	P	Placer	LT	PLA	2.30E-01	1.84E-01	1.19E+00	1.13E-05	6.79E-05	1.06E-04	1.31E-02	1.47E-07	5.05E-06	0.00E+00	1.02E-06
2014	Summer	Mon-Fri	2.27E+09	Generator	D	250	Light Comr	U	N	NHH	P	Placer	LT	PLA	1.28E-01	1.03E-01	9.89E-01	6.11E-06	2.17E-05	8.02E-05	1.09E-02	1.23E-07	2.26E-06	0.00E+00	5.51E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	500	Light Comr	U	N	NHH	P	Placer	LT	PLA	2.86E-01	2.29E-01	3.49E+00	1.93E-05	7.81E-05	2.51E-04	3.85E-02	3.78E-07	7.49E-06	0.00E+00	1.74E-06
2014	Summer	Mon-Fri	2.27E+09	Generator	D	750	Light Comr	U	N	NHH	P	Placer	LT	PLA	1.78E-01	1.42E-01	3.50E+00	2.01E-05	7.83E-05	2.61E-04	3.86E-02	3.88E-07	7.67E-06	0.00E+00	1.81E-06
2014	Summer	Mon-Fri	2.27E+09	Generator	D	9999	Light Comr	U	N	NHH	P	Placer	LT	PLA	4.62E-02	3.70E-02	1.76E+00	1.34E-05	4.65E-05	1.82E-04	1.94E-02	1.95E-07	4.84E-06	0.00E+00	1.21E-06
2014	Summer	Mon-Fri	2.27E+09	Pumps	D	175	Light Comr	U	P	NHH	P	Placer	LT	PLA	2.38E-01	2.27E-01	1.45E+00	1.44E-05	8.39E-05	1.31E-04	1.59E-02	1.79E-07	6.44E-06	0.00E+00	1.30E-06
2014	Summer	Mon-Fri	2.27E+09	Pumps	D	250	Light Comr	U	N	NHH	P	Placer	LT	PLA	1.71E-01	1.64E-01	1.49E+00	9.68E-06	3.32E-05	1.23E-04	1.65E-02	1.85E-07	3.54E-06	0.00E+00	8.73E-07
2014	Summer	Mon-Fri	2.27E+09	Pumps	D	500	Light Comr	U	N	NHH	P	Placer	LT	PLA	3.38E-03	3.23E-03	5.04E-02	2.95E-07	1.15E-06	3.68E-06	5.57E-04	5.47E-09	1.12E-07	0.00E+00	2.66E-08
2014	Summer	Mon-Fri	2.27E+09	Pumps	D	750	Light Comr	U	N	NHH	P	Placer	LT	PLA	5.64E-04	5.38E-04	1.39E-02	8.38E-08	3.18E-07	1.05E-06	1.53E-04	1.54E-09	3.16E-08	0.00E+00	7.56E-09
2014	Summer	Mon-Fri	2.27E+09	Pumps	D	9999	Light Comr	U	N	NHH	P	Placer	LT	PLA	1.24E-02	1.18E-02	7.27E-01	5.72E-06	1.96E-05	7.59E-05	8.02E-03	8.06E-08	2.04E-06	0.00E+00	5.16E-07
2014	Summer	Mon-Fri	2.27E+09	Air Compre	D	175	Light Comr	U	P	NHH	P	Placer	LT	PLA	1.33E-01	2.57E-01	1.04E+00	1.27E-05	6.49E-05	1.01E-04	1.14E-02	1.28E-07	5.64E-06	0.00E+00	1.14E-06
2014	Summer	Mon-Fri	2.27E+09	Air Compre	D	250	Light Comr	U	N	NHH	P	Placer	LT	PLA	1.87E-01	3.61E-01	2.15E+00	1.74E-05	5.16E-05	1.90E-04	2.37E-02	2.67E-07	5.93E-06	0.00E+00	1.57E-06
2014	Summer	Mon-Fri	2.27E+09	Air Compre	D	500	Light Comr	U	N	NHH	P	Placer	LT	PLA	2.44E-01	4.71E-01	4.94E+00	3.71E-05	1.20E-04	3.81E-04	5.46E-02	5.36E-07	1.28E-05	0.00E+00	3.34E-06
2014	Summer	Mon-Fri	2.27E+09	Air Compre	D	750	Light Comr	U	N	NHH	P	Placer	LT	PLA	9.13E-02	1.76E-01	2.86E+00	2.17E-05	6.92E-05	2.28E-04	3.16E-02	3.17E-07	7.53E-06	0.00E+00	1.96E-06
2014	Summer	Mon-Fri	2.27E+09	Air Compre	D	1000	Light Comr	U	N	NHH	P	Placer	LT	PLA	2.25E-03	4.35E-03	9.60E-02	8.54E-07	2.76E-06	1.06E-05	1.06E-03	1.06E-08	3.01E-07	0.00E+00	7.71E-08
2014	Summer	Mon-Fri	2.27E+09	Welders	D	175	Light Comr	U	P	NHH	P	Placer	LT	PLA	1.01E-02	1.55E-02	6.92E-02	7.88E-07	4.21E-06	6.56E-06	7.58E-04	8.53E-09	3.52E-07	0.00E+00	7.11E-08
2014	Summer	Mon-Fri	2.27E+09	Welders	D	250	Light Comr	U	N	NHH	P	Placer	LT	PLA	2.25E-03	3.44E-03	1.85E-02	1.39E-07	4.34E-07	1.60E-06	2.04E-04	2.30E-09	4.88E-08	0.00E+00	1.26E-08
2014	Summer	Mon-Fri	2.27E+09	Welders	D	500	Light Comr	U	N	NHH	P	Placer	LT	PLA	5.64E-03	8.59E-03	6.51E-02	4.47E-07	1.55E-06	4.93E-06	7.19E-04	7.06E-09	1.60E-07	0.00E+00	4.04E-08
2014	Summer	Mon-Fri	2.27E+09	Shredders	D	175	Logging Eq	U	P	NHH	NP	Placer	LT	PLA	1.23E-03	4.59E-04	1.47E-03	1.30E-08	7.97E-08	1.33E-07	1.61E-05	1.81E-10	5.63E-09	0.00E+00	1.17E-09
2014	Summer	Mon-Fri	2.27E+09	Skidders	D	175	Logging Eq	U	P	NHH	NP	Placer	LT	PLA	5.33E-01	2.40E+00	1.53E+01	1.30E-04	9.69E-04	1.01E-03	1.68E-01	1.89E-06	5.41E-05	0.00E+00	1.18E-05
2014	Summer	Mon-Fri	2.27E+09	Skidders	D	250	Logging Eq	U	N	NHH	NP	Placer	LT	PLA	1.96E-01	8.82E-01	8.40E+00	5.75E-05	1.83E-04	4.78E-04	9.28E-02	1.04E-06	1.60E-05	0.00E+00	5.18E-06
2014	Summer	Mon-Fri	2.27E+09	Skidders	D	500	Logging Eq	U	N	NHH	NP	Placer	LT	PLA	1.10E-02	4.96E-02	5.62E-01	3.76E-06	1.17E-05	2.83E-05	6.21E-03	6.09E-08	1.05E-06	0.00E+00	3.39E-07
2014	Summer	Mon-Fri	2.27E+09	Fellers/Bur	D	175	Logging Eq	U	P	NHH	NP	Placer	LT	PLA	8.95E-01	3.57E+00	2.20E+01	1.79E-04	1.37E-03	1.44E-03	2.41E-01	2.71E-06	7.58E-05	0.00E+00	1.62E-05
2014	Summer	Mon-Fri	2.27E+09	Fellers/Bur	D	250	Logging Eq	U	N	NHH	NP	Placer	LT	PLA	5.46E-01	2.18E+00	1.92E+01	1.25E-04	4.11E-04	1.08E-03	2.12E-01	2.38E-06	3.57E-05	0.00E+00	1.13E-05
2014	Summer	Mon-Fri	2.27E+09	Fellers/Bur	D	500	Logging Eq	U	N	NHH	NP	Placer	LT	PLA	1.60E-01	6.39E-01	8.46E+00	5.38E-05	1.74E-04	4.24E-04	9.35E-02	9.18E-07	1.54E-05	0.00E+00	4.85E-06
2014	Summer	Mon-Fri	2.27E+09	Fellers/Bur	D	750	Logging Eq	U	N	NHH	NP	Placer	LT	PLA	1.23E-02	4.88E-02	1.28E+00	8.17E-06	2.64E-05	6.54E-05	1.42E-02	1.42E-07	2.36E-06	0.00E+00	7.37E-07
2014	Summer	Mon-Fri	2.27E+09	Misc Porta	D	175	Other Port	U	P	NHH	P	Placer	LT	PLA	2.18E-01	2.51E-01	1.08E+00	1.11E-05	6.37E-05	9.84E-05	1.19E-02	1.33E-07	4.89E-06	0.00E+00	1.00E-06
2014	Summer	Mon-Fri	2.27E+09	Misc Porta	D	250	Other Port	U	N	NHH	P	Placer	LT	PLA	4.68E-02	5.37E-02	3.86E-01	2.69E-06	9.15E-06	3.15E-05	4.26E-03	4.79E-08	9.62E-07	0.00E+00	2.43E-07
2014	Summer	Mon-Fri	2.27E+09	Misc Porta	D	500	Other Port	U	N	NHH	P	Placer	LT	PLA	1.72E-01	1.97E-01	2.64E+00	1.67E-05	6.45E-05	1.92E-04	2.91E-02	2.86E-07	6.12E-06	0.00E+00	1.51E-06
2014	Summer	Mon-Fri	2.27E+09	Misc Porta	D	750	Other Port	U	N	NHH	P	Placer	LT	PLA	7.80E-02	8.95E-02	1.71E+00	1.11E-05	4.17E-05	1.28E-04	1.88E-02	1.89E-07	4.03E-06	0.00E+00	9.97E-07
2014	Summer	Mon-Fri	2.27E+09	Misc Porta	D	1000	Other Port	U	N	NHH	P	Placer	LT	PLA	3.12E-02	3.58E-02	9.11E-01	7.13E-06	2.57E-05	9.37E-05	1.00E-02	1.01E-07	2.57E-06	0.00E+00	6.43E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	175	Entertainm	U	N	NHH	P	Placer	LT	PLA	3.90E-02	2.78E-02	1.89E-01	1.76E-06	1.08E-05	1.66E-05	2.08E-03	2.34E-08	7.75E-07	0.00E+00	1.59E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	250	Entertainm	U	N	NHH	P	Placer	LT	PLA	6.01E-02	4.29E-02	3.84E-01	2.41E-06	8.78E-06	3.03E-05	4.24E-03	4.77E-08	8.79E-07	0.00E+00	2.18E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	500	Entertainm	U	N	NHH	P	Placer	LT	PLA	9.33E-02	6.66E-02	8.33E-01	4.72E-06	1.96E-05	5.89E-05	9.20E-03	9.03E-08	1.79E-06	0.00E+00	4.26E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	750	Entertainm	U	N	NHH	P	Placer	LT	PLA	1.64E-02	1.17E-02	2.87E-01	1.68E-06	6.74E-06	2.09E-05	3.17E-03	3.18E-08	6.25E-07	0.00E+00	1.51E-07
2014	Summer	Mon-Fri	2.27E+09	Generator	D	9999	Entertainm	U	N	NHH	P	Placer	LT	PLA	2.40E-03	1.72E-03	7.50E-02	5.68E-07	2.04E-06	7.53E-06	8.28E-04	8.32E-09	2.03E-07	0.00E+00	5.13E-08
2014	Summer	Mon-Fri	2.27E+09	Crane (Rail	D	175	Railyard O	U	P	NHH	P	Placer	LT	PLA	1.44E-03	1.03E-03	3.21E-03	2.99E-08	1.83E-07	2.82E-07	3.52E-05	3.96E-10	1.32E-08	0.00E+00	2.70E-09
2014	Summer	Mon-Fri	2.27E+09	Generator	D	175	Railyard O	U	P	NHH	P	Placer	LT	PLA	4.81E-04	3.43E-04	2.18E-03	2.03E-08	1.24E-07	1.91E-07	2.39E-05	2.69E-10	8.91E-09	0.00E+00	1.83E-09
2014	Summer	Mon-Fri	2.27E+09	Generator	D	9999	Railyard O	U	N	NHH	P	Placer	LT	PLA	4.81E-04	3.43E-04	1.37E-02	1.03E-07	3.71E-07	1.37E-06	1.51E-04	1.52E-09	3.70E-08	0.00E+00	9.34E-09
2014	Summer	Mon-Fri	2.28E+09	Vessels w/	D	250	Pleasure C	U	N	NHH	NP	Placer	LT	PLA	2.74E+01	9.10E+00	4.54E+01	2.68E-03	4.03E-03	9.17E-03	4.87E-01	5.48E-06	2.35E-04	0.00E+00	2.42E-04

GHGs Associated with Water Consumption

	gasification alternatives	direct combustion alternative	units	sources/notes
water consumption	14,400	23,040	gal/day	wksht: Operational Parameters
days of operation per year	365	365	days/year	wksht: Operational Parameters
annual water consumption	126,144,000	201,830,400	gal/year	calculation
annual water consumption	126	202	MG/year	conversion calculation
Annual electricity consumption	342,418	547,869	kW-hr/year	calculation
Annual electricity consumption	342	548	MW-hr	conversion calculation
CO2-e Emissions, annual	222	355	MT/year	calculation

Calculation Inputs

	value	units	source
time conversion rate	24	hr/day	wksht: Unit Conversions
volume conversion rate	1,000,000	gal/MG	wksht: Unit Conversions
depth of well(s)	610	feet	Source 2, p. 3-1
electricity consumption rate	4.45	kW-hr/MG/foot (depth)	Source 1, p. 40
electricity conversion rate	1,000	kW-hr/MW-hr	wksht: Unit Conversions
local electric utility	Sierra Pacific Power	none	assumption based on recent merger activity
GHG Emission Rates			
CO2	1,422.78	lb/MW-hr	Source 3; See Note 1
CH4	0.029	lb/MW-hr	Source 3; See Note 1
N2O	0.011	lb/MW-hr	Source 3; See Note 1
Global warming potential			
CH4	21	unitless	wksht: Unit Conversions
N2O	310	unitless	wksht: Unit Conversions
CO2-e emission factor	1,426.80	lb/MW-hr	composite calculation
mass conversion rate	2,204.62	lb/MT	wksht: Unit Conversions

Notes

1 These are the default GHG emission rates associated with the consumption of electricity produced by Sierra Pacific Power Company, as provided by CalEEMod (listed below). Sierra Pacific's generation and distribution assets are now owned and operated by California Pacific Electric Company (CalPeco).

Source

1 California Energy Commission. 2006 (December). Refining Estimates of Water-Related Energy Use in California. Sacramento, CA. CEC-500-2006-118. Available: <http://www.energy.ca.gov/pier/project_reports/CEC-500-2006-118.html>. Accessed May 3, 2012.

2 Placer County Facility Services. 2003. Eastern Regional Landfill Water System Operation and Maintenance Manual.

3 South Coast Air Quality Management District. 2011. California Emissions Estimator Model (CalEEMod) Version 2011.1.1. Available: <http://www.caleemod.com/>.

GHGs Associated with Treatment of Wastewater Generated by the Plant

	gasification alternatives	direct combustion alternative	units	sources/notes
wastewater generation	126	202	MG/year	wksht: Water Consumption; See Note 1
Annual electricity consumption	241,061	385,698	kW-hr/year	calculation
Annual electricity consumption	241	386	MW-hr	conversion calculation
CO2-e Emissions, annual	156	250	MT/year	calculation

Calculation Inputs	value	units	source
electricity consumption rate	1,911	kW-hr/MG	Source 1, Operational - Water and Wastewater module
electricity conversion rate	1,000	kW-hr/MW-hr	wksht: Unit Conversions
local electric utility	Sierra Pacific Power	none	assumption based on recent merger activity
GHG Emission Rates			
CO2	1,422.78	lb/MW-hr	Source 1; See Note 2
CH4	0.029	lb/MW-hr	Source 1; See Note 2
N2O	0.011	lb/MW-hr	Source 1; See Note 2
Global warming potential			
CH4	21	unitless	wksht: Unit Conversions
N2O	310	unitless	wksht: Unit Conversions
CO2-e emission factor	1,426.80	lb/MW-hr	composite calculation
mass conversion rate	2,204.62	lb/MT	wksht: Unit Conversions

Notes

1 It is conservatively assumed that the volume of wastewater generated by the plant would be equal to the volume of water consumed by the plant.

2 These are the default GHG emission rates associated with the consumption of electricity produced by Sierra Pacific Power Company. Sierra Pacific's generation and distribution assets are now owned and operated by California Pacific Electric Company (CalPeco).

Source

1 South Coast Air Quality Management District. 2011. California Emissions Estimator Model (CalEEMod) Version 2011.1.1. Available: <http://www.caleemod.com/>.

Avoided GHG Emissions from Forest Slash Burning

Methodology: *This worksheet estimates the level of GHG emissions that would be avoided if biomass is consumed by the biomass plant instead of piled and burned by crews performing forest thinning and hazardous fuels reduction in the forests. Communications with Scott Conway of the Tahoe National Forest and Dave Fournier of the USFS Lake Tahoe Basin Management Unit indicate that all of the biomass consumed by the plant would otherwise be piled and burned in the forest. In some cases, biomass material is masticated (or chipped) and spread of the forest floor to achieve other forest management goals, which would result in GHG emissions associated with the decomposition of that material.*

Key Note: *There are typically multiple methodologies, called "tiers," for estimating GHG emissions for different sources. A Tier 1 approach relies on default emission factors and default values for other key input parameters such as the high heating value. The default values used in a Tier 1 methodology are conservative in that they result in a high estimate of GHG emissions. A Tier 2 approach uses project specific emission factors and/or input parameters (e.g., high heating value) that result in a more accurate, and lower estimate of GHG emissions. It is important to note that this analysis is conservative because a Tier 2 method is used to estimate GHG emissions from the gasification or combustion of biomass at the proposed plant and a Tier 1 approach is used to estimate the avoided level of GHG emissions that would be generated by open burning the biomass fuel in the forests.*

Annual Consumption of Forest Thinning Biomass by the Plant

	gasification alternatives	direct combustion alternative	units	source
biomass consumed, maximum annually	17,000	20,000	bdt/year	wksht: Operational Parameters
portion sourced from hazardous fuels reduction in forests	75%	75%	%	wksht: Operational Parameters
portion sourced from forest thinning residuals	25%	25%	%	wksht: Operational Parameters
portion sourced from haz fuels reduction or thinning	100.0%	100.0%	%	summation
forest-sourced biomass, maximum annually	17,000	20,000	bdt/year	calculation
mass conversation rate	907.2	907.2	kg/ton	wksht: Unit Conversions
biomass consumed, maximum annually	15,422,141	18,143,695	kg/year	conversion calculation

	value	units	source
Combustion Factor	95%	%	assumption, See Note 1 and Note 4
Emission Factors (per unit of dry matter burned)	value	units	source
CO2	1,550	g/kg dry biomass	Table 2.5 of Source 1
CH4	6.1	g/kg dry biomass	Table 2.5 of Source 1
N2O	0.06	g/kg dry biomass	Table 2.5 of Source 1
Global Warming Potential for Conversion to CO2e			
global warming potential of CH4	21	unitless	wksht: Unit Conversions
global warming potential of N2O	310	unitless	wksht: Unit Conversions
CO2-e Emission Factor	1,697	g/kg dry biomass	calculation

	value	units	source
mass conversation rate	1,000,000	g/MT	wksht: Unit Conversions

Emissions, Annual	gasification alternatives	direct combustion alternative	units	source
CO2-e Emissions	24,858	29,245	MT/year	calc w/CO2-e emiss factor; See Note 4

Notes

1 It is assumed that forest contractors who burn their piles of forest slash seek to burn off as much of the waste as possible.

2 This calculation methodology is consistent with ARB's methodology for estimating its GHG inventory (see http://www.arb.ca.gov/cc/inventory/data/data.htm), which his consistent with 2006 IPCC guidelines.

3 PCAPCD staff agrees that the same ARB-recommended methodology be used to estimate GHG emissions from direct combustion and gasification and that the only difference would be that gasification technologies require less BDT of biomass fuel to produce the same amount of electricity. In short, gasification technologies are more efficient than direct combustion technologies.

4 None of the biomass consumed by the plant would be urban-sourced (e.g., construction and demolition debris, tree clippings and pine needles from developed land uses).

Sources

1 Intergovernmental Panel on Climate Change (IPCC). 2006. *Guidelines for National Greenhouse Gas Inventories, Volume 4, Agriculture, Forestry and Other Land Use* . Hayama, Kanagawa, Japan. Available: <http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_02_Ch2_Generic.pdf>. Accessed June 2010.

Emissions of CAPs and Precursors from Forest Slash Burning

Emission Rates of Forest Fuel Types Likely to Be Included in Burn Piles from Forest Thinning, Moderate Moisture (lb/ton of fuel burned)

Fuel Component	NO_X	ROG	PM_{10}	$PM_{2.5}$	CO	SO_2	burn efficiency
Litter, wood 0-1 in	8.2	3.7	9.3	7.9	52.4	2.5	90%
Wood 1-3 in	8.0	7.8	14.0	11.9	111.4	2.5	65%
Wood 3+ in	7.6	14.4	21.6	18.3	205.8	2.3	65%
Herb, shrub, regen	7.4	17.4	25.1	21.3	249.2	2.3	65%
Average Emission Rate	5.6	7.3	12.0	10.1	103.8	1.7	—

Source: California Air Resources Board. 2006 (May). *Emissions Inventory Default Methodology for Wildland Fire Use.* (Areawide Sources / Miscellaneous Processes / Wildland Fire Use (WFU) Fires. Emissions Inventory Code 670-667-0200-0000. Available: <http://www.arb.ca.gov/ei/areasrc/distmiscprocwstburndis.htm>. Last Updated October 8, 2008. Accessed June 13, 2010.

	gasification alternatives	direct combustion alternative	units	source/notes
green tons of forest residuals used by biomass plant, minimum	28,000	34,000	ton/year	wksht: Operational Parameters

Emissions from Burning Forest Thinning Slash

	NO_X	ROG	PM_{10}	$PM_{2.5}$	CO	SO_2	units	source/notes
Gasification Alternatives								
Open Burning of Forest-Sourced Biomass								
Annual Emissions	156,310	203,490	334,775	284,095	2,907,240	48,055	lb/year	calculation
Annual Emissions	78	102	167	142	1,454	24	tons/year	conversion calculation
Average Daily Emissions	428.2	557.5	917.2	778.3	7,965.0	131.7	lb/day	conversion calculation
Direct Combustion Alternative								
Open Burning of Forest-Sourced Biomass							tons/year	
Annual Emissions	189,805	247,095	406,513	344,973	3,530,220	58,353	lb/year	calculation
Annual Emissions	95	124	203	172	1,765	29		conversion calculation
Average Daily Emissions	520.0	677.0	1,113.7	945.1	9,671.8	159.9	lb/day	conversion calculation

	value	units	source/notes
mass conversion rate	2,000	lb/ton	wksht: Unit Conversions
burn day frequency	365	days/year	assumption; See Note 3

Notes
1 Duff and Canopy Fuels are fuel types likely not to be included in burn piles from forest thinning.
2 It is assumed that emission rates for ROG are the same as TNMHC.
3 For the estimation of average daily emissions from open pile burning it is assumed that burning would occur most days of the year.
4 It is assumed that the burn efficiencies of "Wood 3+ inches," and "Herb, shrub, regen" are 65%, which is the same as "Wood 1-3 in."

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Zero, one, or in between: evaluation of alternative national and entity-level accounting for bioenergy

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Abstract

Accounting for bioenergy's carbon dioxide (CO_2) emissions, as done under the Kyoto Protocol (KP) and European Union (EU) Emissions Trading Scheme, fails to capture the full extent of these emissions. As a consequence, other approaches have been suggested. Both the EU and United States already use value-chain approaches to determine emissions due to biofuels – an approach quite different from that of the KP. Further, both the EU and United States are engaged in consultation processes to determine how emissions connected with use of biomass for heat and power will be handled under regulatory systems. The United States is considering whether CO_2 emissions from biomass should be handled like fossil fuels. In this context, this article reviews and evaluates the three basic bioenergy accounting options.

1 CO_2 emissions from bioenergy are not counted at the point of combustion. Instead emissions due to use of biomass are accounted for in the land-use sector as carbon stock losses – a combustion factor (CoF) = 0 approach;
2 CO_2 emissions from bioenergy are accounted for in the energy sector – a CoF = 1 approach; and
3 End users account for all or a specified subset of CO_2 emissions, regardless of where geographically these emissions occur – 0 < CoF < 1.

Following short descriptions of the basic options, this article discusses variations to these options and uses numerical examples to illustrate the impacts of approaches at a local and international level. Finally, the alternative accounting systems are evaluated against general criteria and for impacts on selected stakeholder goals. General criteria considered are: (a) comprehensiveness, (b) simplicity, and (c) scale independence. Stakeholder goals reviewed are: (a) stimulation of rural economies, (b) food security, (c) GHG reductions, and (d) preservation of forests.

Keywords: bioenergy, carbon accounting, carbon neutrality

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Introduction

In contrast to fossil fuel carbon stocks, biomass carbon stocks can be replenished relatively quickly by growing new biomass to replace biomass combusted for bioenergy. This is the basic reason why bioenergy can mitigate climate change. However, as has been pointed out by numerous authors, the current accounting system for greenhouse gas (GHG) emissions in operation under the Kyoto Protocol (KP) and EU Emissions Trading Scheme (EU-ETS) fails to capture the full extent of emissions caused by bioenergy. Consequently, nations and energy producers with reporting obligations tend to use more bioenergy than is justified by the amount of GHG emission reductions it achieves (Peters *et al*., 2009; Searchinger *et al*., 2009; Pingoud *et al*., 2010). This article poses the question: would an alternative accounting system lead to use of bioenergy more in line with the emission reductions it achieves?

Under the KP accounting system, no carbon dioxide (CO_2) emissions are counted in the energy sector when the biomass is combusted (zero emissions at point of combustion). Measurements of changes in carbon stock levels in the land-use sector are used as a proxy for measurements of combustion emissions, and the results from the land-use sector are reported in the accounting system. While this approach will correctly account for emissions if all nations report all carbon stock changes, developing countries do not report under the KP. In addition, some stock reductions are not reported in nations that have not elected Article 3.4 (i.e., have chosen not to include forest and agricultural management). To the extent that carbon stock losses are not reported, CO_2 emissions due to combustion of biomass will not be accounted for at all under the KP approach, even if

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*Correction added after online publication 27 October 2011: Author's name added the bioenergy is used in nations that have KP obligations.

Under both the KP and the EU-ETS, energy producers have a powerful incentive to use biomass for energy since they do not have to hold permits for these emissions. There is also an incentive to source the biomass from nations where changes in carbon stocks are not counted or from forests not covered by management reports. Bringing more nations and land-use sector emissions into the accounting system would alleviate this situation. However, there are other ways to bring use of bioenergy more in line with the emission reductions it achieves. In particular, increasing the responsibility of energy sector actors for bioenergy emissions holds promise. This can be done either through 1-combusiton factor or end user responsibility. This article reviews these two accounting approaches with current EU and US consultation processes on regulatory options for biomass used for heat and power underscoring the timeliness of such a review.

Methodology

This article explains three different approaches to accounting for emissions due to use of bioenergy that were described in Bird *et al.* (2010). Diagrams, text, and a numerical example are used to portray differences between the approaches. Following the explanations, the alternative approaches are evaluated against three general criteria. The evaluation builds on a landmark paper on accounting systems which recommended five criteria: accuracy, simplicity, scale independence, precedence, and incentives (Apps *et al*., 1997). Accuracy has been renamed comprehensiveness over space and time to make clear the importance of correct accounting in both dimensions. Scale-independence is an issue because accounting systems may be applied by entities within a nation as well as at national or sub-national jurisdictional levels. We believe precedence was selected due to its contribution to simplicity and therefore consider it within simplicity. In this article, incentive issues are considered to be outcomes of accounting systems. Therefore, they are handled separately from the evaluation criteria. Incentives are considered in connection with three stakeholder goals: stimulation of rural economies, GHG reductions, and preservation of forests. These three goals can be used to represent a wider range of reasons for pursuing bioenergy due to synergies between seemingly disparate goals.

Explanations of different accounting systems

Figure 1 shows the physical flows of GHGs to and from the atmosphere and the transfer of biomass (as carbon, C) from a biomass producer to a consumer that occur when biomass grown and used for energy. Variations of this diagram will be used to illustrate where, in a particular approach, emissions are accounted for. Three physical GHG flows occur in connection with biomass production: CO_2 absorbed by plants, CO_2 oxidized by plants (both of which are shown as Bio-CO_2), and fossil-CO_2 and non-CO_2 emissions that occur during biomass production. There are also GHG emissions connected with conversion of biomass to a fuel and its transportation from the point of production to an initial biofuel purchaser. In Fig. 1, these are included in the producer's emissions. The biofuel purchaser, hereafter denoted as the consumer, has two streams: CO_2 from the combustion of biomass (bioenergy CO_2) and fossil-CO_2 and non-CO_2 emissions from combustion and distribution to an end user.

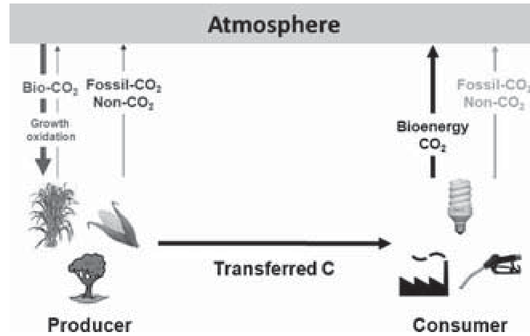

Fig. 1 Physical greenhouse gas emissions and flows of carbon in a bioenergy system.

Figures 2–4 illustrate three basic alternative philosophies that form the basis of all the approaches to accounting for emissions from use of bioenergy.

1 CO_2 emissions produced when biomass is burnt for energy are not counted at the point of combustion. They are accounted for in the land-use sector as carbon stock losses. We term this a *combustion factor* = 0 approach (CoF = 0).

2 CO_2 emissions produced when biomass is burnt for energy are accounted for in the energy sector. We term all such approaches *combustion factor* = 1 approaches (CoF = 1). Here, there are two variations; one in which uptake of CO_2 from the atmosphere by plants and soils is also accounted for and one in which these are not accounted.

3 End users are responsible for all or a specified subset of emissions that occur along the bioenergy value chain. We term these *value-chain* approaches. These approaches can be used to calculate a combustion factor between 0 and 1 (0 < CoF < 1).

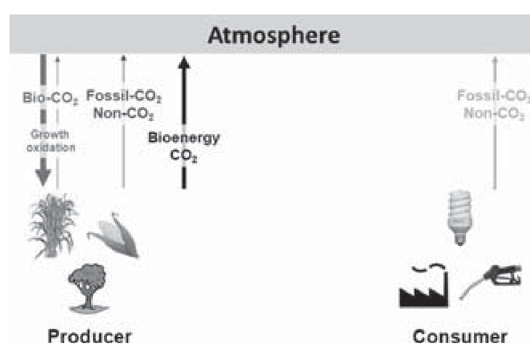

Fig. 2 Location of where the physical flows are theoretically accounted for in a 0-combustion factor approach.

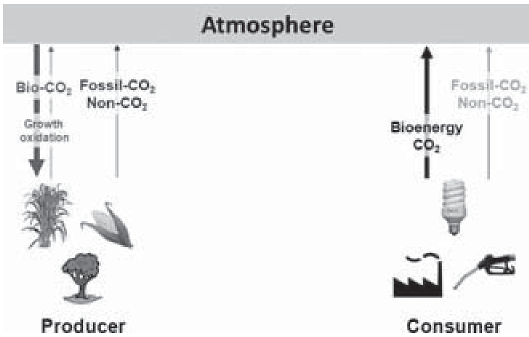

Fig. 3 Location of where the physical flows are theoretically accounted for in a 1-combustion factor approach.

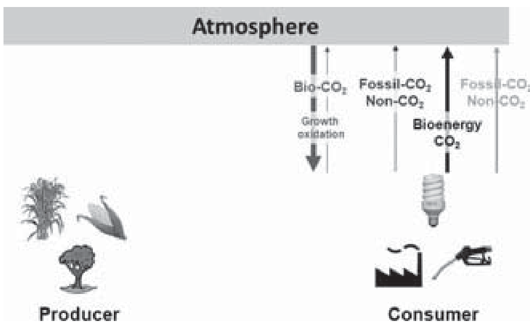

Fig. 4 Location of where the physical flows are theoretically accounted for in a value-chain approach.

*Combustion factor = 0 approaches*In a CoF = 0, approach emissions due to combustion of biomass are counted as carbon stock losses in the land-use sector (see Fig. 2). In this approach, emissions due to transport and conversion of biomass are accounted for outside of the biomass accounting system, i.e., in the fuel combustion or industrial process sectors as appropriate.

The Intergovernmental Panel on Climate Change (IPCC) methodology for calculating emissions from bioenergy, which was adopted under the KP, is an example of a CoF = 0 approach. The concept underlying this approach is that as long as sufficient biomass grows to replace the combusted biomass (Bio-CO_2 $\geq$ Bioenergy CO_2), bioenergy will not result in an increase of CO_2 emissions to the atmosphere. Atmospheric CO_2 increases only if harvesting exceeds growth. In this case, it is assumed that the carbon stock losses will be registered in the accounting system.

*Combustion factor = 1 approaches*The CoF = 1 accounting approaches treat CO_2 emissions from biomass exactly the same as emissions from fossil fuels. Emissions are accounted for in energy sector. Bio-CO_2 (uptake of CO_2 from the atmosphere) is counted by the producer. Emissions other than CO_2 resulting from combusting the carbon in the biomass are accounted for elsewhere in the system (Fig. 3). One can see that the location of where the physical flows would be accounted for in a CoF = 1 approach reflects the actual physical GHG emissions and flows of carbon (Fig. 1).

*Value-chain approaches*In value-chain approaches, GHG emissions and CO_2 removals that occur throughout all the production, conversion, transportation, and consumption processes are considered the responsibility of the consumer. Emissions that are accounted elsewhere in the system in the 0- and 1-combustion factor approaches (blue arrows) are included in the bioenergy account, and all flows appear on the consumer side of our schematic diagram (Fig. 4).

While sharing the life-cycle assessment (LCA) approach of considering impacts throughout a product's life, GHG emission value chains only consider GHG emissions. By not considering energy balances, process details, or other inputs or outputs, they are considerably simpler than full LCAs.

Emissions along the value chain can be used to generate a combustion factor between 0 and 1.[1] The atmospheric removals and emissions over the full production-through-use cycle are aggregated into a single number, percent or ratio. For example, if Bio-CO_2 equals 40 tonnes carbon (tC) removed from the atmosphere while 100 tC is emitted along the value chain, a factor of 0.6 could be applied at the point of combustion.

Value-chain approaches are prone to double counting. If the nation where biomass is produced accounts for GHG emissions throughout its economy, carbon stock losses and emissions due to fertilizer use, harvesting, processing, and domestic transportation emissions will already be counted in the respective sectors. If these emissions are then also included in value-chain accounts of entities using bioenergy, they would be counted twice. A system designed to avoid this problem is described below.

Variations to CoF = 1 and value-chain approaches

Two options under CoF = 1 approaches are referred to hear as 'Tailpipe' and 'Point of Uptake and Release' (POUR). Under a Tailpipe approach, the flow of CO_2 to the atmosphere from the combustion of biomass is counted so that emissions from bioenergy are treated in the same way as emissions from fossil fuels. Carbon stock changes are not measured in determining the impact of use of biomass for energy. However, if carbon stock reductions occur and are counted, this results in double counting.

The POUR approach avoids this potential for double counting while using a CoF = 1 approach. Under POUR, the total net CO_2 uptake by plants from the atmosphere is counted as negative emissions in the national report. Total net uptake includes carbon stock changes in the landscape plus carbon removed from the landscape, i.e., carbon embodied in biomass removed from the landscape for all purposes since this carbon also represents CO_2 removed from the atmosphere. The negative emissions counted for carbon in biomass combusted cancel out the positive combustion emissions, thus avoiding double counting.

[1]Theoretically factors outside the 0–1 range could result from a value-chain approach. However, it is assumed that if the factor were greater than 1, it is no likely that the biomass would be used for energy. Factors lower than 0 will only emerge if, after combustion, the CO_2 is sequestered, which would not influence the factor that would be used at the point of combustion.

In order for POUR to operate as described above where the producer nation does not account for its GHG emissions, a mechanism is needed to grant credits for net carbon uptake in such countries and enable credit transfer – presumably through entitling sales and purchases – to nations or entities with accounting obligations. Failing such a mechanism POUR collapses to Tailpipe where producing nations do not participate in the accounting system.

EU Renewable Energy Directive (EU RED) (European Union, 2009) and US Renewable Fuels Standard (US RFS2) (Federal Register, 2010) approaches include such restrictions use value-chain approaches to biofuels. Emissions along a biofuel's value chain are calculated to determine whether its emissions are sufficiently below those of fossil-fuel alternatives to qualify for use under a mandate. In addition to these calculations, both systems restrict sources or types of biomass, primarily in an attempt to avoid situations where substantial reductions in forest carbon occur to produce biomass for biofuels. In neither case is a combustion factor derived for application at the point of combustion.

In contrast to these systems, DeCicco (2009) proposes use of value-chain emissions to calculate an emission factor. Under this system, credits based on atmospheric removals are allocated to the biomass producer. After subtracting emissions due to cultivation – e.g. from fertilizer use – credits remaining are passed on to the processor. Credits remaining after subtraction of process emissions are passed on to a fuel distributor. All fuels are subject to a 1-combustion factor except insofar as net value chain credits support a lower factor.

A numerical example

In the example, a producer (nation, region, or individual) produces 83 200 t of wood pellets that are shipped to the consumer, who uses them to produce 1.0 PJ of electricity.[2] The calculation is limited to the emissions for this activity only (wood → pellets → electricity) in that occur the year of production only. There are emissions along the entire value chain because the wood must be harvested, dried, pelletised, and transported to the consumer before combustion. In the example, it is assumed that the pellets are shipped from the producer to the consumer by sea and that the consumer's facility is on the coast. Values for harvesting, processing and transportation emissions are based on values for pellets produced in Canada and shipped to Sweden (Magelli *et al*., 2009). As it is assumed that the consumer's facility is on the coast, no transportation emissions are allotted to the consumer.

The biomass is assumed to come from a forest that has been sustainably managed for multiple decades (the average harvest level is less than the net annual increment). To meet increased demand for bioenergy, the rotation length is shortened (frequency of harvest increases), which results in a period of time when the harvest exceeds the net annual increment. After this time, the management returns to a sustainable management regime although with a shorter harvest rotation. In the example, the amount harvested (87 537 Mg – the amount of biomass required to make the wood pellets) exceeds forest growth (i.e. 80 803 Mg).[3] In addition, 5% of the harvested biomass (e.g. harvesting residue left in the forest) is not shipped to the consumer. For simplicity in accounting for GHG emissions, we assume that this residue is burnt, for example, by the local population for heating and cooking. The net photosynthesis is calculated as the stock change plus the amount of biomass removed.

Table 1 illustrates the total emissions in any given year that will be counted, as well as which emissions are counted by each party, under the above options. It is assumed that the consumer is in a nation with GHG accounting obligations but the producer may or may not be.

The row 'Producer total' indicates the total GHG emissions that will be counted in a Producer nation if the nation has an accounting obligation. 'Consumer total' shows the total GHG emissions that will be counted if only the consumer is in a nation with GHG obligations. 'Global total' indicates the GHG emissions that will be accounted for if both producer and consumer have GHG obligations.

Under the KP net photosynthesis is ignored. The producer accounts for the stock loss, harvesting emissions and transportation of the pellets to the coast if it has an accounting obligation. However, if the producer is in a nation without accounting obligations (non-Annex-I country or Annex-I country that has not opted to report under Article 3.4), none of this will be accounted for. As shown in the final row, in this case no emissions will be counted since the consumer nation does not account for emissions when it combusts the biomass.

Under a Tailpipe approach, neither photosynthesis nor carbon stock changes are counted. As a result, if only the consumer accounts, over 152.000 megagrams of CO_2 (Mg CO_2) will be reported, over twice the actual emissions of close to 72.500 Mg CO_2. If both producer and consumer report, total emissions accounted will be even higher.

In POUR the producer records an estimate of net photosynthesis within its bioenergy account if it is in a nation with accounting obligations. Emissions due to harvesting, processing, and domestic transport will be reported elsewhere in his or her account. Taken together with the net photosynthesis, the producer would have net removals from the atmosphere (a net sink) of some 115 103 Mg CO_2. The consumer reports 152 460 Mg CO_2 for a combined report of 37 347 Mg CO_2. In this case, the only emissions not reported are those due to international transportation.

Where the producer does not have a reporting obligation, POUR reverts to Tailpipe. However, a primary motivation for moving to a POUR approach is to insure that carbon stock losses are accounted for without unduly discouraging use of bioenergy. To accomplish this, it is envisioned that a mechanism would be established to transfer net sequestration credits from any producer nation to nations with GHG reporting obligations. The prospect of receiving credits may serve to entice producing nations to participate in the system, with attendant

[2]The electricity generation has an assumed 65% efficiency, and the energy content of the wood is 18 GJ Mg^{-1}.

[3]It is for this reason that the emissions from wood consumption (152 460 t CO_2) are more that the removals from forest growth (148 139 t CO_2).

Table 1 Numerical example: reporting under different accounting approaches (Mg CO_2). There are two values in the global total under value chain to indicate the effect of double counting if both producer and consumer nation report emissions in producing nation. na, not applicable; in cons., in consumer account

Producer component	Actual	KP	Tailpipe	POUR	Value chain	DeCicco
Net photosynthesis	**−148 139**			**−148 139**	**in cons.**	**in cons.**
Stock change		**12 345**	**na**	**na**	**na**	**na**
Harvesting and processing						
Collection and processing	22 540	22 540	22 540	22 540	in cons.	22 540
Process waste (burnt)	8024		8024	8024	in cons.	8024
Subtotal	**30 564**	**22 540**	**30 564**	**30 564**		**30 564**
Transportation						
Transportation	2473	2473	2473	2473	in cons.	2473
Producer total	**−115 103**	**37 357**	**33 037**	**−115 103**	**0**	**33 037**
Consumer component						
Wood consumption	152 460	0	152 460	152 460	72 488	
Consumer total	**152 460**	**0**	**152 460**	**152 460**	**72 488**	**39 452**
Other components						
International transportation	**35 131**	**na**	**na**	**na**	**in cons.**	**in cons.**
Global total	**72 488**	**37 357**	**185 497**	**37 357**	**72 488 or 105 525**	**72 488**
Global total if producer does not participate		**0**	**152 460**	**152 460**	**72 488**	**72 488**

Bold values are totals for a sector.

responsibilities to track net removals and carbon stock changes across their land-use sectors.

Value-chain approaches transfer responsibility for all emissions to the user (consumer). Full emissions, including those due to international transportation, are thus reported regardless of whether a producer nation has a GHG obligation. A nonsophisticated value-chain approach (column 6) can lead to double counting if both the consumer and producer report harvesting, processing, and transportation emissions in the producing nation. In this case, more emissions are reported (105 525 Mg CO_2) than actually occur. In the more sophisticated system shown in column 7, this does not occur. In this system, the correct emissions will be reported regardless of whether the producer nation has a reporting obligation. If it does, that nation will report 33 037 Mg CO_2 and the consumer will report 39 452 Mg CO_2. If the producer does not report, the consumer will report the full 72 488 Mg CO_2 that arise in the example.

The criteria

This article evaluates the alternative accounting systems using three criteria: comprehensiveness over space and time, simplicity, and scale-independence. Comprehensiveness over space and accuracy in time is a measure of environmental integrity and is here used to refer to the degree to which an accounting system counts emissions once and only once.

Following Einstein's dictum, it is always preferable to use a system that is 'as simple as possible, but no simpler'. Simplicity is a main reason that CoF = 0 factor approach was recommended and selected. The approach requires only the measurement of carbon stock changes, for which there is considerable experience from forest inventories. However, under real-world conditions – i.e., the fact that many nations do not report under the KP – this approach may be 'simpler than possible' given the importance of achieving reasonable coverage over space.

Scale-independence encompasses not only the ability for a system to be used at varies scales but also the compatibility of results when this is done. Scale-independence is important because accounting systems may be used not only at the national level but also at sectoral levels and by entities subject to GHG limitations. Scale-independence is particularly challenging in cases where measurements of forest-carbon stock change form part of the system because such measurements give very different results at different scales. For instance, whereas annual forest regrowth at the national or landscape level can exceed or fully compensate for removals for bioenergy, this cannot happen at the stand level within an accounting period. Consequently, while a nation might report no net emissions due to use of bioenergy, an entity whose biomass came from a particular forest might report emissions.

Incentives evaluation

This article evaluates accounting approaches from the perspective of their impact on three goals that generally are pursued in conjunction with use of bioenergy: increase energy security, stimulate rural economies, or reduce GHG emissions. Food security is an issue for many nations, and stakeholders may also be interested in preserving forests and maintaining habitat and other environmental services, including in the context of reducing vulnerability to climate change. While some goals are generally mutually supportive or operate jointly, other goals tend to compete with one another. Goals that tend to operate jointly are food security, energy security, and stimulation of rural economies. These goals are thus handled together in this article.

Promotion of these goals generally seems to threaten preservation of forests and linked environmental services. For example, in absence of an increase in crop yield, use of biomass for energy may reduce biomass available for food and cause an expansion of cropland into forests. Therefore, preservation of forests is handled separately. Use of bioenergy to achieve GHG goals tends to threaten both food security and forest preservation, while working jointly with energy security and stimulation of rural economies. For these reasons, it also is treated separately.

Results

This section first looks at the impacts of the various systems on afforestation, deforestation, and emissions globally. Following this, the approaches are evaluated against the three criteria and then in regard to their impacts on national and stakeholder goals.

Global implications

To illustrate the global implications of the different accounting options, we will use an estimate of the global afforestation, deforestation and forest management and emissions that result from the GLOBIOM model (Havlík *et al.*, 2011). GLOBIOM provides estimates of land-use competition between the major land-based production sectors and assesses the land-use change (LUC) impacts of biofuel production scenarios in terms of afforestation and deforestation. This study developed the LUC events for four future scenarios of biofuel production using a partial equilibrium economic model. The four biofuel scenarios are as follows:

a. No biofuels are produced
b. Baseline (60% of biofuels that are produced are first generation and 40% are second generation);
c. Only first generation biofuels are produced; and
d. Only second-generation biofuels are produced.

As well, for the second-generation biofuels, three options were evaluated. Second-generation biofuels are created from short rotation forestry on:

i agricultural land;
ii marginal land; or from
iii existing forest lands.

Havlík *et al.* (2011) estimated the CO_2 emissions from LUC for the live biomass only assuming that agricultural practices do not have an impact on soil carbon emissions, and in the case of deforestation, the total carbon contained in above and below ground living biomass is emitted.

For the purposes of this article, we will focus on their results from the baseline scenario with the option that the biomass for second-generation biofuels comes from existing agricultural land. To the emissions from LUC, we add the emissions due to changes in dead wood litter and soil organic carbon. These emissions/removals are calculated as the difference of carbon stock in each of the three pools, before and after conversion. The assessment has been made based on default values provided in the 2006 IPCC Guidelines (IPCC, 2006). Default values are provided for carbon stocks in each pool and for each land use. The calculations are done at the regional level for eleven regions (Central-East Europe, Former Soviet Union, Latin America, Mid-East and North Africa, North America, Other Pacific Asia, Pacific OECD, Planned Asia-China, South Asia, sub-Saharan Africa, Western Europe).

Afforestation considers only conversion to short rotation plantations, whereas deforestation is the conversion of natural or managed forests to other land uses, such as cropland and grassland. It is assumed that changes in the litter and dead wood pool occur only with deforestation, whereas no change is assumed in the other cases. A carbon loss equal to the amount of carbon in the litter and deadwood is accounted for when a forest is cut and converted to cropland or grassland. This assumption is based on an IPCC Tier 1 approach, which considers no accumulation of litter and deadwood in cropland and grassland. Therefore, deforestation produces a loss of carbon in these two pools. Initial values of litter and deadwood carbon in forests were derived from table 2.2 of the 2006 IPCC Good Practice Guidance (IPCC, 2006) and table 4.2.2 of the 2003 IPCC Guidelines (IPCC, 2003). Regarding afforestation, the data only include conversions to short rotation plantations which accumulate very little litter and deadwood compared to cropland or grassland. Due to this reason, we conservatively assumed that no carbon is accumulated in litter and deadwood when land is converted to short rotation plantations. The emissions/removals in soil are calculated based on equation 2.25 and default factors in the 2006 IPCC Guidelines. According to this method, the carbon stock in the soil, under a specific land use, is calculated by first selecting a so-called reference soil carbon stock (SOCREF, table 3.3, IPCC, 2006). The SOCREF represents the carbon stock in reference conditions, i.e. native vegetation that is not degraded or improved. The SOCREF is the value that we used as soil carbon stock in Forestland. For other land uses, the soil carbon stock is calculated by multiplying the SOCREF for default factors that are specific for each land use, land management and level of organic inputs (tables 5.5, 5.10, and 6.2, IPCC, 2006). Default SOCREF values were chosen among the figures reported for high activity clay soils which include most of the existing soil types.

Finally ,we include estimates of the GHG emissions from the cultivation, processing, transport and distribution of biofuels, the non-LUC components (to the emissions due to LUC. For example, if corn is transformed into ethanol, then the non-LUC components are the emissions for using machinery to plow the land, transport the biomass to the ethanol plant, and distribute the ethanol to the consumer. As well, there are emissions from the use of inorganic nitrogen-based fertilizers that must be included. These emissions are usually included in a LCA of the impacts of biofuels. The emission factors are listed in Table 2. See Bird *et al.* (2011) for a complete discussion of the calculation methodology.

Figure 5 shows graphically the cumulative emissions from biofuels to 2030 by region under different accounting systems. It shows that under the IPCC accounting system (unmodified CoF = 0), consuming regions (CPA, NAM, SAS, and WEU) benefit greatly and will claim an emission reduction. On the other hand, Latin America, the modeled main producer, is burdened with a large amount of emissions under the IPCC approach. Using POUR accounting, Latin America still has large emissions, but it does not underwrite the emission reductions of the consuming regions. Since there is so large a swing in emissions, it is clear that if POUR is adopted, then emission targets would need to be completely renegotiated.

Evaluations against criteria

Comprehensiveness over space and time. Under conditions in which carbon stock reductions in developing countries are not accounted for within a GHG limitation regime, the CoF = 0 approach rates poorly in terms of comprehensiveness over space. Emissions at the point of combustion of biomass are not counted anywhere in the world, and emissions due to carbon stock reductions are counted only in nations that have accepted GHG limitations under the KP.

CoF = 1 approaches are significantly more comprehensive than unmodified CoF = 0 approaches. If the biomass producing nation does not participate in accounting, uncounted emissions include those from oxidation of biomass left in forests, soil carbon losses, and decay of biomass that was harvested but not converted for use for bioenergy. These are much smaller than emissions that fail to be counted under the same circumstances under an unmodified CoF = 0 approach.

However, if net atmospheric uptake of CO_2 by the land sector in a producer nation is not counted, accuracy will not be achieved. The inaccuracy will be one of over-counting, rather than under-counting emissions, except where drainage of wetlands occurs. In this case, both Tailpipe and POUR may underestimate emissions. As noted earlier, without a mechanism to grant and transfer credits for net atmospheric uptake, POUR reverts to Tailpipe, removing the motivation to use POUR. With such a mechanism accounting will be accurate over both time and space to the extent that credits are transferred to entities with obligations.

The comprehensiveness of value-chain approaches is different than for the CoF = 0 or 1 approaches. On the one hand, such systems tend to be quite comprehensive because they include in the bioenergy account emissions not included in the other approaches, e.g., emissions due to biomass cultivation, its conversion to an energy product, and its transportation to users. However, the spatial coverage of the EU RED is not high. First, it does not include emissions on land that does not change its status. This approach is prone to spatial omissions because, for instance, a forest might move from 80% tree coverage to 50% tree coverage while still remaining its forest status. The US RFS2 approach is not prone to these omissions because wood, except for residues and precommercial thinnings, can only come from natural forests threatened by fire. Second, the EU RED does not include emissions due to indirect land-use change (iLUC), and its attempt to manage these through an incentive mechanism is unlikely to be successful(Lange, 2011). The US RFS2 has attempted to include iLUC by using modeling to estimate the amount associated with each biomass-conversion combinations, e.g., ethanol from corn and ethanol from sugar cane. To the degree

Table 2 Emission factors for life-cycle emissions from biofuels. Emissions do not include combustion or land-use change. Ranges are taken from the range reported in various life-cycle assessment studies. The variation may be caused by differences in system boundaries, cultivation practices and crop yields, use of co-products, allocation of emissions to co-products, etc. For more information, see Cherubini *et al.* (2009)

Fuel	Emissions (g CO_2 eq MJ^{-1})	Range (%)	Source
Biodiesel, palm	54.0	± 35	European Union (2009)
Biodiesel, rape	46.0	± 35	European Union (2009)
Biodiesel, soy	50.0	± 35	European Union (2009)
Biodiesel, wood, farmed	4.0	± 57	European Union (2009)
Ethanol, cane	24.0	± 20	European Union (2009)
Ethanol, corn	37.0	± 30	European Union (2009)
Ethanol, wood, farmed	6.0	± 33	European Union (2009)

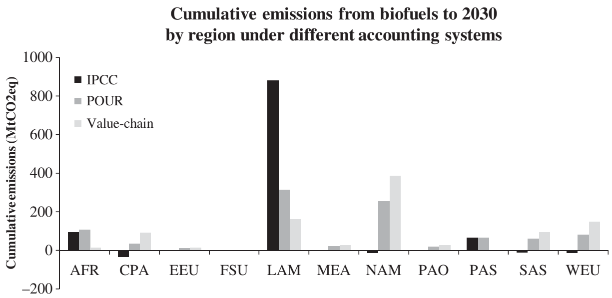

Fig. 5 Cumulative emissions due to biofuels to 2030 by region under different accounting systems. Abbreviations: AFR, sub-Saharan Africa; CPA, centrally planned Asia; EEU, Central and Eastern Europe; FSU, Former Soviet Union; LAM, Latin America; MEA, Middle East and North Africa; NAM, North America; PAO, Pacific OECD; PAS, other Pacific Asia; SAS, South Asia; WEU, Western Europe. Calculations are made by the authors but are based on data from Havlík *et al.* (2011). Please see the body of this article for details of the calculations.

that such modeling is accurate, its spatial coverage should be high.

The EU RED due to its focus on biofuels and the likelihood they come from annual crops has not addressed issues of timing. The US RFS2 attempts to achieve a reasonable degree of accuracy of timing through its restrictions to annual crops, residues and wood from plantations established as of 2007 and areas at high risk of fire. How well this will function will only be clear once significant amounts of woody biomass are used for biofuels.

Simplicity. The tailpipe system is probably the simplest of all approaches, requiring only that bioenergy emissions or the amount of biomass consumed for bioenergy be measured and converted to CO_2. Due to the overestimation that occurs, however, it is unlikely to be adopted. A POUR approach has better chances of adoption but is more complicated. Under the real-world circumstances of partial adoption of GHG limitation obligations, however, a POUR-type approach may be 'as simple as possible' as it ensures that emissions due to combustion of biomass in nations with GHG obligations are counted even if attendant stock reductions are not.

Point of Uptake and Release requires measuring carbon stock changes and reporting amount of biomass removed from the landscape in the producing nation plus measuring bioenergy combustion emissions in the consumer nation. The approach will raise challenges when applied to products that can be used either for food, feed, or fuel. As there is no suggestion to date that emissions due to human food consumption be included in GHG obligations, carbon stock changes due to production of, e.g., oils, sugar crops, and grains used for food would need to be separated out from carbon stock changes due to production for feeds, energy, or other products whose emissions will be counted where and when they occur. An additional complication is that since use of oils or grains may only be determined in the consuming nation, it will be necessary to track origin of dual-purpose biomass. This is necessary under POUR to determine the amount of carbon from a producing country embodied in nonfood products.

Value-chain approaches are more complicated than 0- and 1-combustion factor approaches due to their high data needs. They require information not only on biomass and attendant emissions but also information on emissions due to cultivation, conversion, and transportation used for a particular biofuel type or batch. Such information is needed from each nation that is a source of biomass.

Scale independence. It was expected that the CoF = 0 approach would be scale-independent as carbon stock levels can be measured from the stand level up to the national level. However, measurements of forest-carbon stock changes give very different results at different scales. Thus, the CoF = 0 approach has not turned out to be scale-independent. The POUR approach shares this weakness, but the tailpipe method is fully scale independent.

Value-chain approaches are not inherently scale-independent. They can only achieve this through use of national-level estimates of GHG emissions at each step along the value chain together with assumptions regarding the share of such emissions attributable to each batch of a biofuel. Given that, as previously mentioned, the destiny of agricultural products may only be determined in the consumer nation, this could prove extremely challenging. The value chains used in the EU RED and the US RFS2 are both batch-based and resulting emission calculations do not enter into national GHG accounting.

Summary. Table 3 summarizes the above evaluation of accounting approaches against the chosen criteria. While Tailpipe performs relatively well against all criteria, due to the over counting of emissions it may not be practical. The next best option is POUR which is more comprehensive than 0-combustion approaches and less complex than value-chain approaches. It shares the scale-independent problems of CoF = 0 approaches. Value-chain approaches are rated lower because of their complexity.

Accounting systems can support or hinder stakeholder goals because they tend to provide incentives or disincentives for specific actions. For example, we have already suggested that 0-combustion factor accounting approaches provide strong incentives for energy consumers to use bioenergy to meet GHG obligations, particularly if the carbon stock losses occur in another country or are uncounted.

Impacts of accounting system on stakeholder goals

Stimulation of rural economies and food security. A CoF = 0 factor approach provides a strong stimulus to use bioenergy. This stimulates production of both agricultural and forest biomass (Cortez *et al*., 2010). However, this stimulus may result in dedication of food and feed crops to energy and food and feed price increases. Dedication of food and feed crops to energy may reduce food security and lead to increased need for food imports in nations where agricultural supply is not sufficient to meet both demands (Pimental *et al*., 2009). Price increases tend to benefit farmers but can burden general populations, particularly its poorer segments.

Having the energy consumer account for GHG emissions from bioenergy combustion, as happens under CoF = 1 approaches, removes the incentive to use more bioenergy than justified by the emission reductions it achieves. Since in most applications biomass results in more CO_2 emitted per unit of energy produced than fossil fuels, use of bioenergy may be discouraged where entities are faced with GHG reduction obligations. As a consequence, the CoF = 1 approaches tend to decrease demand for biomass for energy. Thus, they neither stimulate rural economies nor result in food and feed prices increases or food security difficulties. The Tailpipe characterized by all of the effects. The POUR approach may overcome the lack of stimulation to rural economies through a mechanism that provides credit for atmospheric removal of CO_2 by biomass. The extent to which credits would overcome the disincentive to use bioenergy, and thus stimulate rural economies would depend on details of the transfer rules. Thus, until such a program is designed, the impacts cannot be evaluated.

Value-chain approaches have been implemented in conjunction with mandates to reduce GHGs and the mandates rather than the accounting system are driving increased use of bioenergy and thus stimulating rural economies. Insofar as the goal of value-chain approaches is to align use of bioenergy with its emission consequences, value-chain approaches are more likely to resemble CoF = 1 than CoF = 0 approaches.

GHG reductions. Because of the current and expected incomplete participation in binding GHG targets, together with the fact that bioenergy producers incur no costs for their emissions, the CoF = 0 accounting approach fails to promote GHG reductions. In fact, it may actually result in more emission than the continued use of fossil fuels (Havlík *et al*., 2011). The CoF = 1 approaches can be effective ways to control GHG emissions because bioenergy producers do incur costs for emissions. The fact that combustion of biomass generally generates more CO_2 emissions to produce a unit of energy than the combustion of fossil fuels increases the difficulty of achieving the goal of reducing GHG emissions by using woody biomass in the short term (Walker *et al*., 2010; Zanchi *et al*., 2010; McKechnie *et al*., 2011; Repo *et al*., 2011). A POUR approach with a crediting mechanism might be particularly effective in addressing emissions from the land sector. If a crediting mechanism induces nations without GHG obligations to track net atmospheric removals as a condition for receiving and selling credits, there would be a powerful incentive for them to move to practices in which carbon stock reductions are lower than biomass removed from the landscape, e.g., less is harvested annually than grows.

Making users responsible for value-chain GHGs can translate into incentives both to produce and to pur-

Table 3 Evaluation of accounting approaches against criteria

	Criteria			Combined	
Accounting approach	Space and time	Simplicity	Scale	Evenly weighted	Space and time double weight
Combustion factor = 0 approaches (CoF = 0)					
Unmodified	Low	High	Low	Medium	Medium
Existing+emissions correction	Low	Low	Low	Low	Low
Existing+policy overlay	High to low	Medium to low	Low	Medium to low	High to low
Combustion factor = 1 approaches (CoF = 1)					
Tailpipe	Medium	High	High	High	High
POUR	High	Medium	Low	Medium	Medium
Value-chain approaches					
All	Very high	Low	Low	Medium	High

chase biomass with the lowest GHG profiles. This, however, only will happen under value-chain approaches where the profile directly impacts costs, as would happen under a DeCicco-type system where the lower the GHG profile the fewer permits to emit required. Under these circumstances, value-chain approaches may be the most effective way of reducing GHG emissions associated with the use of bioenergy.

Preservation of forests. The extent to which an accounting approach preserves forests is often closely related to its ability to reduce GHG emissions. The 0-combustion factor approach, for example, does neither very successfully, whereas Tailpipe does both effectively.

As the tailpipe approach discourages the use of bioenergy, it can be considered supporting preservation of forests just as it supports reductions in GHG emissions from biomass. In POUR, on the other hand, credits may be received for removals embodied in harvested wood. This leads to the assumption that there would be a strong incentive to harvest. However, credits are received only for carbon in wood sold minus carbon stock losses. Hence, POUR may provide an incentive to sustainable forest management. The actual impact of POUR on forest preservation could, however, only be determined once a program with sufficient detail was developed to enable economic analyses well beyond the scope of this study.

The impact of a value-chain approach to bioenergy on forests will depend greatly on the specifics of its design as well as whether emissions calculated along the value chain are used to determine a combustion factor or it is used in conjunction with mandates. The EU RED approach allows significant degradation of natural forests and even replacement of natural forests with plantations as long as they meet specific criteria. The US RFS2, by restricting use of woody biomass to residues, slash, precommercial thinnings, and forests planted by hand or machine on land cleared prior to 2007 is very likely to prevent loss or degradation of forests.

A major issue is how a value-chain approach will deal with the problem that arises in the case of woody biomass: emissions occur in the near term but compensating regrowth, particularly at the batch level, can take decades to centuries. If little or no attention is paid to this problem, as appears to be the case in the EU RED, a value-chain approach may not preserve forests effectively. Currently, the mandates play a larger role in the impact on forest preservation, than the accounting system.

Summary. Table 4 provides a qualitative summary of our evaluation of accounting approaches in support of stakeholder holders' goals. We find that the unmodified CoF = 0 approach behaves very poorly. A CoF = 0 approach restricted to trading partners that have committed to a GHG limitation rates well across all goals. However, given that it would leave most nations outside of the system, as well as potential objections on free trade grounds, it may not be a desirable solution.

The Tailpipe approach does well for most goals but given its strong discouragement of use of bioenergy together with its over counting of emissions it may also not be a desirable choice. A POUR approach has potential but the design of a crediting and credit-transfer mechanism, as well as the response of nations without GHG obligations, would be critical in performance characteristics. Similarly, a DeCicco-type value-chain approach seems to have considerable potential. As a value-chain approach it brings use of bioenergy into line with its GHG emissions. Thus, while it will encourage use of bioenergy where GHG profiles are favorable, it is unlikely to encourage bioenergy at levels that would unduly affect food and feed prices.

Discussion and conclusions

The current accounting system for emissions from bioenergy gives entities with GHG obligations an incentive to use bioenergy at the expense of maintenance of carbon stocks. In this article, we describe and examine alternative approaches that could potentially redress this system weakness.

The problem arises because the KP's accounting of bioenergy is a '0-combustion factor' (CoF = 0) approach. Emissions from the combustion of biomass for energy are not accounted in the energy sector, but in the land-use sector as carbon stock losses. However, in reality, many carbon stock losses are not accounted for. Many countries do not have GHG targets and some countries that have them do not include carbon stock changes in forests remaining forests, or even from deforestation where net forest area remains steady or increases. In this way, the KP provides an incentive for KP compliant nations to obtain biomass for energy from nations without KP obligations or other sources not accounted for. The EU-ETS in particular provides energy producers with a powerful incentive to use bioenergy regardless of its carbon stock implications as carbon stock changes play no part in the EU-ETS.

This article describes alternative approaches to accounting for bioenergy emissions and proposes that all alternatives fall into one of three categories: (1) application of a 0-combustion factor to bioenergy emissions at the point of combustion (the current approach); (2) CO_2 emissions at combustion are similar to fossil fuels (1-combustion factor approach); and (3) value-chain approaches in which bioenergy consumers are responsible for net GHG emissions generated along the bioenergy value chain and these emissions can be used to calculate a combustion factor between zero and one.

This article examines several options within each of these categories, including use of policy overlays or correction factors in connection with a CoF = 0 approach; counting only emissions (Tailpipe) or also counting atmospheric uptake of CO_2 (POUR) within the CoF = 1 group; and value chains that do and do not use the calculated emissions to determine a combustion factor. The value chains used in the EU RED and US RFS2 do not use calculated emissions to determine a combustion factor, whereas an approach proposed by DeCicco does.

This article points out that the value-chain approaches differ from the other two types of approach in two significant ways. They encompass not only emissions from combustion of biomass and carbon stock losses but also emissions from cultivation of biomass and its conversion and transportation. Second, unlike any of the 0- or 1-combustion factor approaches, they hold a consuming nation responsible for emissions that occur outside of its national boundaries.

Finally, this article evaluates the accounting options against general criteria and selected stakeholder goals. The general criteria are comprehensiveness over space and time, simplicity, and scale-independence. Stakeholder goals are stimulation of rural economies, food prices, and energy security; reductions of GHG emissions; and preservation of forests.

With regard to accuracy of accounting over space and time, value chain and combustion CoF = 1 approaches tend to perform better than CoF = 0 approaches, significantly increasing the fraction of emissions due to bioenergy captured in the accounting system. Emissions that would not be included in CoF = 1 systems are those due to soil and litter pool carbon losses and in the case of drainage of wetlands additional GHG emissions. However, there is a trade-off. Except for Tailpipe, CoF = 1 and value-chain approaches are not as simple as the unmodified 0-combustion factor approaches.

In general, CoF = 0 approaches, by encouraging use of bioenergy, tend to stimulate rural economies but do poorly against other goals, with the exception of restricting trading partners to nations with GHG limitation obligations. The CoF = 1 options have the opposite tendencies. They tend to discourage use of bioenergy and thus fail to stimulate rural economies. POUR may overcome this through inclusion of a credit and credit-transfer mechanism. Producer countries would receive credits for net atmospheric uptake of CO_2 which could be sold to bioenergy consumers. If such a mechanism were available to all nations, POUR could be effective in controlling GHG emissions because it would encourage maintaining carbon stocks while providing biomass for energy. Value-chain approaches are theoretically neutral between use of bioenergy and continued use of fossil fuels and therefore would tend not to encourage use of bioenergy due to its high emissions per unit of energy produced. However, to date, value-chain approaches have been used in conjunction with mandates that drive use of bioenergy, and the specifics of the programs have determined the outcomes on stakeholder goals.

Table 4 Qualitative review: accounting options vs. stakeholder goals. The evaluation of POUR assumes mechanism to award and transfer credits from producer to consumer

Accounting system	Stimulate rural economies	Protect food security	Reduce GHG emissions	Preserve forests
Combustion factor = 0 approaches (CoF = 0)				
Unmodified	High	Low	Low	Low
Existing + emissions correction	Lower than unmodified	Higher than unmodified	Uncertain	DPD
Existing + limited sources	Likely high	Uncertain	DPD	DPD
Existing + limited trading partners	High	High	High	High
Combustion factor = 1 approaches (CoF = 1)				
Tailpipe	Low	High	High	Low
POUR	DPD, potentially high	Potentially low	DPD, potentially high	DPD, potentially high
Value-chain approaches				
EU RED	DM	Low	Medium	Medium
US RFS2	DM	Low	High	High
DeCicco-type	Medium to high	Medium	High	Likely high

POUR, Point of Uptake and Release; GHG, greenhouse gas; DPD, depends on program details; DM, depends on mandate.

Both POUR and a DeCicco approach seem to hold considerable promise to do well again general criteria and stakeholder goals but until programs using them are further developed impacts remain uncertain.

Acknowledgements

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- Bioenergy, sustainability and trade-offs: Can we avoid deforestation while promoting bioenergy? EuropeAid/ENV/2007/143936/TPS
- Climate Change: Terrestrial Adaptation& Mitigation in Europe (CC-TAME) FP7–ENV-2007-1 Grant #212535

The objective of the first of these projects is to contribute to sustainable bioenergy development that benefits local people in developing countries, minimizes negative impacts on local environments and rural livelihoods, and contributes to global climate change mitigation. The project will achieve this by producing and communicating policy relevant analyses that can inform government, corporate, and civil society decision-making related to bioenergy development and its effects on forests and livelihoods. The project is managed by CIFOR and implemented in collaboration with the Council on Scientific and Industrial Research (South Africa), Joanneum Research (Austria), the Universidad Autónoma de México, and the Stockholm Environment Institute.

CC-TAME's primary objective is to bring land-use modeling to the level of sophistication available in energy modeling and thereby support policy-makers and other stakeholders in evaluations of the impacts, efficiency, and effectiveness of land-use options, including production of biomass for energy. The project will link state-of-the art climate, agricultural soil and yield, forest dynamics, and socio-economic models. The GLOBIUM modeling used within CC-TAME reveal the impact of different accounting approaches on different areas in the world.

Disclaimer

The views expressed herein are those of the authors only. They should in no way be taken to reflect the official opinion of the institutions for which the authors work or organizations with which the authors may be affiliated.

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Frontiers in Ecology and the Environment

Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

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Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions?

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It has been suggested that thinning trees and other fuel-reduction practices aimed at reducing the probability of high-severity forest fire are consistent with efforts to keep carbon (C) sequestered in terrestrial pools, and that such practices should therefore be rewarded rather than penalized in C-accounting schemes. By evaluating how fuel treatments, wildfire, and their interactions affect forest C stocks across a wide range of spatial and temporal scales, we conclude that this is extremely unlikely. Our review reveals high C losses associated with fuel treatment, only modest differences in the combustive losses associated with high-severity fire and the low-severity fire that fuel treatment is meant to encourage, and a low likelihood that treated forests will be exposed to fire. Although fuel-reduction treatments may be necessary to restore historical functionality to fire-suppressed ecosystems, we found little credible evidence that such efforts have the added benefit of increasing terrestrial C stocks.

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Various levels of tree removal, often paired with prescribed burning, are a management tool commonly used in fire-prone forests to reduce fuel quantity, fuel continuity, and the associated risk of high-severity forest fire. Collectively referred to as "fuel-reduction treatments", such practices are increasingly used across semiarid forests of the western US, where a century of fire suppression has allowed fuels to accumulate to levels deemed unacceptably hazardous. The efficacy of fuel-reduction treatments in temporarily reducing fire hazard in forests is generally accepted (Agee and Skinner 2005; Ager *et al.* 2007; Stephens *et al.* 2009a) and, depending on the prescription, may serve additional management objectives, including the restoration of native species composition, protection from insect and pathogen outbreaks, and provision of wood products and associated employment opportunities.

In a nutshell:

- Carbon (C) losses incurred with fuel removal generally exceed what is protected from combustion should the treated area burn
- Even among fire-prone forests, one must treat about ten locations to influence future fire behavior in a single location
- Over multiple fire cycles, forests that burn less often store more C than forests that burn more often
- Only when treatments change the equilibrium between growth and mortality can they alter long-term C storage

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Recently, several authors have suggested that fuel-reduction treatments are also consistent with efforts to sequester C in forest biomass, thus reducing atmospheric carbon dioxide (CO_2) levels (Frinkral and Evans 2008; Hurteau *et al.* 2008; Hurteau and North 2009; Stephens *et al.* 2009b). It is argued that short-term losses in forest biomass associated with fuel-reduction treatments are more than made up for by the reduction of future wildfire emissions, and thinning practices aimed at reducing the probability of high-severity fire should therefore be given incentives rather than be penalized in C-accounting programs. This is an appealing notion that aligns the practice of forest thinning with four of the most pressing environmental and societal concerns facing forest managers in this region today – namely, fire hazard, economic stimulus, so-called forest health, and climate-change mitigation. However, we believe that current claims that fuel-reduction treatments function to increase forest C sequestration are based on specific and sometimes unrealistic assumptions regarding treatment efficacy, wildfire emissions, and wildfire burn probability.

In this paper, we combine empirical data from various fire-prone, semiarid conifer forests of the western US (where issues of wildfire and fuel management are most relevant) with basic principles of forest growth, mortality, decomposition, and combustion. Our goal is to provide a complete picture of how fuel treatments and wildfires affect aboveground forest C stocks by examining these disturbance events (1) for a single forest patch, (2) across an entire forest landscape, (3) after a single disturbance, and (4) over multiple disturbances. Finally, we consider how wildfire and/or fuel treatments could initiate alternate equilibrium states

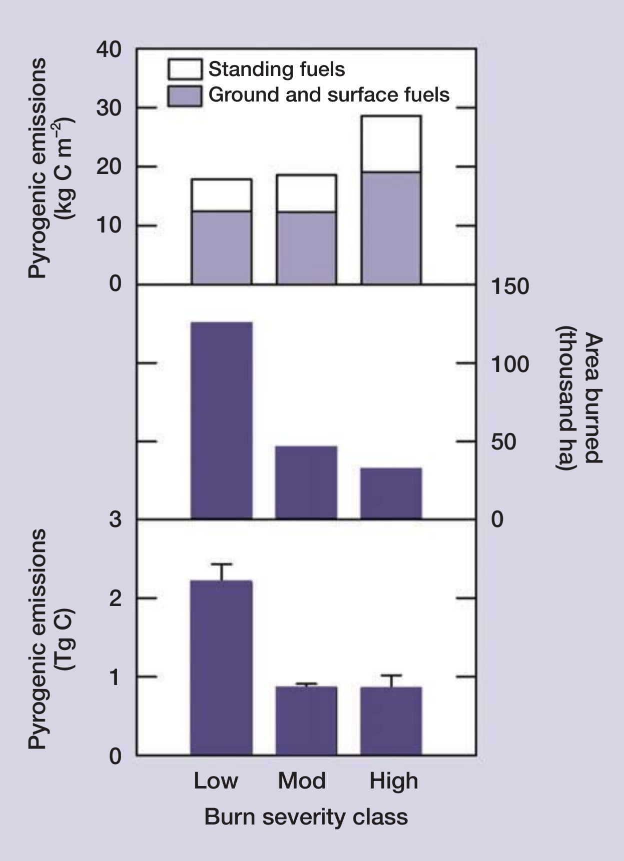

Figure 1. *Sources of pyrogenic emissions across the 2002 Biscuit Fire in southwestern Oregon and northern California. Because most emissions arise from the combustion of ground and surface fuels, pyrogenic emissions from high-severity fires were only one-third higher than those in low-severity fires. Moreover, because most of the fire burned with low severity, the contribution of high-severity fire to total emissions was only about 20%. The Biscuit Fire burned over a mosaic of young, mature, and old-growth stands of mixed conifer growing across a climate gradient ranging from mesic to semiarid. Methods are described in Campbell* et al. *(2007).*

and change the long-term capacity of a forest to accumulate biomass.

■ Immediate stand-level C losses attributed to wildfire and fuel-reduction treatments

Because fuel-reduction treatments are generally designed to reduce subsequent wildfire severity, rather than to preclude fire entirely, it is important to compare the C losses incurred under both high- and low-severity fire scenarios. The amount of biomass combusted in a high-severity crown fire is unquestionably greater than the amount combusted in a low-severity surface fire. The difference, however, is smaller than that suggested by some authors (eg Hurteau *et al.* 2008). Even under the most extreme fuel-moisture conditions, the water content of live wood frequently prohibits combustion beyond surface char; this is evident in the retention of even the smallest canopy branches after high-severity burns (Campbell *et al.* 2007). Moreover, the consumption of fine surface fuels (ie leaf litter, fallen branches, and understory vegetation), though variable, can be high even in low-severity burns. As shown in Figure 1, Campbell *et al.* (2007) found that patches of mature mixed-conifer forest in southwestern Oregon that were subject to low-severity fire (ie 0–10% overstory mortality) released 70% as much C per unit area as did locations experiencing high-severity fire (ie > 80% overstory mortality). When scaled over an entire wildfire perimeter, the importance of high-severity fire in driving pyrogenic emissions is further diminished because crown fires are generally patchy while surface fires are nearly ubiquitous (Meigs *et al.* 2009).

According to Campbell *et al.* (2007), less than 20% of the estimated 3.8 teragrams of C released to the atmosphere by the 2002 Biscuit Fire in the Siskiyou National Forest of southern Oregon and northern California (Figure 1) arose from overstory combustion. Simply put, because most pyrogenic emissions arise from the combustion of surface fuels, and most of the area within a typical wildfire experiences surface-fuel combustion, efforts to minimize overstory fire mortality and subsequent necromass decay are limited in their ability to reduce fire-wide pyrogenic emissions.

The total amount of biomass combusted, or taken off-site, during a fuel treatment is, by definition, a prescribed quantity and can vary widely depending on the specific management objective and techniques used. A review of fuel-reduction treatments carried out in semiarid conifer forests in the western US reveals that aboveground C losses associated with treatment averaged approximately 10%, 30%, and 50% for prescribed fire only, thinning only, and thinning followed by prescribed fire, respectively (WebTable 1). By comparison, wildfires burning over comparable fire-suppressed forests consume an average 12–22% of the aboveground C (total fire-wide averages reported by Campbell *et al.* [2007] and Meigs *et al.* [2009], respectively).

Given that both fuel-reduction treatments and wildfire remove C from a forest, to what degree does the former reduce the impact of the latter? To test this question, Mitchell *et al.* (2009) simulated wildfire combustion following a wide range of fuel-reduction treatments for three climatically distinct conifer forest types in Oregon. As illustrated in Figure 2, fuel treatments were effective in reducing combustion in a subsequent wildfire, and the greater the treatment intensity, the greater the reduction in future combustion. However, even in the mature, fire-suppressed ponderosa pine (*Pinus ponderosa*) forest, protecting one unit of C from wildfire combustion typically came at the cost of removing three units of C in treatment. The reason for this is simple: the efficacy of fuel-reduction treatments in reducing future wildfire emissions comes in large part by removing or combusting surface fuels ahead of time. Furthermore, because removing fine canopy fuels (ie leaves and twigs) practically necessitates removing the branches and boles to which they are attached, conventional fuel-reduction treatments usually remove more C from a forest stand than would a wildfire burning in an untreated stand. In an extreme modeling scenario, wherein only fine-surface fuels were removed, subsequent avoided combustion did slightly exceed treatment removals (Figure 2, circles). However, this marginal gain amounted to less than 0.03% of the total C stores, which is, practically speaking, a zero-sum game.

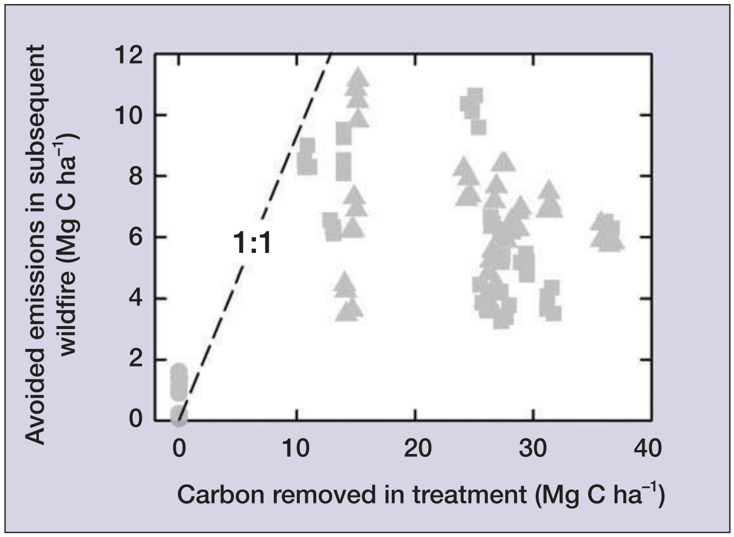

Figure 2. *Simulated effectiveness of various fuel-reduction treatments in reducing future wildfire combustion in a ponderosa pine (*Pinus ponderosa*) forest. In general, protecting one unit of C from wildfire combustion came at the cost of removing approximately three units of C in treatment. At the very lowest treatment levels, more C was protected from combustion than removed in treatment; however, the absolute gains were extremely low. Circles show understory removal, squares show prescribed fire, and triangles show understory removal and prescribed fire. Simulations were run for 800 years with a treatment-return interval of 10 years and a mean fire-return interval of 16 years. Forest structure and growth were modeled to represent mature, semiarid ponderosa pine forest growing in Deschutes, Oregon. Further descriptions of these simulations are given in Mitchell* et al. *(2009).*

Wildfire probability, treatment life span, and treatment efficacy across a landscape

Any approach to C accounting that assumes a wildfire burn probability of 100% during the effective life span of a fuel-reduction treatment is almost certain to overestimate the ability of such treatments to reduce pyrogenic emissions on the future landscape. Inevitably, some fraction of the land area from which biomass is thinned will not be exposed to any fire during the treatment's effective life span and therefore will incur no benefits of reduced combustion (Rhodes and Baker 2008). On the other hand, assuming that landscape-wide burn probabilities apply to all of the treated area is almost certain to underestimate the influence of treatment on future landscape combustion. This is because doing so does not account for managers' ability to target treatments toward probable ignition sources or the capacity of treated areas to reduce burn probability in adjacent untreated areas (Ager *et al.* 2010).

Among fire-prone forests of the western US, the combination of wildfire starts and suppression efforts result in current burn probabilities of less than 1% (WebTable 2). Given a fuel-treatment life expectancy of 10–25 years, only 1–20% of treated areas will ever have the opportunity to affect fire behavior. Such approximations are consistent with a similar analysis reported by Rhodes and Baker (2008), who suggested that only 3% of the area treated for fuels is likely to be exposed to fire during their assumed effective life span of 20 years. Extending treatment efficacy by repeated burning of understory fuels could considerably increase the likelihood of a treated stand to affect wildfire behavior, but such efforts come at the cost of more frequent C loss.

A more robust, though more complicated, evaluation of fuel-treatment effect on landscape burn probability is achieved through large-scale, spatially explicit fire spread simulations (Miller 2003; Syphard *et al.* 2011). In one such simulation, representing both the topography and distribution of fuels across a fire-prone and fire-suppressed landscape in western Montana, Finney *et al.* (2007) showed how strategically treating as little as 1% of the forest annually for 20 years reduced the area impacted by a single large wildfire (expected to occur about once on this landscape in that 20-year period) by half, and how strategically treating 4% of the forest annually reduced the area impacted by a single large wildfire by > 95% (Figure 3). However, even when the treatment effect was highest, the protection of each hectare of forest from fire came at the cost of treating nearly 10 hectares (note the axis scales in Figure 3). Such inefficiencies come not from the treatments' efficacy in curtailing fire spread; rather, they stem from the rarity of wildfire. Put another way, the treatment of even modest areas may lead to high fractional reductions in the area impacted by high-severity wildfire, but because such fires rarely affect much of the landscape, the absolute change in area burned is small.

Carbon dynamics through an entire disturbance cycle

Although there is a body of literature that separately quantifies the decomposition of standing dead trees, dead tree fall rate, and the decomposition of downed woody debris, there are surprisingly few empirical studies that integrate these processes to estimate the overall longevity of fire-killed trees. Combining disparate estimates of standing and downed wood decay with tree-fall rates sug-

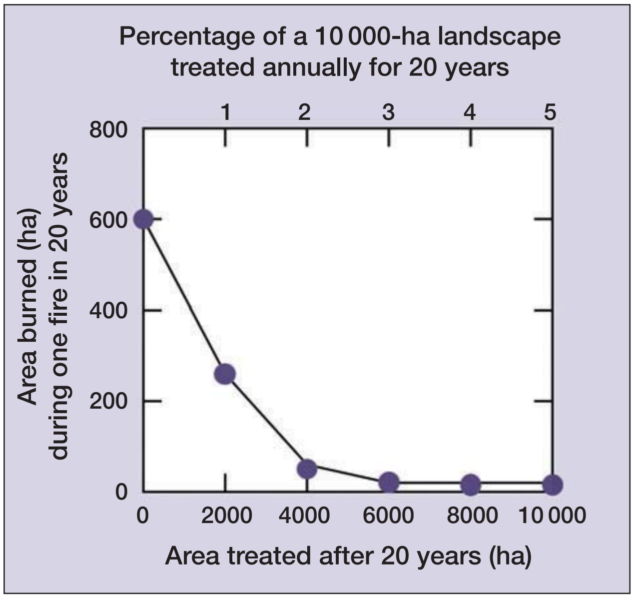

Figure 3. *Simulated effects of strategically placed fuel treatments on wildfire spread across a fire-prone ponderosa and lodgepole pine (*Pinus contorta*) landscape in western Montana. Treating only 1% of the forest annually for 20 years reduced the area impacted by a single large wildfire (assumed to occur about once in 20 years) by more than half. However, across this entire treatment response, the protection of one hectare of forest from fire required the treatment of about 10 hectares. Adapted from Finney* et al. *(2007).*

gests that the overall rate at which fire-killed trees decompose in a semiarid conifer forest likely ranges between 1–9% annually (ie a half-life of 8–70 years). These values are consistent with the observations of Donato (unpublished data), who found that 52% of the biomass killed in a forest-replacing wildfire in southwestern Oregon was still present after 18 years.

It is reasonable to expect that in the first decade or two after a forest-replacing fire, the decomposition of fire-killed trees may exceed the net primary production (NPP) of re-establishing vegetation, thus driving net ecosystem production (NEP) below zero. This expectation is supported by eddy covariance flux measurements (Dore *et al.* 2008) and other empirical studies of post-fire vegetation (Irvine *et al.* 2007; Meigs *et al.* 2009). However, despite a protracted period of negative NEP following a fire event, total C stocks integrated over the entire disturbance cycle may be similar for a forest subject to a fuel-reduction treatment and one subject to a stand-replacing fire. This can easily be shown with a simple C model that simulates growth, mortality, decomposition, and combustion for ponderosa pine forests (Figure 4). How can this be? Simply put, biomass recovery may be slower in the wildfire scenario than in the fuel-reduction scenario, but initial biomass losses may be greater in the fuel-reduction scenario than in the wildfire scenario. Although the parameters used to generate Figure 4 (ie 30% live basal-area removal in the treatment scenario, 100% tree mortality in the wildfire scenario, and rapid post-fire regeneration) are reasonable, real-world responses may not exhibit such parity in integrated C stocks between disturbance types. The point of this simulation is to demonstrate how marked differences in post-disturbance NEP do not necessarily translate into differences in C stocks integrated over time. The quantification of NEP over short intervals is extremely valuable in teasing apart ecosystem C dynamics; however,

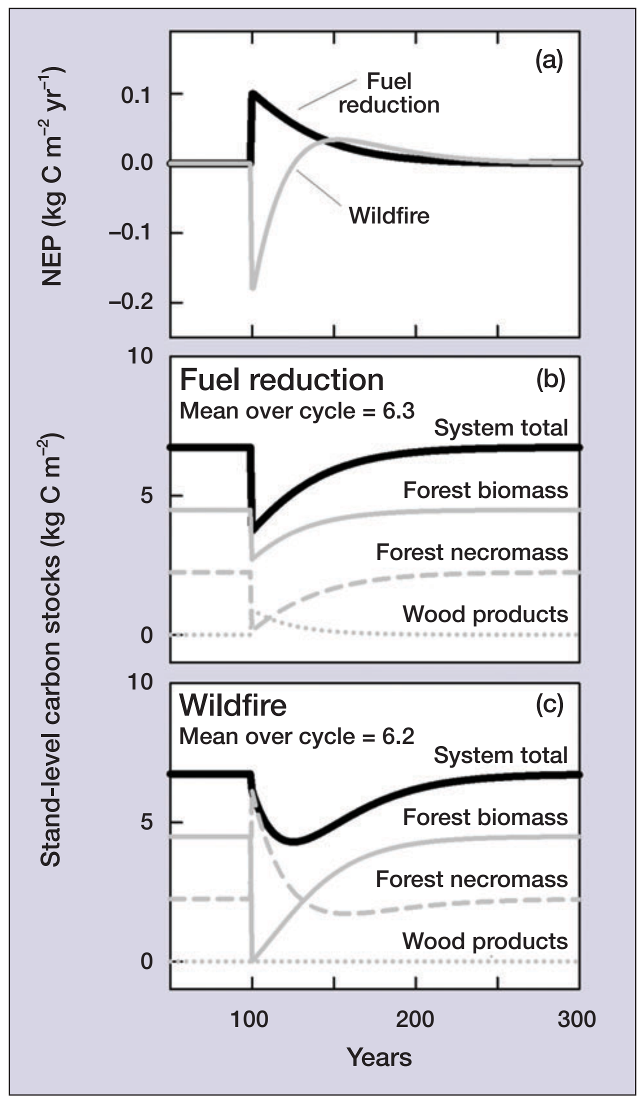

Figure 4. *(a) Simulated net ecosystem production and (b–c) C stocks throughout an entire disturbance interval, initiated by either wildfire or fuel-reduction treatment. Unlike the stand subject to fuel reduction via thinning, the combination of low biomass and high necromass after wildfire functions to drive NEP below zero. Nevertheless, although initial losses associated with wildfire were much lower than those in the fuel-reduction treatment, the two scenarios achieved parity in C stocks over the entire disturbance interval. The model used to generate these simulations was parameterized for a ponderosa pine forest representative of the eastern Cascades and is fully described in WebFigure 1.*

simply comparing C flux rates immediately following different disturbances can give a misleading picture of how disturbances dictate long-term C balance.

Fire frequency and C stocks over multiple disturbance cycles

The C stocks of an ecosystem in a steady state are inversely proportional to the rate constants related to losses, such as those that occur through respiration or combustion (Olson 1963). Whereas Olson (1963) considered ecosystems in steady state, the same phenomenon occurs for the average ecosystem stocks over time or over broad areas (Smithwick *et al*. 2007). As fire frequency increases, the absolute and relative amount of C combusted per individual fire decreases, suggesting that as fire frequency increases, so too will average C stocks. However, using a model that simulates forest growth, mortality, decomposition, and fuel-dependent combustion, researchers can show that a low-frequency, high-severity fire regime stores substantially more C over time than a high-frequency, low-severity fire regime (mean C stocks increased by 40% as the mean fire-return interval was increased from 10 to 250 years; Figure 5). The reason for this is explained by the first principles outlined by Olson (1963). Fractional combustion is, by nature, more constrained than fire frequency. In our example, although fire interval increased from 10 years to 250 years, fractional combustion of ecosystem C for a semiarid ponderosa pine forest only increased from 9% to 18% (Figure 5). To have parity in C stocks across these different fire intervals, fractional combustion per event would, at times, have to exceed 250% – clearly violating the conservation of mass. As long as wildfire does not cause lasting changes in site productivity or non-fire mortality, no forest system is exempt from this negative relationship between fire frequency and average landscape C storage. Although we chose to illustrate the response for a semiarid ponderosa pine forest typical of those considered for fuel reduction, the same relative response was observed when the simulations were run for mesic Douglas fir (*Pseudotsuga menziesii*) forests parameterized for higher production and decomposition rates.

Although stability of C stocks is desirable, stability is a function of spatial extent. In the case of a single forest stand, C stocks under the frequent, low-severity fire regime are more stable than those under an infrequent, high-severity fire regime. However, the fluctuations in C stocks exhibited by a single stand become less relevant as one scales over time or over populations of stands experiencing asynchronous fire events (Smithwick *et al*. 2007). In other words, forests experiencing frequent fires lose less C per fire event than forests experiencing infrequent fires, but the former do not store more C over time or across landscapes.

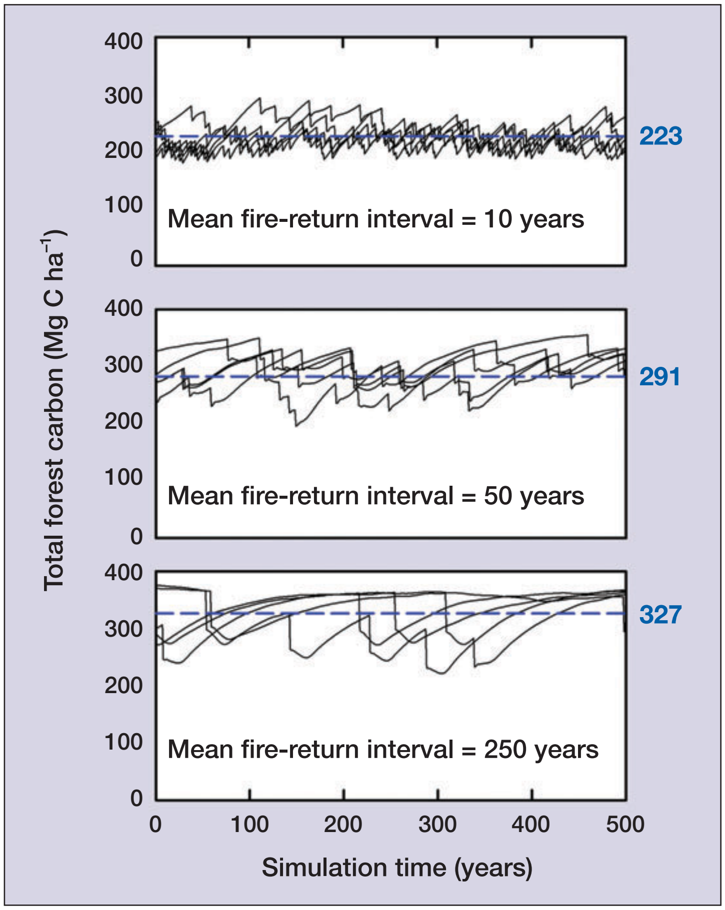

Figure 5. *Total forest C stores simulated for a ponderosa pine forest in the eastern Cascades of Oregon experiencing three different hypothetical fire regimes. Black lines depict the C stores of five individual stands subject to random fire events. Blue lines mark the 500-year average of all five stands. As mean fire-return interval increases, the variation of C stores over time (or space by extension) increases, but so does the long-term average. For simplicity, we show the results of only five stands per fire regime; however, the mean trends do not change with additional simulations. Nearly identical patterns result when alternate forest types are used. We performed simulations using STANDCARB, as described in WebFigure 2 and in Harmon* et al. *(2009).*

The capacity of fire and fuel-reduction treatments to alter equilibrium states

In the sections above, we have assumed that forests eventually succeed toward a site-specific dynamic equilibrium of growth and mortality. Although the concept of a site-specific carrying capacity usefully underlies many of the models of forest development, it is worth considering situations where disturbances might initiate alternate steady states by effecting changes in growth, mortality, or combustibility that persist through to the next disturbance.

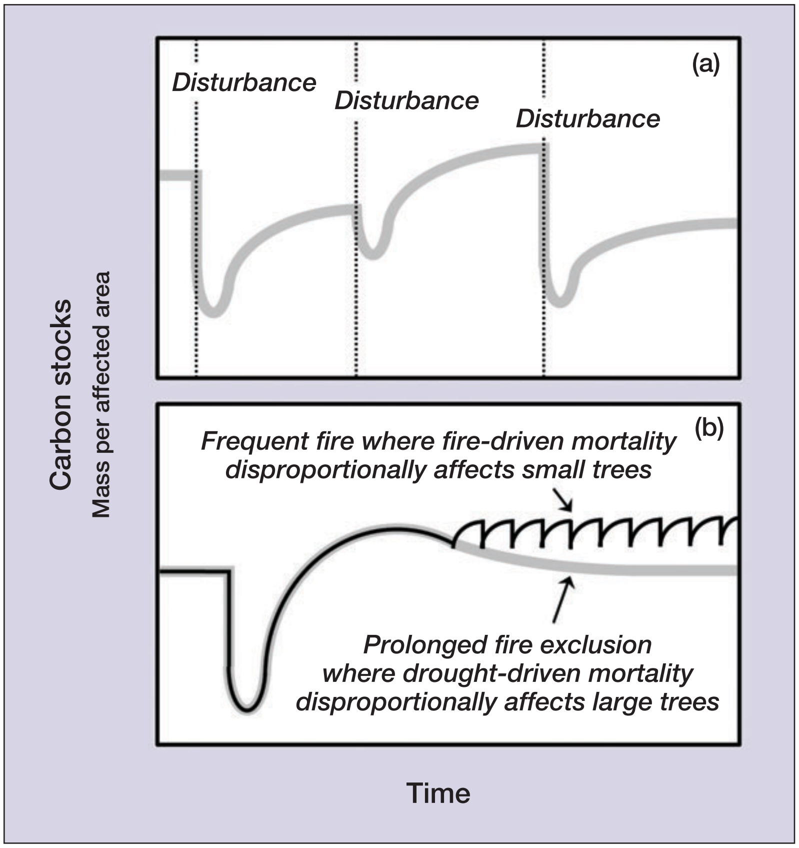

***Figure 6.** Hypothetical examples of how disturbances could initiate alternate steady-state C stocks. (a) Illustration of what C stocks might look like if long-term successional trajectories were contingent more on seed availability at the time of fire than they were on fixed site conditions, as suggested by Kashian* et al. *(2006). (b) Illustration of how frequent fires could shift mortality away from larger trees and toward smaller trees, thus increasing steady-state C stocks, as suggested by North* et al. *(2009).*

A simple example of disturbance-altering, long-term forest growth involves the loss in soil fertility that can accompany certain high-severity fires (Johnson and Curtis 2001; Bormann *et al.* 2008). Another mechanism by which disturbance can initiate changes in steady-state C stocks involves the persistent changes in tree density that may follow some disturbance events. For instance, Kashian *et al.* (2006) determined that forest biomass in the lodgepole pine (*Pinus contorta*) forests of Yellowstone National Park was relatively insensitive to changes in fire frequency but very dependent on the density to which forests grew after fire. In a system where long-term successional trajectories are contingent more on forest condition at the time of disturbance (eg serotinous seed availability) than on permanent site conditions, C stocks could well stabilize at different levels after different disturbances, as illustrated in Figure 6a.

A final example of how changes in disturbance regime could persistently alter equilibrium between growth and mortality involves size-dependent mortality in the semiarid conifer forests of the Sierra Nevada (Smith *et al.* 2005). Both Fellows and Goulden (2008) and North *et al.* (2009) found fewer large trees and lower overall biomass in current fire-excluded forests than were believed to exist at these locations before fire exclusion. These authors suggest that small trees are disproportionally vulnerable to fire mortality, and large trees are disproportionally vulnerable to pathogen- and insect-based mortality; therefore, as biological agents replace fire as the primary cause of mortality, the number of large trees decreases accordingly. Under such scenarios, the thinning of small trees combined with frequent burning could, over time, increase biomass by maintaining a greater number of larger trees (see Figure 6b). However, not all studies support the notion that fire exclusion reduces stand-level biomass (Bouldin 2008; Hurteau *et al.* 2010). Specifically, another study conducted by North *et al.* (2007) in the Sierra Nevada found that net losses in large-diameter trees between 1865 and 2007 were more than compensated for by the infilling of small-diameter trees, such that total live-wood volume remained unchanged over this period of fire suppression. Furthermore, Hurteau and North (2010) reported that fire-suppressed control plots aggraded as much C over 7 years as did comparable thinned plots.

Presuming that maximum steady-state C stocks are not dictated entirely by permanent site qualities and depend, at least in some part, on the nature and timing of disturbance, it is conceivable that prescriptions such as fuel reduction and prescribed fire could eventually elevate (or reduce) C stocks at a single location slightly beyond what they would be under a different disturbance regime (Hurteau *et al.* 2010). However, exactly how stable or self-reinforcing this alternate state is remains unknown.

■ Additional considerations

The purpose of this paper is to illustrate the basic biophysical relationships that exist between fuel-reduction treatments, wildfire, and forest C stocks over time. Understanding these dynamics is necessary for crafting meaningful forest C policy; however, it is not by itself sufficient. A full accounting of C would also include the fossil-fuel costs of conducting fuel treatments, the longevity of forest products removed in fuel treatments, and the ability of fuel treatments to produce renewable "bioenergy", potentially offsetting combustion of fossil fuels. A detailed consideration of these factors is beyond the scope of this paper, but it is worth pointing out some limits of their contribution. First, the fossil-fuel costs of conducting fuel treatments are relatively small, ranging from 1–3% of the aboveground C stock (Finkral and Evans 2008; North *et al.* 2009; Stephens *et al.* 2009b). Second, only a small fraction of forest products ever enters "permanent" product stocks; this is especially true for the smaller-diameter trees typically removed during fuel treatments. Primarily, half-lives of forest products (7–70 years) are not significantly different than the half-life of the same biomass left in forests (Krankina and Harmon 2006). Third, the capacity of forest biofuels to offset C emissions from fossil-fuel consumption is greatly constrained by both transportation logistics and the lower energy output per unit C emitted as compared with fossil fuel (Marland and Schlamadinger 1997; Law and Harmon 2011).

Conclusions

The empirical data used in this paper derive from semiarid, fire-prone conifer forests of the western US, which are largely composed of pine, true fir (*Abies* spp), and Douglas fir. These are the forests where management agencies are weighing the costs and benefits of up-scaling fuel-reduction treatments. Although it would be imprudent to insist that the quantitative responses reported in this paper necessarily apply to every manageable unit of fire-prone forest in the western US, our conclusions depend not so much on site-specific parameters but rather on the basic relationships – between growth, decomposition, harvest, and combustion – to which no forest is exempt. To simply acknowledge the following – that (1) forest wildfires primarily consume leaves and small branches, (2) even strategic fuels management often involves treating more area than wildfire would otherwise affect, and (3) the intrinsic trade-off between fire frequency and the amount of biomass available for combustion functions largely as a zero-sum game – leaves little room for any fuel-reduction treatment to result in greater sustained biomass regardless of system parameterization. Only when treatment, wildfire, or their interaction leads to changes in maximum biomass potential (ie system state change) can fuel treatment profoundly influence C storage.

In evaluating the effects of wildfire and fuel-reduction treatments on forest C stocks across various spatial and temporal scales, we conclude that:

(1) Empirical evidence shows that most pyrogenic C emissions arise from the combustion of surface fuels, and because surface fuel is combusted in almost all fire types, high-severity wildfires burn only 10% more of the standing biomass than do the low-severity fires that fuel treatment is intended to promote (Figure 1).
(2) Model simulations support the notion that forests subjected to fuel-reduction treatments experience less pyrogenic emissions when subsequently exposed to wildfires. However, across a range of treatment intensities, the amount of C removed in treatment was typically three times that saved by altering fire behavior (Figure 2).
(3) Fire-spread simulations suggest that strategic application of fuel-reduction treatments on as little as 1% of a landscape annually can reduce the area subject to severe wildfire by 50% over a 20-year period. Even so, the protection of one hectare of forest from wildfire required the treatment of 10 hectares, owing not to the low efficacy of treatment but rather to the rarity of severe wildfire events (Figure 3).
(4) It is reasonable to expect that after a forest-replacing fire, the decomposition of fire-killed trees exceeds NPP, driving NEP below zero. By contrast, the deliberate removal of necromass in fuel-reduction treatments could result in a period of elevated NEP. However, despite marked differences in post-disturbance NEP, it is possible for average C stocks to be identical for these two disturbance types (Figure 4).
(5) Long-term simulations of forest growth, decomposition, and combustion illustrate how, despite a negative feedback between fire frequency and fuel-driven severity, a regime of low-frequency, high-severity fire stores more C over time than a regime of high-frequency, low-severity fire (Figure 5).
(6) The degree to which fuel management could possibly lead to increased C storage over space and time is contingent on the capacity of such treatments to increase maximum achievable biomass through mechanisms such as decreased non-fire mortality or the protection from losses in soil fertility that are sometimes associated with the highest-severity fires (Figure 6).

There is a strong consensus that large portions of forests in the western US have suffered both structurally and compositionally from a century of fire exclusion and that certain fuel-reduction treatments, including the thinning of live trees and prescribed burning, can be effective tools for restoring historical functionality and fire resilience to these ecosystems (Hurteau *et al.* 2010; Meigs and Campbell 2010). Furthermore, by reducing the likelihood of high-severity wildfire, fuel-reduction treatments can improve public safety and reduce threats to the resources provided by mature forests.

On the basis of material reviewed in this paper, it appears unlikely that forest fuel-reduction treatments have the additional benefit of increasing terrestrial C storage simply by reducing future combustive losses and that, more often, treatment would result in a reduction in C stocks over space and time. Claims that fuel-reduction treatments reduce overall forest C emissions are generally not supported by first principles, modeling simulations, or empirical observations. The C gains that could be achieved by increasing the proportion of large to small trees in some forests are limited to the marginal and variable differences in biomass observed between fire-suppressed forests and those experiencing frequent burning of understory vegetation.

Emerging policies aimed at reducing atmospheric CO_2 emissions may well threaten land managers' ability to apply restoration prescriptions at the scale necessary to achieve and sustain desired forest conditions. For this reason, it is imperative that scientists continue research into the processes by which fire can mediate long-term C storage (eg charcoal formation, decomposition, and community state change) and more accurately quantify the unintended consequences of fuel-reduction treatments on global C cycling.

Acknowledgements

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WebTable 1. Biomass reductions associated with various fuel reduction treatments as prescribed at various fire-prone forests of western North America

Treatment type, forest type, and location	*Fraction of live basal area cut or killed in prescribed burn*	*Fate of logging slash*	*∆ surface fuels (estimated fraction)*	*∆ total aboveground biomass through both combustion and removal (estimated fraction)*
Prescribed fire only				
Sierran mixed conifer (Central Sierras)[a]	0.00	None	–0.70	–0.11
Sierran mixed conifer (Central Sierras)[b]	0.15	None	–0.02	–0.13
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.11	None	–0.19	–0.12
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.08	None	–0.32	–0.12
Ponderosa pine (Southwestern Plateau)[b]	0.04	None	–0.50	–0.11
Ponderosa pine/true fir (Southern Cascades)[b]	0.30	None	0.67	–0.16
Thinning only				
Sierran mixed conifer (Central Sierras)[a]	0.36	Left on site	0.96	–0.16
Sierran mixed conifer (Central Sierras)[a]	0.60	Left on site	1.60	–0.27
Ponderosa pine (Southern Rockies)[c]	0.36	Pile burned	0.01	–0.30
Ponderosa pine (Central Sierras)[d]	0.50	Pile burned	0.01	–0.42
Sierran mixed conifer (Central Sierras)[b]	0.34	Left on site	0.92	–0.15
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.54	Left on site	1.43	–0.24
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.24	Left on site	0.64	–0.11
Ponderosa pine (Southwestern Plateau)[b]	0.53	Pile burned	0.01	–0.45
Ponderosa pine/true fir (Southern Cascades)[b]	0.58	Removed	0.00	–0.49
Thinning and prescribed fire				
Sierran mixed conifer (Central Sierras)[a]	0.37	Left on site	–0.40	–0.38
Sierran mixed conifer (Central Sierras)[a]	0.66	Left on site	–0.17	–0.59
Ponderosa pine (Central Sierras)[d]	0.50	Pile burned	–0.69	–0.53
Sierran mixed conifer (Central Sierras)[b]	0.42	Left on site	–0.37	–0.41
Ponderosa pine/Douglas fir (Northern Rockies)[b]	0.78	Left on site	–0.08	–0.67
Ponderosa pine/Douglas fir (Blue Mountains)[b]	0.46	Left on site	–0.33	–0.44
Ponderosa pine (Southwestern Plateau)[b]	0.59	Pile burned	–0.68	–0.61
Ponderosa pine/true fir (Southern Cascades)[b]	0.73	Removed	–0.70	–0.72

Notes: Total biomass losses were approximated solely from basal reported area reduction according to the following assumptions: total aboveground biomass was assumed to be composed of 45% live merchantable boles (subject to removal proportional to basal area reduction), 40% live tree branch and foliage (converted to slash proportional to basal area reduction), and 15% surface fuels (both live and dead biomass and subject to combustion in prescribed fire). Prescribed fire was assumed to combust 70% of surface fuels and logging slash; pile burning was assumed to combust 99% of logging slash. [a]North *et al.* (2007); [b]Stephens *et al.* (2009); [c]Finkal and Evans (2008); [d]Campbell *et al.* (2008).

WebTable 2. Burn probability for forests of Oregon, Washington, and California from 1985 to 2005

Forest type (ecoregion)	*Fraction of forest area burned annually*		*Fuel-treatment life expectancy (range in years)*	*Random probability of a treated stand being exposed to any fire*
	Any severity	*High severity*		
Cool–wet conifer (Coast Range)	0.00018	0.00002	5–15	0.0009–0.00274
Cool–mesic conifer (West Cascades, North Cascades)	0.00177	0.00046	5–15	0.00884–0.02651
Cool–dry conifer (East Cascades, North Rockies, Blue Mts)	0.00411	0.00054	10–25	0.04112–0.10279
Warm–mesic conifer (Klamath Mountains)	0.00622	0.00119	10–25	0.06217–0.15542
Warm–dry conifer (Sierra Nevada, South California Mts)	0.00780	0.00178	10–25	0.07798–0.19495

Notes: This simple prediction of wildfire-treatment occurrence by multiplying regional fire probability by fuel treatment life assumes random interaction of wildfire and treatment and does not account for strategic placement of fuel treatments. Area burned annually based on Monitoring Trends in Burn Severity fire perimeter and severity classification maps from 1985 to 2005 (http://mtbs.gov). Total forested area in each ecoregion based on 2005 National Land Cover Dataset land-cover maps (http://landcover.usgs.gov). Ecoregions correspond to Omernik Level 3 classification (Omernik 1978). Treatment life expectancies are crude estimates based on Rhodes and Baker (2008) and Agee and Skinner (2005). Being that these numbers were derived from actual region-wide land-surface-change detection, they include regional fire suppression activities. Natural burn probabilities, as well as those that may result from future management decisions or climate change, are likely to be higher.

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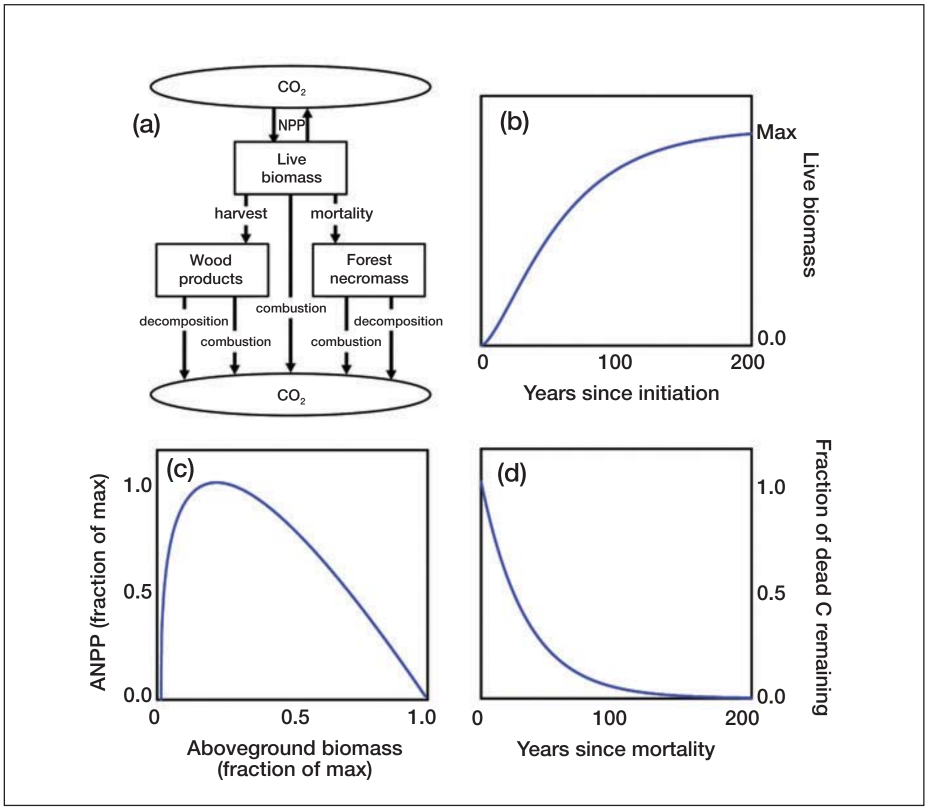
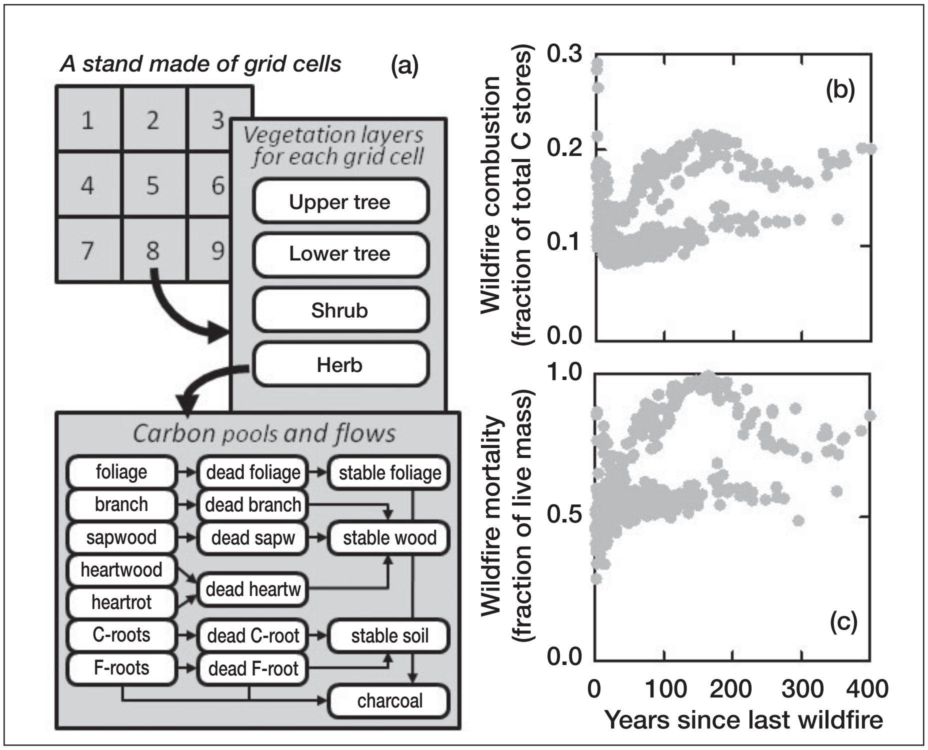


WebFigure 1. *(a) Structure and (c and d) dynamic functions behind the forest carbon model used to produce Figure 4. (b) Live biomass is assumed to aggrade over time according to a Chapman-Richards function* $y_1 = a^*(1 - \exp[-b_1x_1])^c$, *the derivative of which,* $y_2 = c^*b_1{}^*y_1{}^*(1 - \exp[\ln\{y_1/a\}/c])/\exp(\ln[y_1/a]/c)$, *allows (c) ANPP (aboveground net primary production) to be calculated annually according to current biomass;* y_1 *is aboveground live biomass in kg C m*$^{-2}$, y_2 *is ANPP in kg C m*$^{-2}$ *yr*$^{-1}$, a *is the maximum aboveground live biomass that the site can sustain,* x_1 *is the time in years since initiation (which can be back-calculated from any assigned biomass),* b_1 *is a constant proportional to the time required to achieve maximum biomass, and* c *is a constant proportional to the initial growth lag. (d) Decomposition, the heterotrophic mineralization of each necromass pool including wood products, is determined according to an exponential loss function* $y_4 = M^* - k$, *where* y_4 *is loss of necromass in kg C m*$^{-2}$ *yr*$^{-1}$, M *is the current mass of necromass in kg C m*$^{-2}$, *and* k *is a pool-specific decomposition constant. For the simulations shown in Figure 4, we used the following parameters to represent growth, harvest, combustion, and decay in a semiarid, fire-prone pine forest of western North America:* a = *4.8 kg C m*$^{-2}$; b_1 = *0.02;* c = *1.6;* k = *0.005 yr*$^{-1}$ *for both forest necromass and wood products; starting* y_1 = *4.5 kg C m*$^{-2}$*; starting* $M_{necromass}$ = *2.2 kg C m*$^{-2}$*; starting* $M_{products}$ = *0 Mg C ha*$^{-1}$*; treatment removals = 2.9 kg C m*$^{-2}$*; treatment related mortality (uncombusted slash) = 1% of* y_1 *at time of treatment; wildfire mortality = 95% of* y_1 *at time of fire; wildfire combustion = 10% of* y_1 *and 40%* M *of at time of fire.*



WebFigure 2. *(a) Structure and (b and c) disturbance responses behind STANDCARB, the forest carbon model used to produce Figure 5. STANDCARB simulates the accumulation of C over succession in mixed-species and mixed-age forest stands at annual time steps. The growth of vegetation and subsequent transfer of C among the various carbon pools shown in (a) are regulated by user-defined edapho-climatic inputs and species-specific responses. The imposition of wildfire in any given year results in the instantaneous transfer of C from each live pool into its corresponding dead pool (wildfire mortality) and the instantaneous loss of C from each live and dead pool to the atmosphere (wildfire combustion). The exact amount of mortality and combustion incurred in a given wildfire depends on stand-specific species composition, and the amount of biomass in each separate C pool, which at any given time may not be in equilibrium (gray circles in [b] and [c] reflect this variation). For the simulations shown in Figure 5, STANDCARB was parameterized for a semiarid ponderosa pine forest growing in eastern Oregon: max attainable biomass = 210 Mg C ha^{-1}; mean ANPP = 5.1 Mg C ha^{-1} yr^{-1}; non-fire mortality rate constants = 0.37, 0.5, 0.032, 0.017, and 0.013 yr^{-1} for foliage, fine roots, branches, coarse roots, and stems, respectively; decomposition rate constants = 0.21, 0.15, 0.08, 0.11, 0.023, and 0.017 yr^{-1} for foliage, fine roots, branches, coarse roots, stems, and soil C, respectively. It is worth noting that patterns nearly identical to those illustrated in Figure 5 result from STANDCARB parameterized for a mesic Douglas-fir forest having much larger ANPP, potential biomass, and decomposition rates. For a full description of STANDCARB structure and parameterization, see Harmon* et al. *(2009) and http://andrewsforest.oregonstate.edu/lter/pubs/webdocs/models/standcarb2.htm.*

Impacts of Thinning on Carbon Stores in the PNW: A Plot Level Analysis

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Final Report

College of Forestry
Oregon State University
25 May 2011

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Executive Summary

This report provides an analysis of forest carbon stores, fluxes and avoided emissions directly related to fuel reduction thinnings for sample plots in eastern and western Oregon.

Primary Goals

- Determine the level of on-site carbon storage under different thinning prescriptions and in different forest types.
- Analyze plot-level forest carbon pools and carbon fluxes over a 50-year period. Compare alternative thinning treatments with a no thinning scenario.
- Estimate the amount of carbon transferred to harvested wood products, carbon emissions of biomass burning for energy production, and avoided carbon emissions from not burning fossil fuels.
- Determine if revenue from harvested wood products from the thinning treatment could pay for the thinning under specified market and harvest unit assumptions for one thinning scenario (the “breakeven” scenario).

Methods

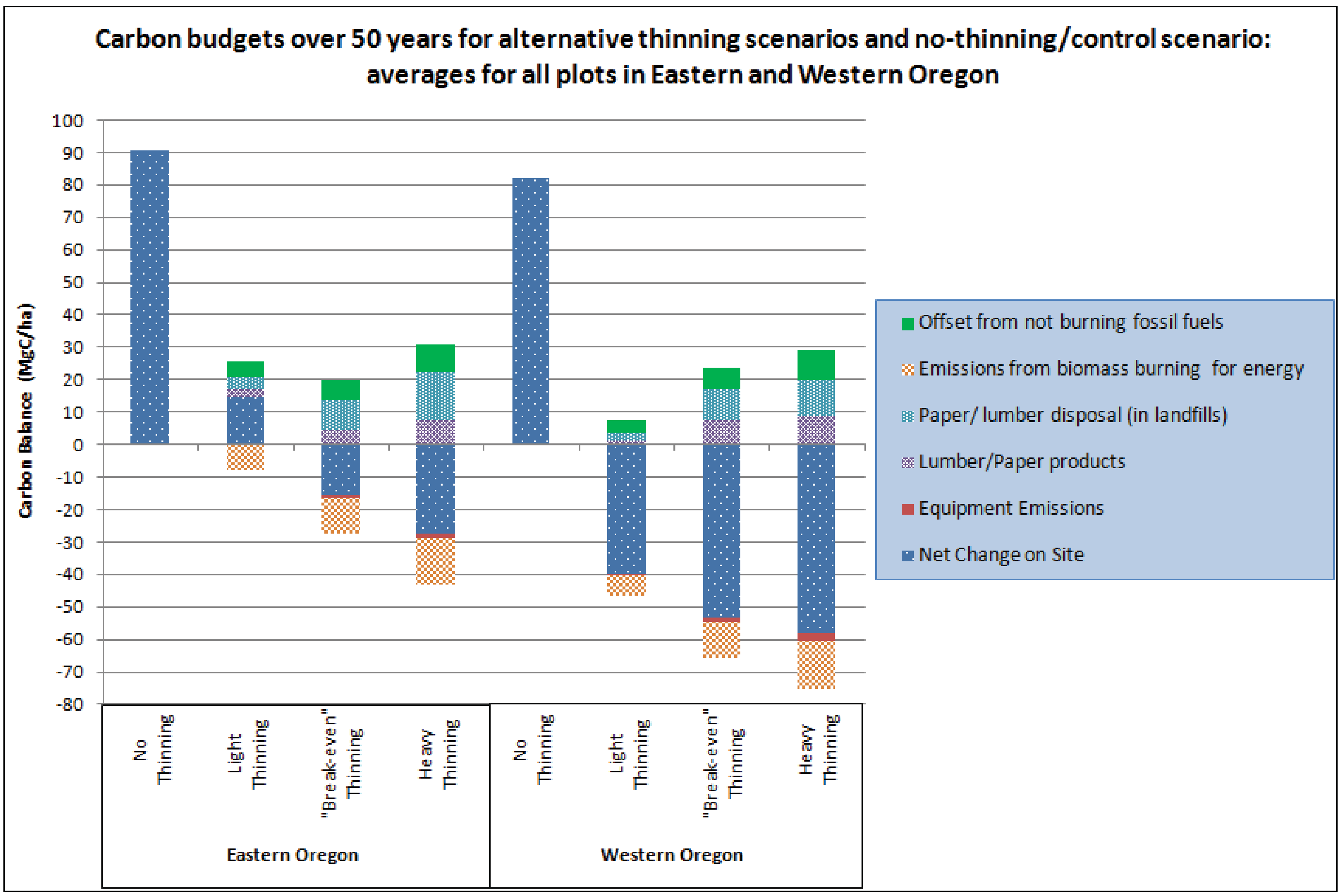
- Plots were chosen from the Forest Inventory and Analysis (FIA) National Program and the Current Vegetation Survey (CVS) to represent a range of common landscape types with stand conditions that show a potential for fuel reduction.
- Plots were all from Oregon, including the Eastern Cascade, Western Cascade, and Blue Mountain regions. A wide range of stand ages was included (21-269 years for Eastern Oregon/Blue Mountains and 10-220 years for Western Oregon).
- Thinning scenarios were developed to meet specified torching and crowning thresholds. All simulated thinnings use a “thin from below” (low thinning) approach. A control (no harvest scenario) is compared to different treatments.
- Carbon pools were estimated using the Fire and Fuels Extension (FFE) of the Forest Vegetation Simulator (FVS) with manual adjustments and additions to address known model limitations.
- Estimated harvest costs were based on the Fuel Reduction Cost Simulator (FRCS-West). Estimated timber revenues were based on ODF data.

Findings

- Forest carbon pools always immediately decreased as a result of a fuel reduction thinning, with larger differences in total carbon pools resulting from heavier thinning treatments.
- After thinning, forest carbon pools (both total and standing live aboveground) remain lower throughout a 50-year period for all simulated plots in eastern and western Oregon. The difference in total carbon pools between a thinned and unthinned plot is dependent on the level of live standing tree inventory reduction. A heavier thin tends to reduce carbon pools more than lighter thins throughout a 50-year simulated period.
- Carbon pool estimates for thinned stands were still lower than unthinned stands even after accounting for carbon transfer to wood products and avoided emissions from fossil fuels for energy production. After simulating growth

in the stands for 50 years the average difference in net carbon balance between unthinned and thinned plots for the three age groups ranged between 73.5– 103.4 MgC/ha in Eastern Oregon to 121.8 – 128.6 MgC/ha in Western Oregon. Carbon losses on site account for the bulk of the effect of thinning on carbon. Carbon retention in wood products and avoided emissions from fossil fuels tend to offset the equipment emissions and emissions from burning biomass for energy, but not the loss of carbon from forest on site.

- The following figure (adapted from Table 15) shows that, regardless of the single-entry thinning regime used, the "No Thinning" scenario resulted in the most carbon remaining on-site following 50 years. The figure accounts for emissions from equipment and emissions from biomass burning, and also accounts for paper/lumber products sequestered after 50 years, and offsets from burning biomass for energy instead of fossil fuels. The "Net Change" in the graph includes all gains and losses in carbon on-site 50 years after either no thinning, or 50 years following a thinning from a single entry.



- For the plots examined, it is generally possible to reach specific fuel reduction goals with revenues exceeding treatment costs. There are notable exceptions in younger plots, particularly in plots with relatively few larger trees (as measured by DBH). If administrative costs are included, treatment costs may exceed harvest revenues on federal lands. Financial viability is significantly affected by many stand-dependent variables, including current stand structure, average distance of wood from roadside, average distance of stand to mill/plant, and current market prices.

- Burning biomass from forest fuel reduction thinnings results in avoided carbon emissions from fossil fuels. Due to relatively low energy density, biomass has greater carbon emissions from the boiler per energy unit produced (CO_2 emissions per kWh or BTU produced) when compared to carbon emissions from fossil fuels (coal, natural gas) per energy unit produced.
- All thinning scenarios on all plots without exception resulted in a significant loss of carbon relative to a no-thinning scenario. This suggests that the findings may be applicable to other forest types and thinning prescriptions.

Key Assumptions and Limitations

Our key assumption is that the life cycle analysis of carbon stores and fluxes begins with initial carbon stores in the stand prior to thinning as described by Maness 2009. In other words, our analysis starts with existing forest condition and measures the net change in carbon stores due to the thinning treatments. This assumption contrasts with other studies (e.g., Lippke et al. 2004) that start with bare ground as a system boundary. The results (and potentially the conclusions) can be dramatically affected by the choice of system boundary.

- Not considered in this analysis:
    - Effects of fire on carbon pools and flux. This includes any potential post-thin treatments. In this study, we do not estimate whether carbon emissions from prescribed fire and/or wildfire would (over repeated cycles) be higher or lower after thinning.
    - Soil carbon and fine roots (roots less than 2 mm in diameter).
    - Emissions due to consumption of electric power in lumber and paper production. Including these emissions would increase the greenhouse gas emissions for each of the thinning scenarios.
    - Disposal methods for wood products (e.g., recycling and use as biofuel). In this analysis, wood products are assumed either taken to a landfill or burned as an energy source.
    - Effects of climate change (e.g., temperature, precipitation).
    - Vegetation in-growth. This report assumes that in-growth is managed with regular treatment (e.g., with herbicides) that limits in-growth. If in-growth is allowed and fire is suppressed, estimates of carbon pools on-site may significantly increase, especially for longer time periods.
    - Emission reductions from substitution effects of wood products for more energy intensive alternative building materials (such as concrete, brick, or steel). Inclusion of substitution effects would decrease carbon emissions for thinning scenarios.

Because this is a plot-level study, where plots were chosen based on specific criteria (stand age, specific stand structures, specific dominant species), study results cannot be extrapolated directly to a regional analysis. The analysis assumes that there is no re-entry onto the site in the next 50 years. The stand projection is shown for illustrative purposes only; it is not intended to be a management prescription.

Future Work

There are several potential areas of study that can support and enhance work begun in this report. This would close the gap on some of the limitations presented within this report.

An expanded analysis would improve regional understanding of forest carbon stores in varying conditions. Inclusion of one or more of the following variables would not only expand the scope of this report but also enhance the results presented from the study.

- Effects of prescribed fire and wildfire intensity and frequency on carbon stores.
- Effects of strategic placement of thinning on carbon stores for larger areas.
  - Effects of thinning in easily accessible areas (e.g., near roads) vs. thinning over larger areas.
  - Urban thinning.
- Effects of varying the price for biomass.
  - Sensitivity analysis of biomass price (and potential impact of financial subsidies on thinning regime).
- Inclusion of thinning regimes as part of a broader strategy to improve forest health or in response to insects/disease (e.g., beetle kill).
- Establish a more detailed time profile of carbon. This would include an annual carbon budget over a given time frame instead of a carbon budget at less frequent intervals.
- Since all thinning treatments reduced carbon storage over a 50-year period, it is possible that additional entries would further reduce carbon stores. In order to more fully understand the effects, a more complete forest management should be included in future work, instead of a single management action (thinning).

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Introduction

There is growing interest in improving the resilience of forests to fire, insects, and disease in the Pacific Northwest and in biomass recovery for energy production (Graham et al. 2004; Lord et al. 2006). There has also been extensive analysis and discussion on the impact of forest management (and other disturbances) on forest carbon stores and fluxes (Krankina and Harmon 2006). Other studies have developed regional estimates of forest carbon stores (Dushku et al. 2007).

The purpose of this study is to determine the level of on-site carbon stores at a plot level under different fuel reduction thinning operations in different forest types in Oregon. Some off-site carbon estimates are made as well. A collection of relatively densely stocked plots was chosen from five Oregon counties in the Western and Eastern Cascades, southwest Oregon, and the Blue Mountains region.

The carbon pools of each plot for thinned and unthinned scenarios are projected and compared. The resulting simulated carbon stores and carbon fluxes from this model are not intended to be extrapolated to regional or landscape levels, and are restricted to a plot-level analysis. To simplify the analysis, we limit our examination to a subset of possible product end uses. Therefore, the model does not comprehensively describe all potential carbon fluxes. A life cycle analysis would more fully define carbon transfers for alternative product uses.

The report is organized as follows:

- Plot-level model approach and design
- Choice of plot-level simulator for tree growth
- Carbon fluxes
- Scope of this study
- Plot selection
- Detailed example plot to show methodology
- Broader analysis of plots, fewer details shown
- Overall results from analysis
- Discussion
- Suggestions for future analysis
- References
- Appendices (primarily detailed results)

Suggestions for further research are included. The reader is encouraged to use the reference section to access more detailed information. Some of the topics discussed in this report (such as fuel reduction for wildfire mitigation) currently either have mixed results or may lack scientific consensus, and we identify these areas when appropriate.

Model Overview

This section describes a model that simultaneously analyzes the economic feasibility of a fuel treatment (thinning) and the impact of the forest treatment on forest carbon pools and fuel loading at a plot level. For each plot, a customized treatment is implemented following an analysis of the current situation using several criteria. The procedure and results for an example plot are described in detail and the procedure is then applied to all plots. The analysis groups plots into age groups and regions, then notes differences between groups and possible causes for these differences.

The objectives for this study integrate both carbon accounting and economic considerations.

Model objectives include (not necessarily in order of importance):

- Implement thinning regimes for each plot that reduce modeled fuel loading.
- Identify and quantify carbon losses in the carbon pools that occur for each plot after thinning.
- Estimate carbon fluxes for removed trees and any potential carbon displacement by replacing fossil fuels with biomass for energy usage.
- For each plot, include one breakeven forest treatment with a forest harvest system (including transportation, processing, move-in, and setup costs) that, when implemented, does not result in a net financial loss for the landowner. To facilitate harvesting cost accounting, harvesting system choice was limited to a whole tree harvesting system. The harvesting system choice may affect the breakeven thinning scenario, but does not significantly affect the relative carbon budget for the light and heavy thinning scenarios.

The parameters for the model are customized for each plot. The general construction of the model and the interaction between objectives is shown (Figure 1).

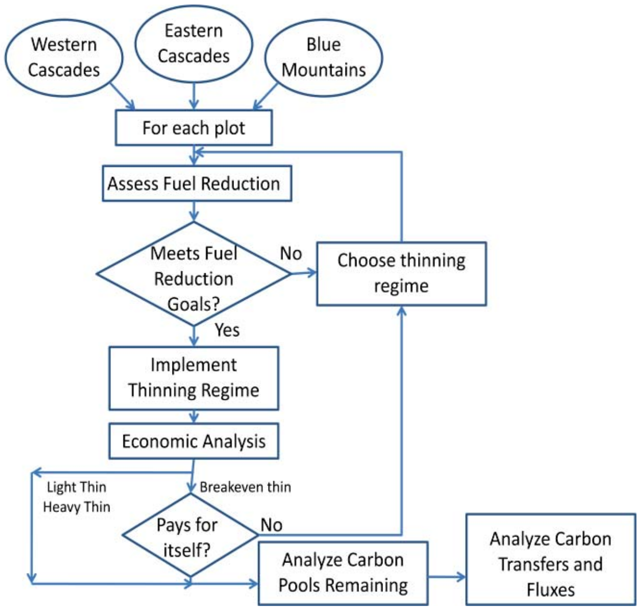

Figure 1. Model flowchart with objective interaction. Light Thin and Heavy Thin scenarios are not expected to pay for themselves, but the Breakeven Thin is expected to pay for itself.

Thinning Prescriptions

There are many potential thinning prescriptions that can vary due to landowner objectives and constraints. Objectives may include (1) increased wood production, (2) increased resistance to fire, insects and disease, and (3) enhancement or control of plant and animal habitats (Nyland 2002; Graham et al. 2004). The purpose of this report is not to advocate one thinning prescription over another, but to show carbon stores and fluxes given one set of objectives. A regional plan would likely integrate multiple spatially-dependent objectives into a larger scope. Several thinning intensities are simulated, ranging from a light thin to heavier thinnings.

To maintain consistency between plots in this analysis, the general criteria for thinning each plot includes:

- Stands are to be thinned from below (low thinning), where smaller diameter trees are removed from dense stands. Pollet and Omi (2002) have shown this thinning regime to be effective in reducing crown fire severity in ponderosa pine.
- Since the smallest trees removed often do not "pay for themselves" in a thinning (USFS 2005), a proportion of larger diameter trees (up to 20" DBH) may also be removed in the breakeven scenarios or to achieve low stocking levels, but the largest trees within a plot are left if possible. Largest trees are determined by diameter at breast height (DBH), which is a diameter estimate 4.5 ft (1.37 m) from the ground.
- Brush and smaller trees in the understory are identified as a potential fuel ladder, and smaller vegetation not removed from the stand is trampled or crushed in the simulation (this includes all trees <3" DBH).
- Treated plots should meet both fuel hazard measurement goals and, for the breakeven scenario, economic requirements immediately following the thinning, if possible.

It is not implied that this thinning prescription should be applied across a more complex landscape level. This prescription strategy is simulated only for these isolated plots. A thinning prescription at a regional scale (e.g., Finney et al. 2006) could consider many factors, including

- Long-term prescription alternatives for the stand.
- Prescriptions/species/ fuel loadings for surrounding stands
- Fire hazards that are not necessarily measured by fuel loading (e.g., topography)
- Desired combination of tree species and stand structures (e.g., Fiedler et al. 1998)
- Wildlife considerations (e.g., endangered species, fish/bird/animal habitat requirements) (Hayes et al. 1997)
- Susceptibility to insects and/or disease (Hessburg et al. 1993)
- Watersheds and proximity to riparian areas
- Aesthetics and recreational potential (Scott 1996)
- Accessibility to harvesting equipment

Thinning and fuels treatment only temporarily reduces fuel loading within a stand. In order to be more effective over the long term, it is necessary to implement a strategy (such as prescribed burning) that would periodically reduce surface fuels (Weatherspoon 1996) and possibly to re-enter the stand for periodic thinnings (Keyes and O'Hara 2002). The carbon fluxes associated with a prescribed burn or re-entries is not included in this model. Even though fire behavior may be more influenced by weather conditions and topography (Bessie and Johnson 1995), fuel loading is still an important variable affecting stand mortality in a wildfire. From a strict carbon savings perspective, there are currently two views concerning the effects of wildfire following a fuel reduction treatment (Ryan et al. 2010):

- Some studies and models show less carbon loss from thinned stands (compared to unthinned stands) following a crown fire.
- Some studies and models show that in most forest types, thinned stands have less carbon than unthinned stands at a landscape level following a crown fire.

Regional research comparing Eastern and Western Cascades suggests that if thinning ever reduces total net carbon loss from thinning combined with subsequent wildfire, it would likely only be in Eastern Cascade ponderosa pine stands with dense understory (Mitchell et al. 2009).

Choice of Model to Project Forest Carbon

There are several models developed to simulate forest carbon – for example, Harmon and Marks (2002) simulate forest carbon on a landscape level. This analysis is conducted using a growth and yield model. There are several forest growth and yield models available for the Pacific Northwest region (Marshall 2005). The Forest Vegetation Simulator (FVS) was chosen as the growth and yield model for this study – it is commonly used for both national and regional stand projections, has an integrated graphical user interface (SUPPOSE – Crookston 1997), and also has a built-in Fire and Fuels Extension (FFE - Reinhardt and Crookston 2003) that has been used to estimate forest carbon pools over time (e.g. Manomet 2010).

Carbon Fluxes

Figure 2 shows an example of carbon stores and associated carbon fluxes used in calculations for this report.

The stores are calculated as follows:

- Total Carbon on Site – Carbon on site in any given year.
- Biomass for Energy – Carbon processed (burned) for biomass energy in the year of harvest. Combination of slash/small trees (primary source) and residues from the lumber/paper manufacturing process (secondary source).
- Lumber Products - Carbon store transferred to lumber products from harvest and manufacturing process.
- Paper Products - Carbon store allocated to paper products from harvest and manufacturing process.
- Paper/Lumber Residue – Carbon store transferred to paper/lumber process, but not converted to paper or lumber products. Some of this store is allocated to biomass for energy, and the remaining portion is assumed disposed in a landfill (1% decay rate assumed – decay rate used in other models: e.g., Hennigar et al. 2008).
- Landfill – Carbon store to where paper and lumber products are assumed transferred following use. The landfill decay rate is assumed to be 1%.

Some other carbon fluxes are not specifically quantified in this report (e.g., impact of thinning on soil carbon, fossil fuel emissions associated with energy needs of product manufacturing, effects of substitution of wood products for more energy-intensive materials). Accounting for these additional C fluxes is a complicated process and is beyond the scope of this report. However, these factors collectively would not be expected to change the overall conclusions of the study.

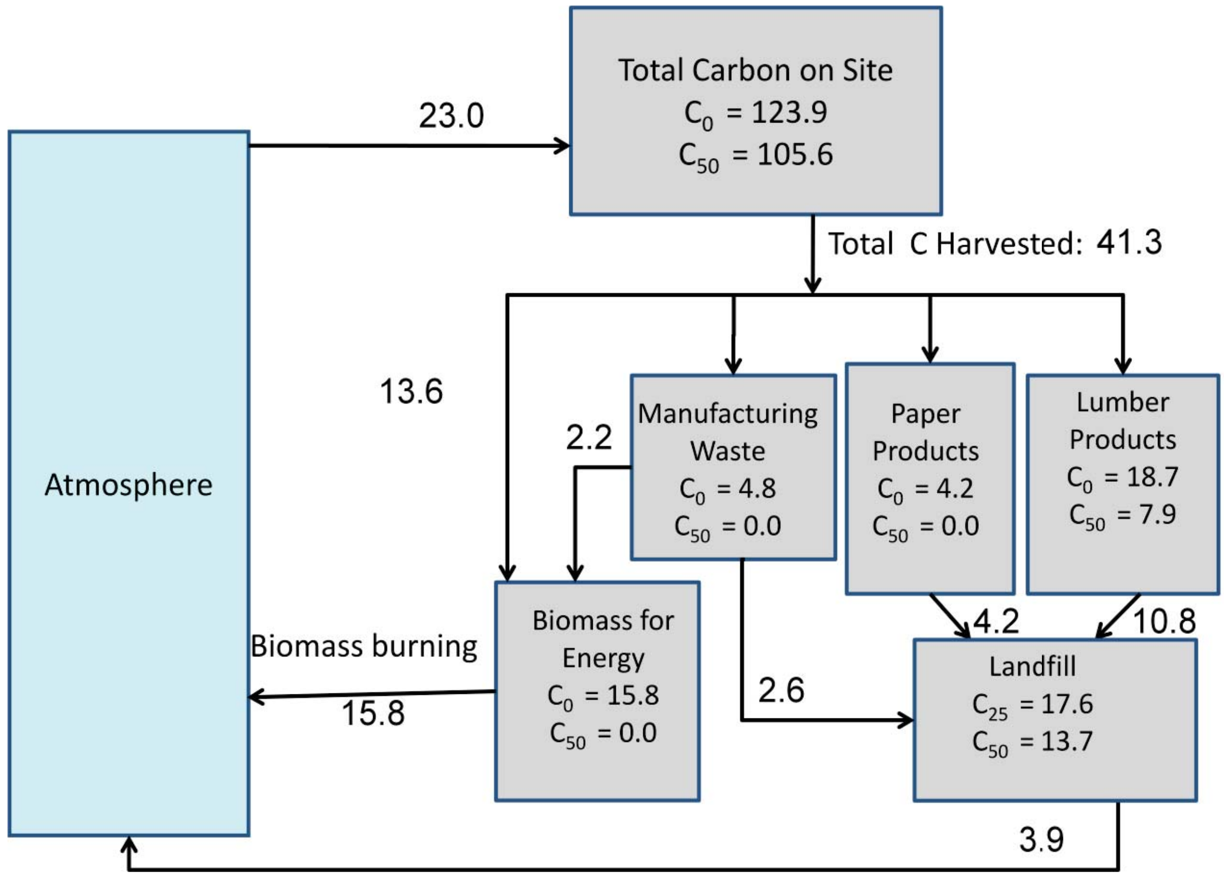

Figure 2. Calculated carbon stores and fluxes associated with a thinned plot. Example for "Heavy thinning scenario". All carbon stores are in MgC/ha. Subscripts indicate year after thinning. For example, C_0 is the carbon store in year 0 immediately following a thinning. The two fluxes accounted for (but not shown) are (1) fossil fuels emissions in harvest operations (1.7 MgC/ha) and offset of fossil fuels from burning biomass (8.3 MgC/ha).

Carbon Accounting Methods Used in this Report

Carbon pools are calculated at 1 year intervals over a 50 year timeframe for each selected plot with the goal to account for all C emissions and sequestration associated with thinning and no-thinning scenarios (Figure 2). The results are shown in Appendix F and the summary carbon budget is calculated by summing up change over 50 years in the following C pools:

- C store on site
- C removed from site by harvest:
  - paper and lumber products
  - manufacturing waste
  - product and waste disposal in landfills
  - biomass for energy

In addition two fluxes (or changes in fossil fuel C store resulting from thinning) were accounted for:

- Emissions from equipment
- Avoided carbon emissions when burning biomass for energy instead of fossil fuels.

Carbon Store on Site

Forest carbon pools are divided into seven categories in the FVS FFE extension:
(1) Standing live trees (above ground),
(2) Below ground live,
(3) Standing dead trees,
(4) Below ground dead,

(5) Forest floor,
(6) Downed dead wood, and
(7) Shrubs and herbs.

The FVS-FFE extension simulates periodic carbon estimates for each of the seven categories. The FVS-FFE biomass estimates (and subsequent carbon estimates) do not include stem bark biomass or stump biomass. Both components have been manually added (using allometric equations) for each tree. Additional details of the model (including allometric equations) are included in Appendix E.

The FVS-FFE model simulations for each thinning prescription projects the following transfers of carbon:

- C in roots of harvested trees is added to below ground dead store.
- C from slash, logging residue, and whole trees ≤3" DBH left on site following a thinning scenario is added to downed dead wood.
- Default regional decay rates with the FVS-FFE model are used for slash/duff/litter.
- C removed from the site is reported as "Carbon removed".

Carbon Fluxes from Thinning Operations

Sources of carbon as a direct result of a thinning operation include carbon emissions from logging equipment (both in the field and on the landing) and carbon emissions from trucks/chip vans. There are several sources of carbon for a thinning scenario, and estimates are based on machine fuel consumption. We assumed all equipment is powered by diesel engines – approximately 6.06 lbs of C are emitted for each gallon of diesel fuel (EPA 2005).

Once a thinning scenario is defined for a given forest stand (e.g., 30 green tons removed/acre, 10% slope, 1 acre/day, 8 hr day, 90 minutes to transport to mill/plant), the amount of carbon released to the atmosphere as part of a thinning scenario can be estimated. Diesel consumption rates vary based on work-load. We estimate fuel consumption rates using an engine work-load factor (Caterpillar 2010), where a load factor of 1.0 indicates that the engine is continuously producing full rated horsepower. For thinning scenarios in this report, relatively low load factors are assumed (0.4-0.5) except for plots with steeper ground slopes, where higher factors are assumed. Diesel is assumed to be 7 lbs/gal, and diesel usage is estimated at 0.4 lbs per hp-hr. Carbon emissions from harvesting equipment can be estimated at a plot level (Table 1).

Table 1. Example of estimated tons of carbon emitted during harvesting and transport for each ton of carbon removed from a thinning. Harvest and transport estimates are based on fuel consumption (lbs) per productive machine hour (PMH). Harvested wood is at 50% moisture content.

Equipment	Est. Maximum Power (HP)	Est. Diesel (gal/PMH)	Est. C (lbs/PMH)	Productivity (tons C from forest/PMH)	Operations (tons emitted/tons from forest)
Feller/buncher*	240	5.49	33.26	3.75	**0.0177**
Grapple skidder*	120	2.74	16.63	3.75	**0.0089**
Log loader	200	4.57	27.72	7.50	**0.0037**
Chipper	300	17.14	103.94	15.00	**0.0069**
Processor	200	4.57	27.72	7.50	**0.0037**
Log Truck	400	8.00	48.50	4.33	**0.0112**
Chip van	400	8.00	48.50	4.33	**0.0112**
Total					**0.0633**

*Assuming that 30 green tons/acre are processed, at 1 acre/day.

In this example, an estimated 0.06 tons (120 lbs) of carbon are emitted by the thinning activity for each ton of carbon extracted (assuming wood that is extracted has 50% moisture content). This estimate would increase for trees farther from the road, and for sites farther from mills/plants decrease for a thinning nearer to the road or the mill. The emissions estimate assumes that chipping is done on site – if forest residues are transported then chipped with an electric-powered chipper (more efficient), overall carbon emissions would likely decrease depending on load density of the transported unchipped residues to the chipping location.

Carbon in harvested material

Carbon removed from each plot by thinning was estimated with FVS. The allocation of removed biomass into forest products depends on many factors, including regional market supply/demand, proximity of processing facilities, wood product quality/species, log sizes, and mill efficiencies. Several assumptions are made in order to estimate final wood products.

In the model, trees are separated into 3 categories: (1) smallest trees (<3" diameter over bark at breast height), (2) small trees (>3" and < 6" diameter over bark at breast height) and (2) larger trees (≥ 6" diameter over bark at breast height). Smallest trees are trampled and left in the field. Small trees have only one product use (biomass for energy), but the end products for larger trees are more diverse. Since most of the trees removed in thinning are relatively small, it is assumed that all logs greater than 6" DBH are transported to a sawmill and then sawn into dimensional lumber, with residues used for paper and energy or disposed of in a landfill.

Wood products are separated as follows:

- Hog fuel ("dirty" chips): All smaller trees (< 6" DBH) and the branches/tops for larger trees that are transported to the landing are fed into a chipper and processed into chips.
- Primary sawmill products: Include dimensional lumber.
- Mill residues: Include "leftover" portions not used in the primary product, such as bark, sawdust, planer shavings, and chips.
  - Bark – may be used for "beauty" bark, energy.
  - Sawdust – may be used for paper, particle board.
  - "Clean" chips – may be used for paper, particle board.

Estimates of sawmill residues and final products are available for Oregon (Brandt et al. 2006). The resulting estimates of sawmill outputs are based on a statewide average recovery factor of 2.07, which varies due to mill efficiency, log size, and scaling. The carbon allocations from mill gate to final product are used to estimate the carbon transferred to various wood products (Figure 3). We assume that lumber and paper products are separated as 62% toward lumber and 27% toward paper.
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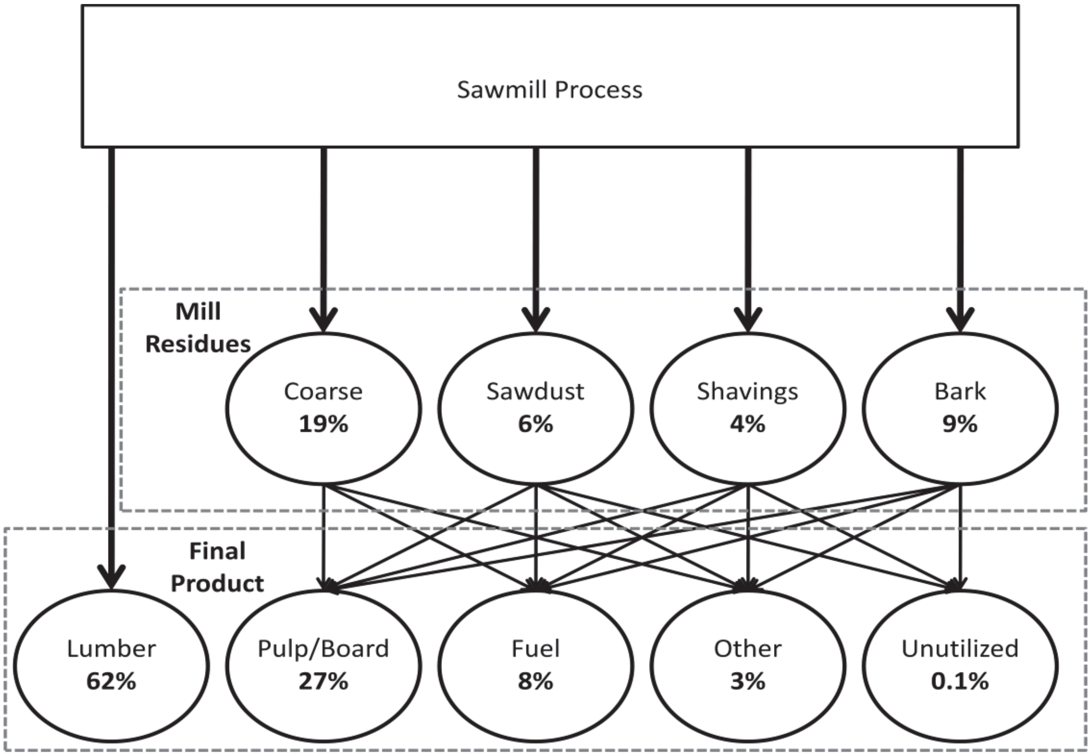

Figure 3. Estimated sawmill residues and final products (by weight), based on Brandt et al. 2006.

Manufacturing waste includes “Fuel”, “Other” and “Unutilized” from Figure 3 as well as carbon from the paper manufacturing process that is assumed not stored within paper. The “Fuel” portion is assumed used toward biomass for energy, and the remaining manufacturing waste is assumed transferred to landfill (with a 1% annual decay rate).

Carbon in wood products

The amount of carbon retained in wood products over time is estimated with an exponential function with set half-lives for each wood product. The method used in this report to estimate transferred carbon over time is similar to the “simple decay” method (Ford-Robertson 2003).

$$Sequestered\ Carbon\ (Year\ x) = \sum_{i=1}^{n} \text{Carbon(Year 0)}_i * \left(\frac{1}{1+\frac{\ln(2)}{\text{halflife}_i}}\right),$$

where $n = number\ of\ products$

There is a wide range of half-lives for wood products - Table 2 shows some examples (Skog and Nicholson 1998). This report takes a simple approach - paper products are assumed to have a half-life of 1 year, timber products a half-life of 40 years, and biomass for energy is assumed to be burned and emitted to the atmosphere within a year.

Table 2. Harvested wood product estimated half-life of carbon (years) for different end uses (Skog and Nicholson 1998).

End Use	Half-Life
Single-family homes(post 1980)	100
Pallets	6
Furniture	30
Paper (long-lived publications)	6
Paper (other)	1

Carbon in landfill

We assume that carbon that is not retained in wood products (both paper and lumber) is transferred to landfill. We make simplified calculations for this pool to estimate the amount at the end of 50-year projection period (while all other pools are estimated on an annual basis (Appendix F). The decomposition rate is 1% per year and the time interval is 25 years (half of 50-year projection period)

Carbon in slash harvested and utilized as source for energy

In the model for this study, all stems <3" DBH are "trampled" (using an FVS keyword) and left on site. This keyword affects crowning and torching index estimates; trampled stems contribute to the downed dead wood carbon pool. The amount of slash from larger trees (>3" DBH) removed from the forest in a mechanized logging operation varies widely. Removal rate estimates of slash from cut-to-length mechanized logging range from 50-75% (Mellström and Thörlind 1981; Sondell 1984).

It is assumed that the removal rate of slash is 80%, using a whole-tree logging system for this study. We assume that the slash removed from site is transported and burned as biomass fuel, instead of piled and burned on site. Transportation costs are included in the model. The 20% of slash left on-site is included as downed-dead wood, and decays over time using default FVS regional decay rates.

In FVS, the torching and crowning indices are impacted by increased fuel loading from slash but the effects are seen only in the short term (less than 5 years) as the slash decays. The effect of slash removal on soil nutrients is an important site dependent factor that should be considered (e.g. Page-Dumroese et al. 2010), but an analysis is not included in this report.

Avoided carbon emissions - comparison of carbon emissions between biomass and other energy sources

Both heat and electricity can be extracted from biomass. The biomass input requirement per MW-hour for a stand-alone biomass electric power generation plant depends on biomass moisture content. The relationship between input biomass and output electric power can be found, assuming that 33% of energy output from the boiler can be utilized for electric power (Table 3). The dry tons of biomass required per MW-hour are a function of biomass moisture content.

Table 3. Estimated forest biomass requirements as a function of wood moisture content.

MC dry basis	MC wet basis	Dry Fraction wet basis	Recoverable BTU/green lb***	Recoverable BTU/ green ton	To Electricity BTU/green ton	To Electricity Kw-hr/ green ton	Green Tons Per Mw-hr	Dry Tons Per Mw-hr	MW-hr Per Dry Ton
0	0.0	1.00	6500	13,000,000	4,333,333	1270	0.79	0.79	1.27
15	13.0	0.87	5400	10,800,000	3,600,000	1055	0.95	0.82	1.21
30	23.1	0.77	4700	9,400,000	3,133,333	918	1.09	0.84	1.19
53.9	35.0	0.65	3700	7,400,000	2,466,667	723	1.38	0.90	1.11
66.8	40.0	0.60	3300	6,600,000	2,200,000	645	1.55	0.93	1.07
81.7	45.0	0.55	3000	6,000,000	2,000,000	586	1.71	0.94	1.07
100	50.0	0.50	2650	5,300,000	1,766,667	518	1.93	0.97	1.04
122	55.0	0.45	2100	4,200,000	1,400,000	410	2.44	1.10	0.91
150	60.0	0.40	1800	3,600,000	1,200,000	352	2.84	1.14	0.88

Given the assumptions from Table 3, the carbon emissions from biomass-produced energy from a stand-alone unit can be estimated and compared to emissions from alternative sources of energy (USDOE 2010) (Table 4). The efficiency of a biomass plant depends on moisture content – the analysis in Table 4 assumes 45% moisture content for forest residues. Table 4 compares carbon emissions between energy source alternatives for biomass combined heat and power (CHP) units, assuming 33% electrical conversion from the boiler. Biomass fuel produces more CO_2 per MW-hour compared to other fossil fuel sources when used as a stand-alone source for power. The difference between biomass and fossil fuel is closer if electric power is not generated, and instead 80% of the energy from the boiler is used for heating. When comparing CO_2 output between forest biomass and fossil fuels, forest biomass has a higher CO_2 production per energy unit produced. This analysis applies only to boiler output, and does not include alternatives or other emissions for each energy source.

Table 4. CO2 output ratios of fossil fuels compared to wood biomass. (fossil fuel estimates from U.S. Dept. of Energy 2000). For example, natural gas releases 38% of CO2 per MW-hour of electricity or 54% of CO2 per MM BTU as compared to the wood biomass.

Stand-alone Electric Plant			
Assumptions:	45% MC (Wet Basis)		
	25 MW plant		
	Uptime: 20 hrs/day		
	33% from boiler converted to electricity		
Calculations	0.94	bone dry tons per MW-hr	
Biomass	0.47	tons Carbon per MW-hr	
	940	lbs Carbon per MW-hr	
	3450	lbs CO2 per MW-hr	
Compare to Biomass			**Percentage of Biomass**
Coal	**2117**	lbs CO2 per MW-hr	**61%**
Petroleum	**1915**	lbs CO2 per MW-hr	**56%**
Natural Gas	**1314**	lbs CO2 per MW-hr	**38%**

Combined Heat and Power			
Assumptions	80% from boiler recovered for heat		
Calculations	4800000	BTU recoverable for heating per green ton	
	0.94	bone dry tons per 4800000 BTU	
	3450	lbs CO2 per 4800000 BTU	
	719	lbs CO2 per MM Btu	
Compare to Biomass			**Percentage of Biomass**
Coal	**620**	lbs CO2 per MM Btu	**86%**
Petroleum	**561**	lbs CO2 per MM Btu	**78%**
Natural Gas	**385**	lbs CO2 per MM Btu	**54%**

Carbon emissions for Energy Alternative

There are several types of coal that are utilized for electric power in the US, and can be classified by its density of carbon. The CO_2 output per pound of coal is lower for ranks of coal with a lower percentage of carbon, but the energy output per pound of coal is smaller as well. Historically, not just carbon emissions are considered when comparing different types of coal – for instance, sulfur compounds are lower for sub-bituminous coal. Coal plants find it cheaper to use coal with lower sulfur content instead of scrubbing coal with higher sulfur content. In the example, sub-bituminous coal outputs are compared to biomass as a substitute source of electric power. Production and transportation emissions are relatively low, estimated as less than 2% of potential energy produced for coal (Spath et al. 1999).

Life of Wood Products – Other Considerations

At least three factors (not directly dealt with in this report) make wood product life cycle assessments difficult (Profft et al. 2009):

- Wood products may be replaced by new products before the physical end-of-use period, for a variety of reasons.
- Some long-lived products (e.g. laminated beams) have largely unknown life spans.
- Some wood waste is disposed of in landfills, and burned wood waste may or may not be used toward energy production.

Regional demands and mill locations may lead to significantly different allocations to different wood products. This could affect the allocation between long-term and short-term wood products, particularly when choosing between particleboard/medium density fiberboard (MDF) (longer lifespan) vs. pulp/paper products (shorter lifespan). Another effect will be the final disposal of wood products. Products would release carbon more quickly if they were burned for energy or other purposes, as opposed to slower release of carbon for wood products that are disposed of in a landfill (Micales and Skog 1997).

Other Carbon Fluxes

Some of the carbon stores and fluxes within a forest as a result of a thinning are recognized, but not quantified. For example, a mechanical thinning will disturb the forest soil (rutting and compaction), and increased disturbance likely increases carbon flux from the soil. However, the net effect on carbon pools within the soil and soil respiration into the atmosphere, while potentially relatively large, is difficult to measure (Ryu et al. 2009), even though some estimates of carbon soil losses have been estimated in agricultural processes (e.g., Smith et al. 2010). As a result of the difficulty in measuring soil carbon stores and fluxes (and no estimates through FVS) it is not included in the model.

Plot Selection

There are 100 plots from five counties (three FVS regions) that have been selected for simulation in FVS (Table 5). The plots are separated into age groups for simplicity when results are presented.

Table 5. Plot Location Summary.

Region	County	Plot Count	Plots at least 160 years
Eastern Cascade	Wasco	21	4
	Jefferson	22	3
Western Cascade	Linn	17	0
	Douglas	15	4
Blue Mountains	Crook	25	4
	Total Plots	100	15

The approximate coordinates of plots in each county are known (Appendix A). The Forest Service plot database uses “fuzzy coordinates”, but estimated locations are within 1 mile of actual plot centers. Plots were selected to represent a range of the “more common” Landscape Ecology, Modeling, Mapping and Analysis (LEMMA 2010) landscape assignments with stand conditions that represent potential for fuel reduction treatments. No other statement of statistical significance is implied.

Dominant Tree Species for each Plot

Basal area was used to determine the dominant species for each plot (Appendix B). Basal area is the total area occupied by the cross-sections of all trees of a species per unit area. Only species with greater than 10% of total basal area are included for each plot in the tables attached in Appendix B, so the cumulative percentage of species for each plot does not always add up to 100% in the tables. In the analysis, all trees are included in the growth model. For most plots, the primary species are Douglas-fir and ponderosa pine. Several other species were commonly found in these plots, including white fir, incense-cedar, and western hemlock.

Plot Understory Vegetation

Plots that were measured from CVS had vegetation codes (Hall 1998) that were input into FVS. Understory vegetation is divided into four classes:

- Forbs
- Grasses
- Shrubs
- Trees

Vegetation species are reported by the number of plots in which they occur (Appendix C). Understory species were used in estimating the vegetation type when not directly reported in the FIA database, but are considered too bulky for this report. The tables use the following definitions:

- Species listed under “trees” refer to trees that are currently growing at the same height as other understory vegetation (shrubs, forbs, grasses). This does not necessarily indicate the species of the dominant trees within a plot.
- Some of the species are ambiguous – for example, “snowberry” is listed separately from “common snowberry” and “creeping snowberry”. The plant definitions for this study are only as precise as the definitions that are available from the source database.
- Only the most common plants were included – if a plant was counted in fewer than 3 plots, it is not included in the summary (but is available).

Table C5 summarizes the number of different plants/ plant groups within each vegetation class that were counted for each plot in four counties.

Carbon Pool Estimates for Plots Prior to Treatment

The Fuels and Fire Extension (FFE) to the Forest Vegetation Simulator (FVS) has integrated reports that estimate forest carbon pools as forest stand growth is simulated. Carbon pool estimates are separated into seven categories:

- Standing live trees
- Belowground live
- Standing dead trees
- Belowground dead
- Downed dead wood (including coarse woody debris)
- Forest floor (including duff)
- Shrubs and herbs

In this analysis, each plot is grown in FVS for 50 years – both the initial carbon pool as well as carbon growth rates are examined and compared to forest volume growth rates to determine site productivity. FVS uses region-specific variants that adjust growth conditions based on regional differences. The Eastern Cascade, Western Cascade, and Blue Mountains variants are used in this study. The plots from each county use the variant recommended by FVS for that county. All plots are simulated and analyzed separately, but only a few of the plots are shown in this report. Plots are chosen from a range of initial conditions. A more detailed explanation of FVS calculations is in Appendix E.

Figure 4 shows carbon estimates for a relatively young stand and Figure 5 for a relatively older stand, assuming no thinning. Note the difference in carbon scales – there is a much lower amount of carbon in the younger stand, but the percentage increase from initial carbon for the younger stand is much higher over the 50-year time frame.

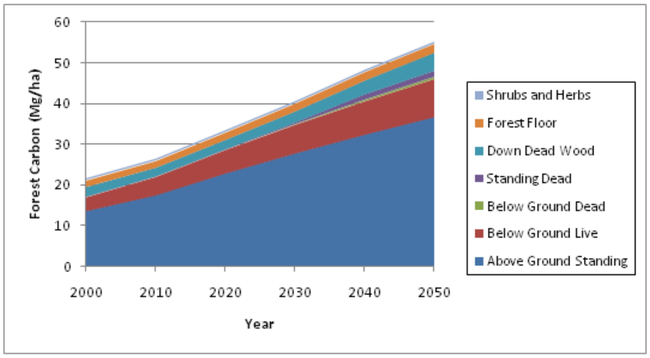

Figure 4. Carbon pool estimates for younger stand.

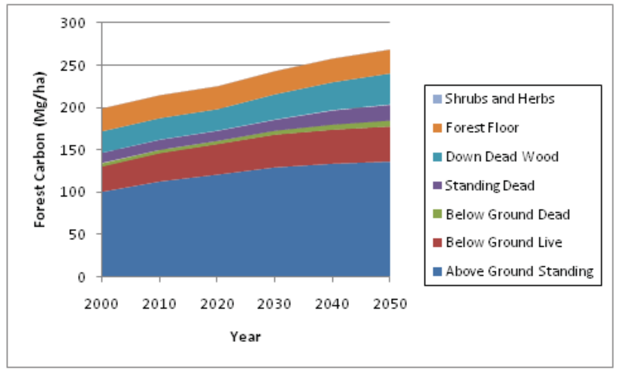

Figure 5. Carbon pool estimates for older stand.

Criteria for Stand Treatments

When thinning the plots, fire hazard was measured using two standard metrics provided by FFE - Torching Index (TI) and Crowning Index (CI). TI is a function of both the vertical stand structure and the height to crown base and CI is a function of crown bulk density (Scott and Reinhardt 2001). The metrics provide the minimum wind speeds required to initiate individual tree torching (TI) and to support a crown fire (CI). The lower the minimum wind speeds the more susceptible the stand is to tree mortality. We use the TI and CI wind speed thresholds used in a recent Oregon/California regional study (Daugherty and Fried 2007). Using these thresholds the stand is a candidate for treatment under one of two conditions:

- TI and CI are both less than 25 mph.
- CI is less than 40 mph, regardless of TI.

Thinning Strategies

In order to determine to test both the sensitivity of forest carbon to thinning intensity and also to include some thinnings that were financially feasible, three different thinning strategies were conducted for each plot.

Light thin

The primary goal of this thinning is to take as few trees as possible while meeting (or exceeding) torching and crowning index criteria. The general approach is to take the smallest trees (0"-6" DBH), and increase by 1" intervals until fuel reduction goals are met. If the TI threshold is met, but the CI threshold was not met, a portion of larger trees (12"-20") is removed. Several plots could not meet the torching and crowning index criteria. These plots tended to be younger stands with smaller diameters and with relatively low crowns.

"Breakeven" thin

In general, the light thinning does not take enough merchantable timber to pay for the thinning. In order to find a feasible thin, larger trees are taken, but trees less than 20" DBH are targeted. Smallest trees are taken first, but in some plots, some of the smaller trees are left behind (because of the relatively higher cost of removal), and some of the larger trees are taken.

Heavy thin

In this thinning strategy, standing trees are thinned to a relatively low number of trees per acre, leaving only the largest trees. Different tree densities are used for plots from eastern Oregon (40-50 trees per acre) and western Oregon (90-100 trees per acre) (Fitzgerald 2005, Tappeiner et al. 1997).

Stand Treatment Considerations

When selecting a system to treat the stands, three primary criteria are considered in this study.

- Impact to carbon pool within each plot (simulated 50 years from current stand condition).
- Comparison of crowning index and torching index before and after treatment.

- Economics of the treatment (treatment must pay for itself for the breakeven thinning scenario).

Other criteria that are important to consider, but beyond the scope of this study, include

- Laws/regulations and public acceptance of potential treatments, particularly on public lands.
- Safety standards and certifications of contractors hired for potential thinning.

A financial analysis was conducted using the Fuel Reduction Cost Simulator (FRCS-West 2010) and LogCost10.2 (2010), while the FVS FFE extension is used to estimate the Torching Index (TI) and Crowning Index (CI), both of which measure stand conditions and hazards that may contribute to a catastrophic fire. The effectiveness of fuel treatment was assessed based on TI and CI estimates before and after thinning. A detailed analysis of TI and CI at a group level is in Figure F1 and F2.

A financial break-even point (where revenues and cost are equal) depends upon a host of factors, some of which are known, and some of which are estimated. There are many potential fuel treatments available within FRCS, including ground-based operations and cable-based operations. In general, the lowest cost systems are ground-based. Ground-based thinning operations can be separated into whole-tree and cut-to-length operations, both which have advantages and disadvantages. One harvesting system is used for plots on more gentle terrain (slopes ≤ 30%), and a slightly different system is used for plots with steeper terrain (slopes >30%).

For more gentle slopes, the following whole-tree system is used:

- Drive-to-tree feller/buncher
- Grapple skidder
- Processing/chipping/loading at the landing
- Truck and trailer transport to nearest mill/plant.

For steeper slopes, the drive-to tree feller/buncher is replaced with a swing-boom feller/buncher, which is more stable on steeper slopes, but is limited to the length of the boom and may lead to less flexibility in tree removal. For longer skidding distances, the cut-to-length system (CTL) becomes less expensive than whole-tree skidding due to the higher load carrying capability of forwarders. CTL systems can also have lower mobilization costs, important in small, low volume treatment units, because fewer pieces of equipment are transported between harvest units.

Example Plot

The following example details a plot that is assessed with the model created for this study. In order to fully describe the analysis for each plot, one of the plots (21561) from Jefferson County (eastern Oregon) was chosen. Plot parameters are known (Table 6), and the analysis for this plot follows.

Table 6. Summary information for the example plot (metric, English units).

Plot Attributes	
Species	Ponderosa pine, Douglas-fir
Age	72 yrs avg for dominant/codominant trees
	Uneven aged stand ranging from seedlings to >200 years
Basal Area	152 ft^2/acre (35 m^2/hectare)
Height	69 ft (21 m) avg for dominant/codominant
Initial Wood Volume	3390 ft^3/acre (237 m^3/hectare)
Initial C Store (live aboveground)	28.7 tons/acre (64.4 MgC/hectare)
Initial C Store (total)	49.3 tons/acre (110.4 MgC/hectare)

Torching and Crowning Index

Initial FVS estimates for TI (38 mph) and CI (32 mph) indicate that the stand is a candidate for fuel treatment, because CI < 40 mph. The slope is gentle for this particular stand (<5%), so a drive-to-tree feller/buncher is chosen as part of the whole-tree mechanical thinning system

Silvicultural Prescription and Carbon Effects

- The plot initially has 380 trees/acre. Similar to the other plots, this plot has three implemented scenarios for thinnings (light, heavy, and break-even); this example has three scenarios to illustrate general relationships between economics and fuel reduction for most plots. Silvicultural prescriptions implemented for this particular stand includes:

Trampling smaller fuel sources to reduce fuel loading as part of the drive-to-tree feller/buncher operation. Including trampling as an option in FVS reduces fuel depth by a factor of 0.75. This affects fire intensity (increases TI and CI) but does not affect fuel consumption in a potential fire. (Reinhardt et al. 2003).

"Light" Thinning
(208 trees/acre remaining – TI=38, CI=54):

- Removing 100% of trees less than 10 in. DBH
- The resistance to crown fire is improved and resistance to individual tree torching is unchanged.

"Break-even" Thinning
(164 trees/acre remaining – TI =40, CI=54):

- Removing 100% of trees less than 7 in. DBH
- Removing 20% of trees 7-20 in. DBH
- Corresponds to a removal of fewer smaller trees and a higher number of larger trees while marginally meeting fuel reduction goals.

"Heavy" Thinning
(46 trees/acre remaining – TI=39, CI=66):

- Removing 100% of trees less than 12 in. DBH
- Removing 30% of trees 12-16 in. DBH
- Removing 10% of trees 16-20 in. DBH
- Leaves the stand in a relatively park-like condition, with little understory and only a few of the largest trees remaining. This stand structure might simulate some eastern Oregon historical structures (Fitzgerald 2005). Both resistance to torching and crowning have significantly increased.

All thinnings reduce forest carbon pools, and heavier thinnings lead to less carbon on-site than lighter thinnings, both immediately and over the 50-year simulated period. Plot-level estimates of carbon pools, carbon transfer to wood products, and potential avoided carbon emission by biomass burning for energy (compared to a coal alternative) are compared (Figure 6). Twenty percent of the slash created from harvested trees is left in the stand following a thinning. The live wood volume in Figure 6 is total live green volume/unit area (m^3/hectare), and is included as both a reference and as an additional metric to manually check for any gross discrepancies in the growth and yield model.

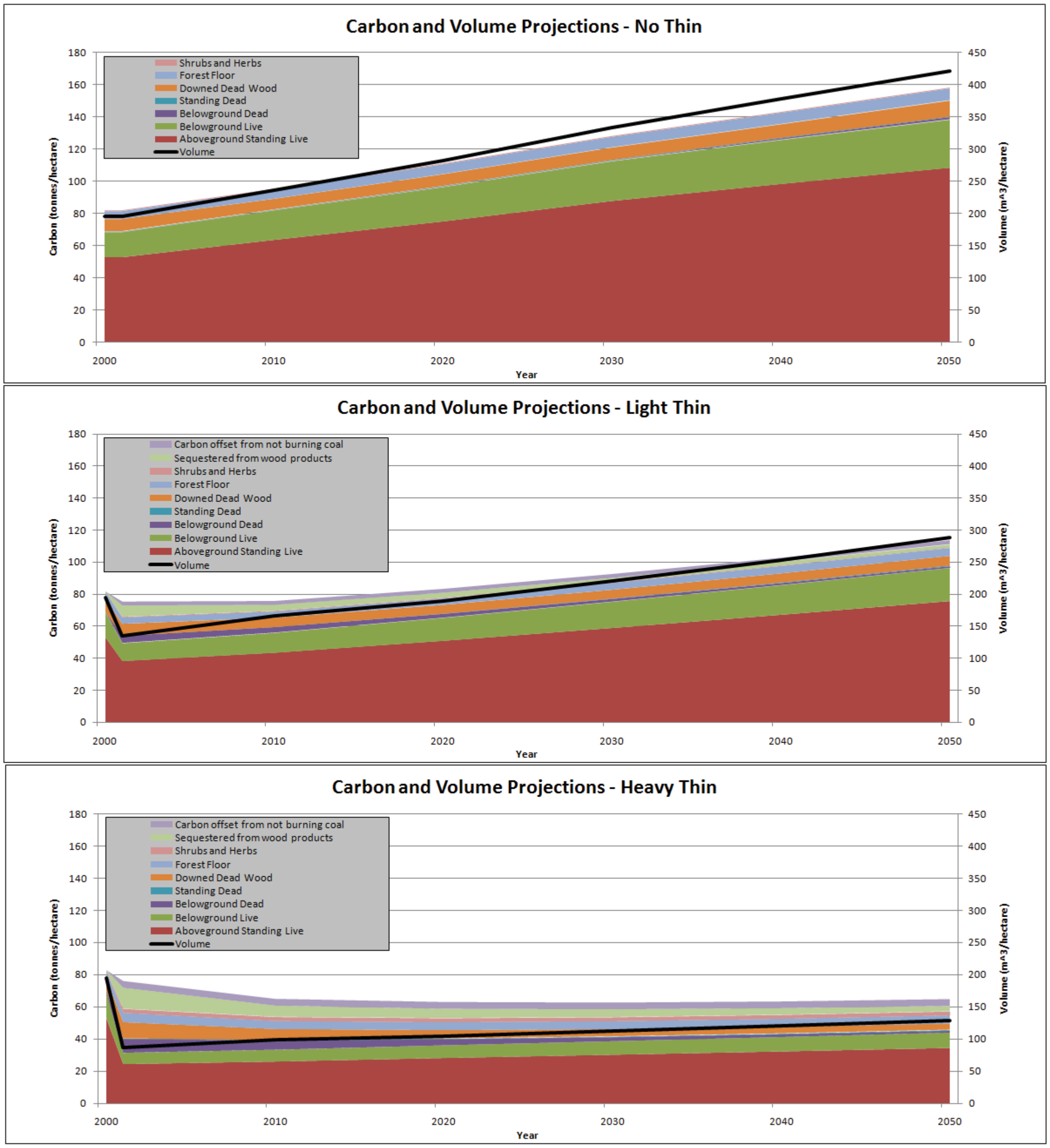

Figure 6. Simulation of carbon pools for the forest stand – No Thin (top), Light Thin (middle) and Heavy Thin (bottom).
All carbon components reference the left axis. Only standing green tree volume (Volume) references the right axis.

Harvesting System

The harvesting system for this stand includes five major pieces of equipment and two types of transportation vehicles:

- Drive-to-tree feller/buncher – Mechanically falls each tree and lays trees into groups (bunches) for efficient handling.
- Grapple skidder – Grabs whole tree bunches and drags trees to a roadside landing.
- Processor – Located at the roadside landing. Delimbs and bucks trees into merchantable lengths.
- Chipper – Located at the roadside landing. Chips small whole trees (< 6" DBH) and tops and branches from larger trees directly into a chip van.
- Loader – Located at the roadside landing. Maneuvers small whole trees and residues into the chipper and logs into log trucks.
- Truck with Chip Van – Transports chips from landing to destination. Capacity for vans in this example is 110 cubic yards.
- Truck with Log Trailer – Transports logs from landing to mill.

This is a thinning system that removes whole trees to the landing. There is a potential for residual stand damage that must be considered in both harvest planning and operations.

A Cut-to-Length (CTL) system could be used at a comparatively lower cost for thinning at longer skidding distances when compared to a whole-tree system (Kellogg et al. 2010), but a CTL system was not included in the final economic analysis, since average skidding distance in this report is assumed to be 500 feet (also assumed by Dempster el al. 2008).

Costs

Costs are separated into four components:

- Planning/administration costs – includes timber sale preparation and administration. Sales preparation and administration estimates for nonfederal (Nall 2010, Sessions et al. 2000) and national forest land (TSPIRS 2001, adjusted for inflation) are estimated in Table 7. The federal land administrative costs are not included in the "breakeven" analysis, and administrative costs vary widely from sale to sale, according to federal requirements, including compliance with the National Environmental Policy Act (NEPA) and other federal laws (e.g., USFS 2010). In general, federal land sales preparation and administration costs are higher compared to private land. The estimate used in the example is a general example only, and should not be used to estimate actual costs.
- Setup costs – includes one-time move-in cost to an area, moving costs from landing to landing, sales preparation cost, and road maintenance costs (Table 8).
- Cost from field to truck, including felling/bunching, skidding, chipping, processing, and loading (Table 9).
- Cost to transport each wood product (Table 10).

The planning/administration costs are shown, but are not included in the final analysis.

Table 7. Sales preparation and administration costs associated with the three thinning scenarios.

Preparation/Administration Costs	light	heavy	break-even	units
Sales Preparation/Admin (Non-federal)	42	26	32	$/mbf
	141	**143**	**142**	$/acre
Sales Preparation/Admin (Federal)	173	173	173	$/mbf
	581	**964**	**765**	$/acre

Table 8. Estimated equipment setup costs for the three thinning scenarios.

Setup Costs	light	heavy	break-even	units
Move-in Cost (5 equipment pieces)	33	33	33	$/day
1 acre average/day	33	33	33	$/acre
Moving Costs (Landing to Landing)	75	75	75	$/acre
Road Maintenance Costs	33	59	37	$/acre
Total Setup	**141**	**168**	**145**	**$/acre**

Table 9. Estimated costs from field to truck for the three thinning scenarios.

Cost from Field to Truck	light	heavy	break-even	units
Felling/bunching	225	441	133	$/acre
Skidding	210	455	252	$/acre
Chipping whole trees	26	51	13	$/acre
Chipping loose residues	14	40	32	$/acre
Processing Logs	162	392	183	$/acre
Loading Logs	71	192	130	$/acre
Total to Truck	**708**	**1571**	**743**	$/acre

Table 10. Estimated truck transport cost for the three thinning scenarios.

Cost from Truck to Final Destination	light	heavy	break-even	units
Chip Trucking (Transport + Delays)	16	16	16	$/green ton
	133	**213**	**127**	$/acre
Log Trucking (Transport + Delays)	48	48	48	$/mbf
	134	**267**	**167**	$/acre
Total Truck to Final Destination	**266**	**480**	**295**	$/acre

Wood Products

The volume and mix of wood products derived from the thinning is critical when calculating total revenue from the stand. The mix of trees removed from the plot is separated by diameter class (Table 11). FVS simulated the total volume (ft^3) per plot and merchantable volume (Mbf) in order to estimate timber value. A 16 ft scaling rule (Scribner) was used for plots in eastern Oregon, and the midrange diameter was used to estimate the Mbf:cf ratio for each diameter class (e.g., 7" was used for 6"- 8" sawtimber) which was found with a conversion chart (Mann and Lysons 1972 – Fig 4).

Table 11. Allocation of thinned trees into wood products.

Products	light	heavy	break-even	units
Saw timber				
6"-8"	1.77	3.53	1.41	CCF/acre
8"-10"	2.78	5.56	2.23	CCF/acre
10"-12"	1.01	2.03	0.81	CCF/acre
12"-16"	1.01	1.52	2.54	CCF/acre
16"-20"	0	0.45	0.5	CCF/acre
Total Saw Timber	**2.78**	**5.57**	**3.51**	mbf/acre
Chips	**8.1**	**13.0**	**8.2**	tons/acre

Sawlog prices are estimated using the Oregon Department of Forestry Log Price Information (Oregon Dept. of Forestry 2010). The biomass market returns significantly lower prices than the pulp market, but it is assumed that the biomass chip quality does not meet pulp chip standards (Table 12).

Table 12. Estimated delivered harvested wood product prices.

Market	price	units
Sawlogs	285	$/mbf
Chips	60	$/BDT

Overall Cost/Revenue Analysis

For this particular scenario with the given assumptions, there is **a net profit of $72/acre** for the "breakeven thin" scenario on non-federal lands (Table 13). Both the "light" and "heavy" thin result in treatment costs exceeding revenues given the initial assumptions. These three different thinning scenarios demonstrate that increasing gross revenue or total volume does not necessarily improve net revenue, and depending on original stand structure, may significantly increase harvesting costs. In order for this thinning to not incur financial losses on federal lands, a relatively high proportion of high-value stems and a relatively low proportion of low-value stems would need to be thinned.

Table 13. Total costs and revenues, using non-federal costs - per acre basis.

Revenue	light	heavy	break-even	units
Sawlogs	794	1421	1011	$/acre
Biomass	243	390	243	$/acre
Gross Revenue	1037	1811	1254	$/acre
Minus Costs	1116	2219	1182	$/acre
Net Revenue	-79	-409	72	$/acre

The amount of carbon in the stand after 50 years compared to the initial carbon pool varies with the intensity of the thinning and the type of thinning. Using the initial amount of live aboveground carbon and total aboveground carbon as a benchmark, the net effect on carbon after 50 years (excluding wood products or avoided carbon emissions) can be estimated (Table 14).

Table 14. Simulated carbon outputs, excluding harvested wood products.

Carbon Pool	Treatment	Year 0	Year 50
Total Carbon (MgC/ha)	No Thin	83.2	158.7
	Light Thinning	71.2	99.2
	Heavy Thinning	59.8	70.6
Aboveground Live Standing Carbon (MgC/ha)	No Thin	53.6	105.6
	Light Thinning	35.8	59.1
	Heavy Thinning	29.8	43.6

Analysis

Other plots in this analysis were analyzed in a similar way to the example plot, with the primary difference in prescriptions between plots being the number and class of trees removed. The analysis methodology was the same between plots and regions.

Several harvesting assumptions are made – average skidding distance is 500 ft for all plots, which is highly variable, and directly affects cost. There are 16-foot Scribner scaling rules used for plots east of the Cascades and 32- foot Scribner scaling rules for plots west of the Cascades. Different prices per Mbf are used for both eastern and western Oregon, and a 20% premium is assumed for plots in western Oregon, due to differences in scaling rules. However, the price will also differ between regions at any given time due to species differences, market conditions, and other factors.

Biomass price is assumed to be $60/ton throughout the region – biomass price fluctuates, and the profitability will be greatly impacted by the market price. Lower prices would make it much more difficult for the landowner to “breakeven”. To reduce cost, the landowner may take the approach of only removing the most “profitable” biomass (e.g., biomass near a roadside, biomass in areas with shorter transport distance to final destination).

Detailed thinning prescriptions and plot-level ranges of carbon estimates were made for each plot. The general trends (minimum, maximum, and average) of carbon estimates for all plots are split into two regions (eastern Oregon and western Oregon), and are included in Appendix D. Detailed Tables are included (Appendix F).

Results

For most plots, forest carbon pools (both live aboveground and total) are significantly reduced when comparing thin to no thin. After simulating growth in the stands for 50 years the average difference in net carbon balance between unthinned and thinned plots for the three age groups ranged between 73.5 – 103.4 MgC/ha in Eastern Oregon to 121.8 – 128.6 MgC/ha in Western Oregon.

Carbon levels of thinned plots do not reach the carbon levels of unthinned plots within a simulated timeframe of 50 years, even after including carbon transferred to harvested wood products and the avoided emissions from using biomass instead of fossil fuels for energy. See Table 15 for an overall carbon budget by thinning scenario and region. See Appendix F for a group-level summary of carbon stores (Table F1), relative carbon flux over time (Table F2), fuel loading measurement (Table F3), and plot-level comparison of carbon stores (Table F4).

- Older stands, which tended to have lower carbon flux annually (as a percentage of initial carbon stores), did not "recapture" carbon as quickly as younger stands following a light thinning.
- All stands had lower carbon flux into the stand from the atmosphere following a heavy thinning, when compared to a lighter thinning or no thinning.
- Stands in eastern Oregon tended to have less carbon flux when compared to stands in western Oregon.

Regarding wood products:

- Larger trees had a greater percentage of carbon transferred to wood products with a relatively longer half-life for carbon. Smaller trees had a greater percentage of carbon transferred to products with a shorter carbon half-life (such as paper or burning for biomass).
- Carbon dioxide output per unit energy produced is higher for biomass stand-alone facilities compared to fossil fuels, but the gap is closed somewhat if energy is used for heating instead. This study ignores other pollutants (such as SO_x emissions), that are higher for coal when compared to biomass (NREL 2000).

Financial analysis:

- With the additional goal of no financial loss, a higher percentage of larger, more valuable trees must be thinned in order to cover the cost of removing smaller, less valuable trees.
- Heavy thins were often unprofitable, and depended on the assumptions in the economic model as well as original stand structure. There are many fuel reduction treatments that were not included, such as mastication or slash piling. These alternative techniques might reduce costs by leaving smaller stems in the field, but would also affect carbon impacts and potentially affect crowning and torching indices.

The estimated carbon budget for these plots (based on carbon stores and fluxes - Figure 2) is shown (Table 15).

Table 15. Carbon budgets for thinning and no-thinning scenarios (all age groups combined; time interval = 50 years; units are MgC/ha).

Region	Thinning Scenario	Net Change on Site	Equipment Emissions	Paper products	Lumber products	Paper/ lumber disposal (in landfills)	Emissions from biomass burning for energy	Offset from not burning fossil fuels	Net Carbon balance	Average Annual Sequestration Rate MgC/ha/year	Difference between Thin and No Thin
Eastern Oregon	No Thinning	90.67	0.00	0.00	0.00	0.00	0.00	0.00	90.67	1.81	0.00
	Light Thinning	14.53	-0.59	0.00	2.08	4.05	-7.48	4.56	17.15	0.34	-73.52
	"Break-even" Thinning	-15.81	-1.05	0.00	4.52	8.80	-10.56	6.44	-7.66	-0.15	-98.33
	Heavy Thinning	-27.54	-1.62	0.00	7.47	14.54	-14.35	8.76	-12.75	-0.25	-103.42
Western Oregon	No Thinning	81.97	0.00	0.00	0.00	0.00	0.00	0.00	81.97	1.64	0.00
	Light Thinning	-40.22	-0.48	0.00	1.20	2.10	-6.21	3.79	-39.82	-0.80	-121.79
	"Break-even" Thinning	-53.62	-1.30	0.00	7.06	9.88	-10.85	6.62	-42.21	-0.84	-124.18
	Heavy Thinning	-58.31	-2.26	0.00	8.64	11.08	-14.88	9.08	-46.65	-0.93	-128.62

*All Units in MgC/ha except for annual sequestration rate

Financial Sensitivity

Some of the plots dominated by smaller stems could not be thinned without financial loss, given the assumptions for these plots. For instance, Plot 26510 (Wasco County) has a relatively high density (538 trees per acre), but quadratic mean diameter (QMD) is 7", and the largest trees are 10" DBH. Varying the thin affects the financial loss per acre, even for nonfederal land. For instance, a thinning to 200 trees per acre using initial assumptions results in a net loss of -$503/acre (Table 16).

Table 16. Initial financial loss for Plot 26510.

Revenue		
Sawlogs	1104	$/acre
Biomass	432	$/acre
Gross Revenue	**1536**	$/acre
Minus Costs (non-federal)	**2038**	$/acre
Net Revenue (non-federal)	**-503**	$/acre

However, given different assumptions, it is feasible for this thinning to break even or turn a small profit for the landowner. Financial feasibility is improved if (1) the harvested wood is closer to the landing, (2) the transport distance to a mill/plant is shorter, (3) higher wood product market prices exist and (4) harvest units are larger and closer together. Incremental changes to these four factors can together dramatically affect cost or revenue for this plot (Table 17).

Table 17. Favorable conditions allow the landowner to financially break even. Net revenue reflects cumulative changes of assumptions. For example: reducing skidding distance improves net revenue from -$503/ac to -$291/ac and simultaneously shortening log truck travel time improves net revenue to -$201/ac.

Variable	Original Value	New Value	Net Revenue ($/acre)
Initial Condition			-503
Average Skidding Distance	500 ft	100 ft	-291
Log truck 1-way travel time	1.5 hr	0.5 hr	-201
Chip van 1-way travel time	2 hr	0.5 hr	-96
Revenue for biomass	$60/BDT	$80/BDT	48
Revenue for timber	$285/mbf	$305/mbf	136
Size of harvest unit	40 acres	100 acres	211
Cost to move into harvest unit	$3000 total ($600 each)	$2500 total ($500 each)	219

If the landowner's decision is largely focused on profit or loss, these factors must be carefully considered. In order to decrease skidding costs, the landowner may decide to harvest only near the roadside, or may only harvest on flatter terrain. It is also more likely that regions nearest mills and plants and existing road infrastructure would be thinned, due to decreased transport distance. Activities in marginal stands may be postponed until periods of higher markets or treatment in marginal stands combined with more profitable stands to create a breakeven situation. Depending on objectives, the landowner may leave the plot untouched, or may apply another management prescription.

Other socio-political factors could affect landowner decisions in both short and long term. Subsidies for forest biomass (e.g., $10/green ton subsidy – HB2210 Oregon 2007) can increase revenues and allow thinning to become more economically viable. Price premiums for carbon from public or private sources may also affect a landowner's decision. Uncertainty associated with these potential sources of revenue would be considered by the landowner in long-term planning.

Potential Alternative Management for Younger Stands

For many of the younger stands (especially stands with relatively low QMD and relatively high trees/acre), it was not possible to simultaneously thin the stand to the desired TI and CI while maintaining a profit, given the harvesting and market assumptions. For these stands, there are several alternatives that may be considered for fuel reduction:

- Alternative silvicultural prescriptions, such as prescribed fire, could be used to reduce fuels while initiating some level of stand mortality and raising base to the live crown.
- Only the least expensive areas could be thinned – for example, treating only the areas nearest roadside, areas with flatter terrain, or areas nearest the mill would reduce cost while still implementing some level of fuel reduction.
- Leave the stand "as is", and potentially treat the stand at a later time after the stand naturally reaches a different stand structure.

Other Carbon Fluxes

The thinning analysis in this paper addresses the effect on carbon pools from removing selected trees from a stand in an effort to improve forest resilience to fire. The reference scenario is the "no treatment" scenario. For some owners, this may be appropriate, but for others, alternative reference scenarios may be more useful. For example, do longer rotations with one or more thinnings sequester more carbon than shorter rotations with no thinnings? In this case a short rotation with no thinning becomes the reference scenario. Or does uneven-aged management sequester more carbon than even-aged management? In this case even-aged management becomes the reference scenario.

We also do not address carbon fluxes from precommercial thinning (PCT) where trees are currently thinned to waste as compared to the options of planting lower tree densities or delaying PCT until the trees increase commercial value.

Lastly, we not address the effect on carbon pools from utilization of forest residues following a commercial harvest operation where residues are piled at roadside as part of the normal harvesting operations and later burned to reduce fuel hazard, release area for new plantations, and to reduce habitat for rodents. In this case slash burning and short term release would be the reference scenario as compared to residue utilization for energy substitution.

Next Steps

Future analysis could

- Simulate wildfire and prescribed fire over long timeframes in stands with and without thinning in order to more fully understand the effects of wildfire on carbon pools.
- Broaden carbon accounting to include the substitution of wood products for building materials such as concrete, steel, and aluminum.
- Simulate the effects on carbon pools and fire after either natural seedling in-growth or planting in the understory.

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Appendix A. Coordinates of Plots for each County

Figure A1. Wasco County plot locations.

Figure A2. Jefferson County plot locations.

Figure A3. Linn County plot locations.

Figure A4. Douglas County plot locations.

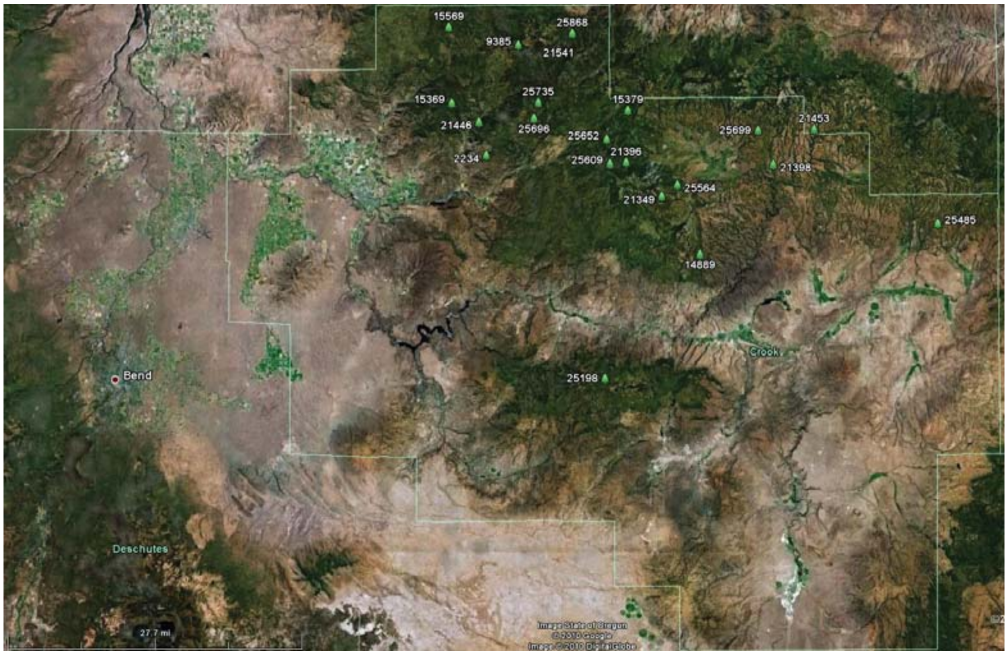

Figure A5. Crook County plot locations.

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Appendix B. Stand Level Characteristics for each Plot, by County

Table B1. Wasco County - dominant species (percentage of total basal area) and associated tree data.

Plot	Douglas-fir	Ponderosa pine	Oregon white oak	Grand fir	Western hemlock	Noble fir	Incense-cedar	Western juniper	Average Diameter All > 1 inch* (in)	Stand Height (ft)	Trees/acre > 1 inch Diameter	Age of Dominant Trees (years)
3505		100%							6	15	164	21
26510	100%								7	41	538	25
21980		73%							7	26	223	29
26469	19%	79%							12	55	176	30
3520	34%	32%		34%					8	30	153	32
3514	67%	16%					17%		8	28	277	36
3502	46%	54%							9	26	328	39
3487	93%								5	31	1255	45
3515		100%							9	34	224	49
3501	70%	14%		16%					11	52	308	53
3479		80%	20%						6	28	217	64
3465	61%			20%					17	76	283	77
17061	43%	51%							11	45	202	77
3491	79%	20%							11	52	138	83
26512	63%			36%					16	80	314	86
26557	70%	10%	20%						10	50	324	93
26556	65%			31%					16	76	528	93
16868	54%		40%						10	42	271	93
3528			89%					11%	13	30	120	100
3495	24%	76%							14	66	101	104
26554	49%				10%				29	105	826	160
16781	51%	41%							21	89	544	175
21889	20%				70%				15	46	219	212
26553	18%				65%				22	102	292	269
21979	15%			18%		54%			20	88	370	179

Table B2. Jefferson County - dominant species (percentage of total basal area) and associated tree data.

	Douglas-fir	Ponderosa pine	White fir	Grand fir	Incense-cedar	Mountain hemlock	Average Diameter All > 1 inch* (in)	Stand Height (ft)	Trees/acre > 1 inch Diameter	Age of Dominant Trees (years)
2626	86%	14%					5	47	593	27
2638	14%	86%					8	32	234	41
2667	63%	12%			25%		8	35	408	53
2636		100%					9	43	104	55
25835	38%		53%				11	41	860	57
2652	42%			37%	12%		10	41	948	61
15703	49%	39%					11	43	560	63
2624	18%	82%					8	37	184	69
25766		95%					9	28	460	71
21561	33%	52%			16%		11	43	380	71
25725		87%			12%		9	33	854	72
21585						87%	10	45	989	80
2668		90%			10%		7	6	279	80
15472	21%		65%				16	55	219	92
2641						75%	12	50	506	95
25856	45%	18%			29%		16	59	134	101
2639	39%	30%	21%				16	74	319	108
25834	77%	19%					13	42	417	110
9376	52%		42%				20	69	234	112
25926	46%	25%	21%				18	70	554	114
25905	35%	45%	10%				18	70	1410	171
21440	32%	59%					24	60	416	177
15476	14%	46%	40%				25	89	849	185

Table B3. Linn County - dominant species (percentage of total basal area) and associated tree data.

Plot	Douglas-fir	Western hemlock	Bigleaf maple	Red alder	Western redcedar	Average Diameter All > 1 inch* (in)	Stand Height (ft)	Trees/acre > 1 inch Diameter	Age of Dominant Trees (years)
1513	81%			17%		4	18	714	10
25468	87%	12%				9	61	321	26
30478	94%					10	66	342	26
29297	95%					9	47	198	27
30464	61%	37%				8	47	467	27
1641	67%	31%				8	48	1038	29
1536	88%	12%				10	69	768	32
1529	84%	16%				9	65	877	35
1573	88%	12%				14	75	102	35
15694	82%	13%				14	89	329	47
30451	60%		28%			19	98	236	50
29290	86%					19	108	107	53
1658	97%					20	109	163	55
1522	79%		21%			17	116	121	58
30370	97%					20	116	134	64
30446	59%	13%	19%			14	70	284	66
30383	98%					15	91	175	66
30465	94%					24	121	118	75
21385	95%					12	76	569	84
25874	57%	13%			14%	12	63	1505	88
21705	92%					13	83	508	91

Table B4. Douglas County - dominant species (percentage of total basal area) and associated tree data.

Plot	Douglas-fir	White fir	Lodgepole pine	Incense-cedar	Western hemlock	Mountain hemlock	Bigleaf Maple	Pacific madrone	Ponderosa pine	Average Diameter All > 1 inch* (in)	Stand Height (ft)	Trees/acre > 1 inch Diameter	Age of Dominant Trees (years)
746	85%			15%						7	43	454	23
662	100%									8	54	367	26
739	90%						10%			12	89	193	43
714	96%									12	95	264	47
24035	87%									15	94	622	60
776	100%									15	85	614	66
8530	61%	33%								19	72	276	69
24055	51%	30%		17%						13	63	599	70
23952	67%							30%		11	54	792	73
24317	21%	31%	48%							8	44	1128	74
704	100%									23	124	114	84
24357	73%	21%								16	87	251	84
24239			99%							12	66	863	89
24202			96%							6	34	1319	91
24397	56%				39%					16	62	1087	92
24131	72%	14%		12%						22	96	596	94
12695	76%						23%			31	156	123	99
24275			43%			57%				8	41	729	105
24098	59%	10%			30%					23	100	333	122
24015	45%			35%					17%	16	78	927	123
24209	49%			16%		18%				27	110	366	194
24213	49%	13%		32%						39	147	263	198
24289	59%					30%				18	99	463	212
30188	74%						16%			46	181	201	216
23929	73%					19%				37	135	120	220

Table B5. Crook County - dominant species (percentage of total basal area) and associated tree data.

Plot ID	Ponderosa Pine	Douglas-fir	Grand Fir	Western juniper	Average Diameter All > 1 inch* (inch)	Stand Height (ft)	Trees/acre > 1 inch DBH	Age of Dominant Trees (years)
21446	58%	28%		14%	10	40	530	92
2234	89%	11%			10	44	456	93
25868	57%	43%			13	48	156	108
25699	66%	33%			14	54	869	113
15379	80%	17%			17	65	276	117
25485	82%			18%	13	41	196	117
25198	7%	29%	64%		13	59	838	118
25652	9%	26%	62%		17	70	560	123
25564	57%	39%			17	65	387	126
25609		27%	61%		17	66	725	127
21398	98%				11	43	171	128
15369	74%	25%			21	81	328	144
25735	91%				21	76	208	145
9385		9%	72%		21	92	463	155
21541	23%	18%	58%		27	100	135	155
25696	61%	38%			12	36	267	158
14889	82%				17	63	228	159
21396	22%		65%		20	81	458	160
21453	69%	10%		21%	17	51	334	172
21349	85%	15%			25	85	171	200
15569	75%	14%	10%		20	68	194	227

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Appendix C. Understory Vegetation by County

Table C1. Most common understory vegetation for Wasco County plots.

Category	Common name	Plots
Shrubs	dwarf rose	17
	common snowberry	15
	Cascade barberry	12
	snowbrush ceanothus	10
	antelope bitterbrush	9
	greenleaf manzanita	9
	oceanspray	9
	Saskatoon serviceberry	9
	creeping snowberry	8
	giant chinquapin	8
	pipsissewa	8
	California blackberry	7
	snowberry	6
	willow	6
	California hazelnut	5
	Oregon boxleaf	5
	vine maple	5
	creeping barberry	3
	honeysuckle	3
	plum	3
	thinleaf huckleberry	3
Trees	ponderosa pine	7
	Douglas-fir	6
	Oregon white oak	3

Category	Common name	Plots
Forbs	common yarrow	8
	blue windflower	7
	bride's bonnet	7
	fragrant bedstraw	7
	largeleaf sandwort	7
	lupine	7
	woodland strawberry	7
	Columbian windflower	6
	Pacific trillium	6
	sweetcicely	6
	Virginia strawberry	6
	houndstongue hawkweed	5
	western brackenfern	5
	American trailplant	4
	arrowleaf balsamroot	4
	twinflower	4
	western rattlesnake plantain	4
	white hawkweed	4
	broadleaf starflower	3
	leafy pea	3
	liverleaf wintergreen	3
	sidebells wintergreen	3
	sweet after death	3
Grasses	Idaho fescue	12
	California brome	9
	cheatgrass	8
	sedge	8
	bluebunch wheatgrass	5
	Grass, annual	5
	western fescue	5
	Geyer's sedge	4
	Kentucky bluegrass	4
	pinegrass	4
	squirreltail	4
	Columbia brome	3

Table C2. Most common understory vegetation for Jefferson County plots.

Category	Common name	Plots
Shrubs	greenleaf manzanita	13
	snowbrush ceanothus	12
	antelope bitterbrush	12
	Saskatoon serviceberry	12
	pipsissewa	12
	common snowberry	10
	dwarf rose	8
	giant chinquapin	7
	prostrate ceanothus	5
	Cascade barberry	3
	hollyleaved barberry	3
Trees	incense-cedar	5
	ponderosa pine	5
	Douglas-fir	4

Category	Common name	Plots
Forbs	Virginia strawberry	10
	tailcup lupine	9
	white hawkweed	9
	houndstongue hawkweed	8
	arrowleaf balsamroot	7
	common yarrow	7
	Forb, dicot	7
	western brackenfern	6
	Nevada pea	5
	broadleaf starflower	4
	bull thistle	4
	glaucous beardtongue	3
	largeleaf sandwort	3
Grasses	Idaho fescue	9
	long-stolon sedge	8
	squirreltail	7
	California brome	5
	western fescue	5
	cheatgrass	4
	pinegrass	4
	bluebunch wheatgrass	3
	Grass, perennial	3
	Ross' sedge	3
	western needlegrass	3
	Wheeler bluegrass	3

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Table C3. Most common understory vegetation for Linn County plots.

Category	Common name	Plots
Shrubs	vine maple	18
	California blackberry	17
	Cascade barberry	16
	salal	16
	red huckleberry	14
	dwarf rose	9
	California hazelnut	9
	Pacific rhododendron	8
	pipsissewa	7
	oceanspray	7
	creeping snowberry	4
	oval-leaf blueberry	4
	common snowberry	3
	willow	3
	salmonberry	3
	thinleaf huckleberry	3
Trees	western hemlock	9
	Douglas-fir	5
	grand fir	4

Category	Common name	Plots
Forbs	western swordfern	19
	western brackenfern	10
	common beargrass	8
	white insideout flower	8
	Forb, dicot	6
	fragrant bedstraw	5
	twinflower	5
	British Columbia wildginger	4
	broadleaf starflower	4
	redwood sorrel	4
	Siberian springbeauty	4
	common whipplea	3
	sweet after death	3
	western rattlesnake plantain	3
	white hawkweed	3
Grasses	sedge	6

Table C4. Most common understory vegetation for Douglas County plots.

Category	Common name	Plots
Shrubs	dwarf rose	15
	California blackberry	15
	Cascade barberry	12
	pipsissewa	10
	oceanspray	9
	salal	9
	vine maple	7
	red huckleberry	7
	California hazelnut	7
	honeysuckle	6
	creeping snowberry	6
	Saskatoon serviceberry	5
	thimbleberry	5
	Pacific rhododendron	5
	greenleaf manzanita	4
	Oregon boxleaf	4
	pinemat manzanita	4
	whitebark raspberry	4
	snow raspberry	4
	grouse whortleberry	3
	common snowberry	3
	giant chinquapin	3
	hollyleaved barberry	3
	Himalayan blackberry	3
	Pacific poison oak	3
Trees	Douglas-fir	4

Category	Common name	Plots
Forbs	western swordfern	12
	western brackenfern	11
	woodland strawberry	11
	common whipplea	10
	darkwoods violet	9
	western rattlesnake plantain	9
	broadleaf starflower	8
	Columbian windflower	8
	twinflower	8
	white hawkweed	8
	American trailplant	7
	drops of gold	7
	Forb, dicot	7
	Pacific trillium	6
	purple sweetroot	6
	stickywilly	6
	sidebells wintergreen	5
	starry false lily of the vally	5
	sweet after death	5
	bride's bonnet	4
	pioneer violet	4
	white insideout flower	4
	broadleaf arnica	3
	common beargrass	3
	fragrant bedstraw	3
Grasses	Idaho fescue	7
	long-stolon sedge	3

Table C5. Summary of understory vegetation variety for plots at the county level.

		Minimum	Maximum	Average
Wasco	Forbs	1	17	7.0
	Grasses	0	10	3.2
	Shrubs	1	16	7.6
	Trees	0	3	0.8
Jefferson	Forbs	0	12	5.4
	Grasses	0	8	3.2
	Shrubs	2	13	6.1
	Trees	0	3	1.1

		Minimum	Maximum	Average
Linn	Forbs	2	18	5.8
	Grasses	0	2	0.6
	Shrubs	5	17	8.1
	Trees	0	4	1.2
Douglas	Forbs	0	22	10.7
	Grasses	0	2	0.6
	Shrubs	1	15	9.5
	Trees	0	7	0.6

Appendix D. Detailed Carbon Simulations, Grouped by Age, Region, and Thinning

For the general analysis, it is simpler to separate the plots into groups and to look at general trends. Plots are separated into two regions – eastern Oregon and western Oregon. For each region, stands are grouped into three age classes:

- Young (less than 60 years old in western Oregon, less than 70 years old in eastern Oregon)
- Medium (60-120 years old in western Oregon, 70-120 years old in eastern Oregon)
- Old (greater than 120 years old in western or eastern Oregon).

Stands are further groups into four scenarios for each age group: (1) no treatment, (2) light thinning, (3) break-even (economically) thinning, and (4) heavy thinning, or park-like tree density (in an analysis similar to the example provided in the report).

Table D1. Classification of plots into two regions, six groups, and twenty-four scenarios.

Region	Group	Scenario	Region	Group	Scenario
Western Oregon	Young (< 60 years)	no treatment	Eastern Oregon	Young (< 70 years)	no treatment
		light thinning			light thinning
		break-even thinning			break-even thinning
		heavy thinning			heavy thinning
	Medium (60-120 years)	no treatment		Medium (70-120 years)	no treatment
		light thinning			light thinning
		break-even thinning			break-even thinning
		heavy thinning			heavy thinning
	Old (> 120 years)	no treatment		Old (> 120 years)	no treatment
		light thinning			light thinning
		break-even thinning			break-even thinning
		heavy thinning			heavy thinning

From the analysis of these particular plots, several patterns emerge:

- The relative amount of carbon and total volume after 50 years is highest in the "No Treatment" scenario for each of the six groups.
- The relative amount of carbon and total volume after 50 years is lowest in the "Heavy Thinning" scenario for each of the six groups including considerations of downstream wood utilization in forest products and bioenergy.
- The average relative amount of carbon and total volume is higher in all scenarios after 50 years for the "Light Thinning" scenario, when compared to the "Break-even Thinning" and the "Heavy Thinning" scenario.
- Younger stands – Tended to show the highest rate of carbon accumulation, but not necessarily the greatest absolute accumulation of carbon.
- Older stands – These stands tended to be thinned heavily for dense stands, which tended to have significant understory that led to fuel ladders. Largest trees were preserved, and the approach was to develop a "park-like" scenario with most fuels in the understory removed (all stems <12" diameter and a relatively low residual density of stems 12-20" diameter).

- Eastern Oregon vs. western Oregon – The plots in western Oregon tended to have higher amounts of initial carbon, and higher rates of carbon and volume accumulation. This relationship was observed for all scenarios.

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This set of plots does not necessarily indicate carbon levels at a regional level, and a spatial analysis should be conducted before making broader conclusions based on these simulations.

Guide to Reading Graph Legends

Average Live Carbon. Simulates aboveground carbon store of all live standing trees and shrubs/herbs. There is one solid line that represents average simulated carbon for all plots in the given scenario (MgC/ha).

Average Total Carbon. Simulates sum of forest carbon pools estimated by FVS and allometric equations. There is one solid line that represents average simulated carbon for all plots in the given scenario (MgC/ha).

Average Carbon Offset from not Burning Coal. When burning biomass for energy instead of coal, the carbon emissions for biomass replaces the carbon emissions for coal. This bar includes the estimated "avoided" carbon emissions for each thinning scenario when burning biomass for energy instead of coal (MgC/ha).

Average Carbon stored in Wood Products- This is the estimated carbon transferred and stored in harvested wood products for each thinning scenario (MgC/ha).

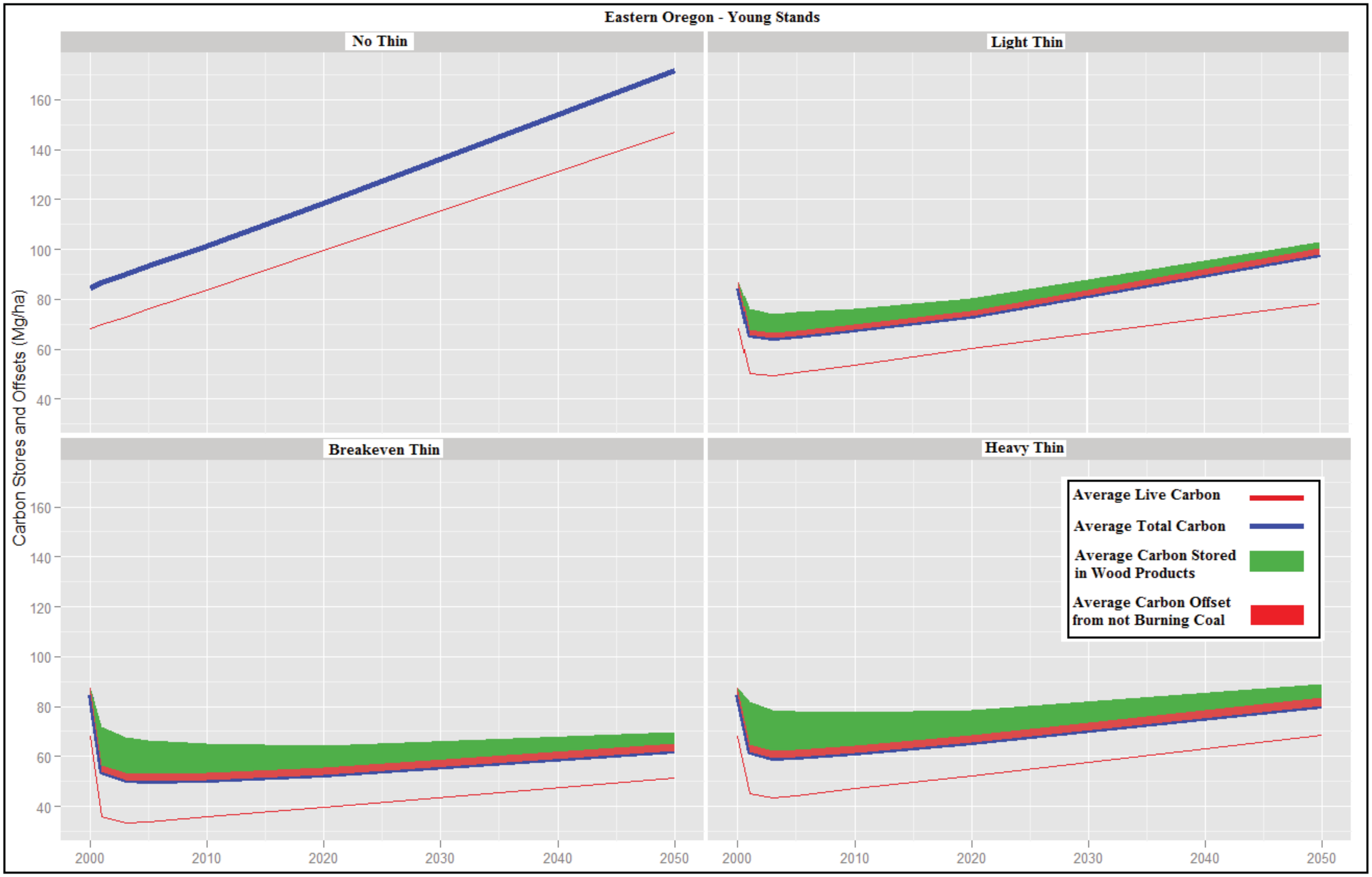


Figure D1. Eastern Oregon – young stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

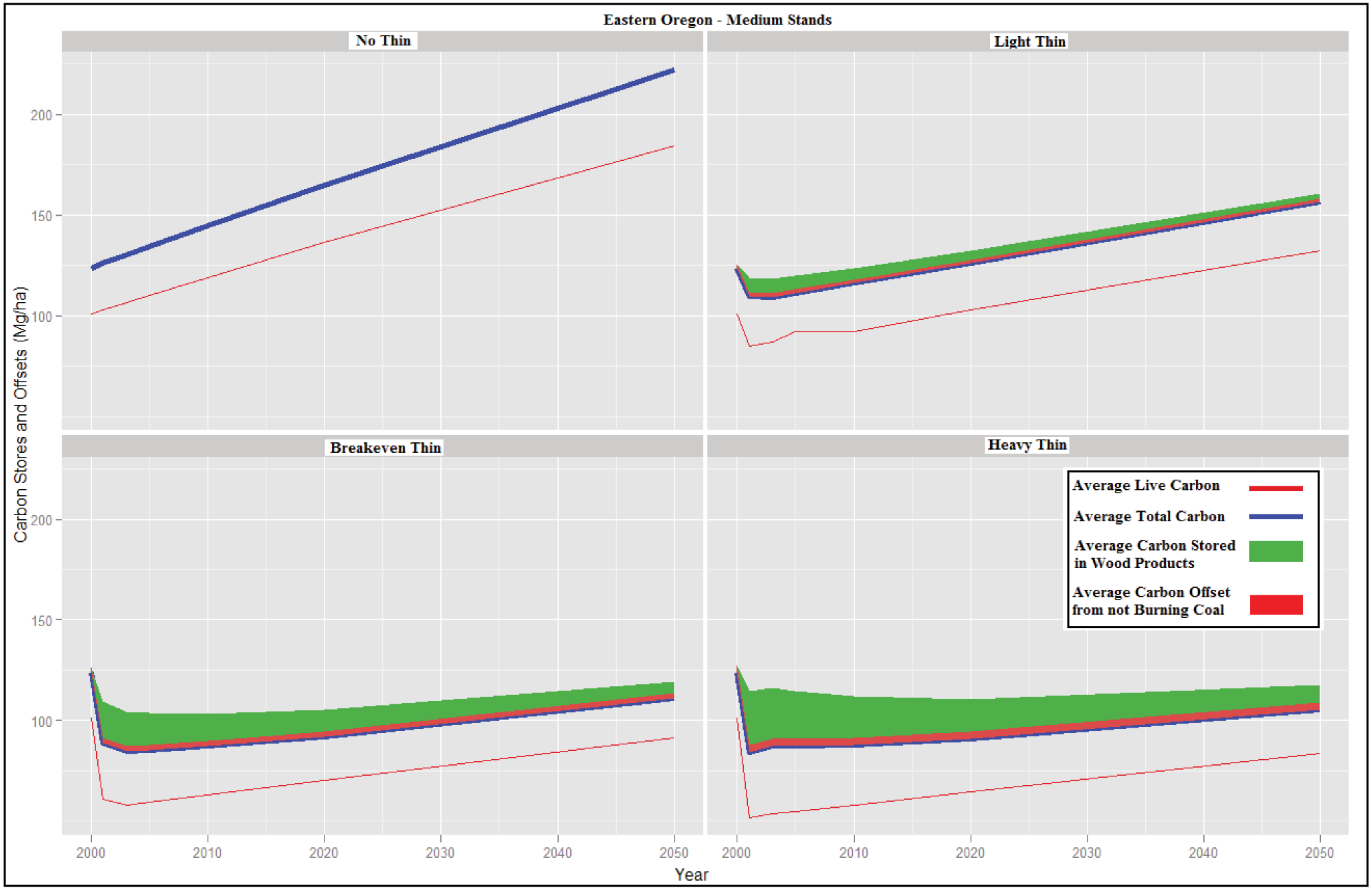


Figure D2. Eastern Oregon – medium stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

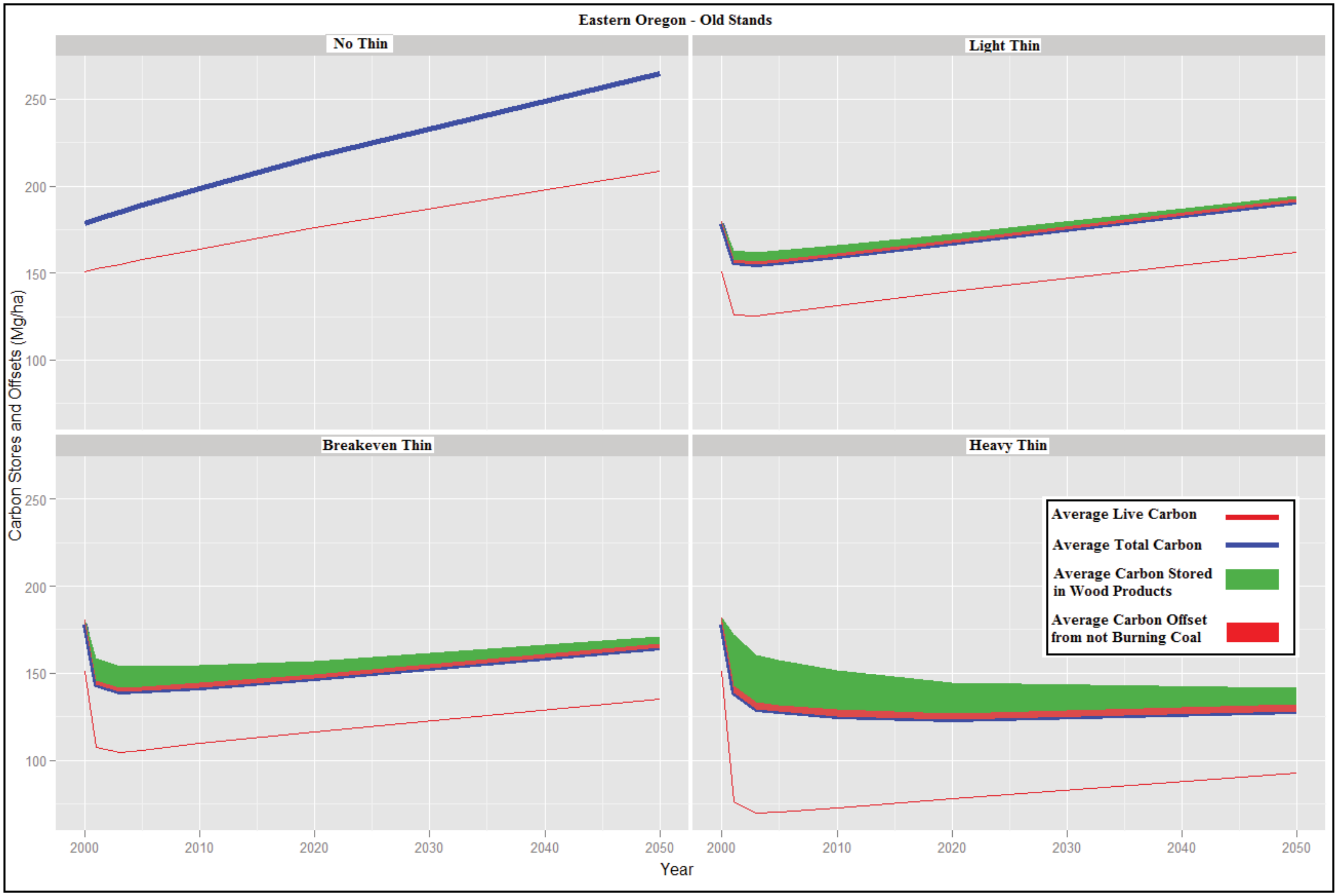


Figure D3. Eastern Oregon – old stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

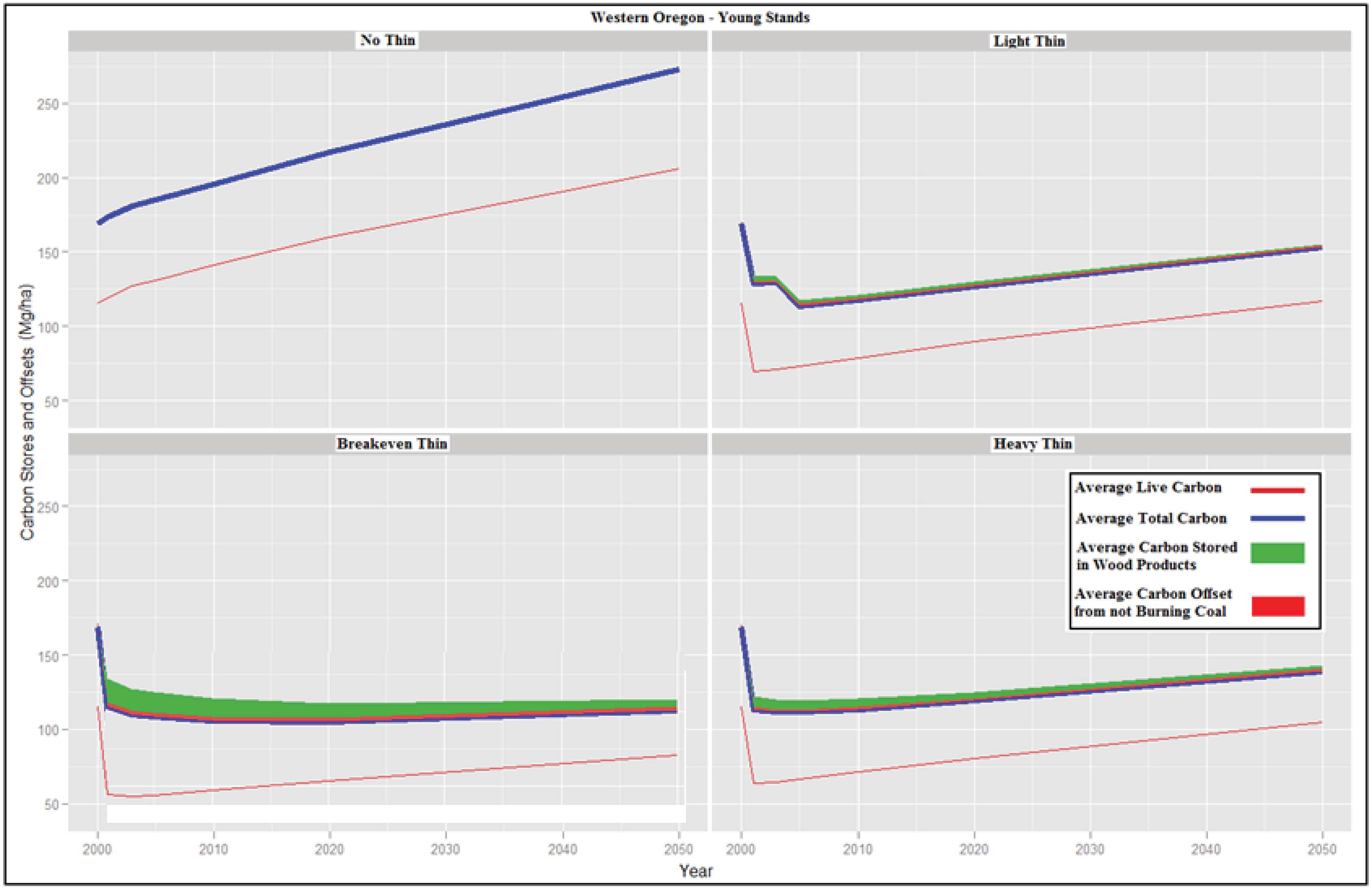


Figure D4. Western Oregon – young stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

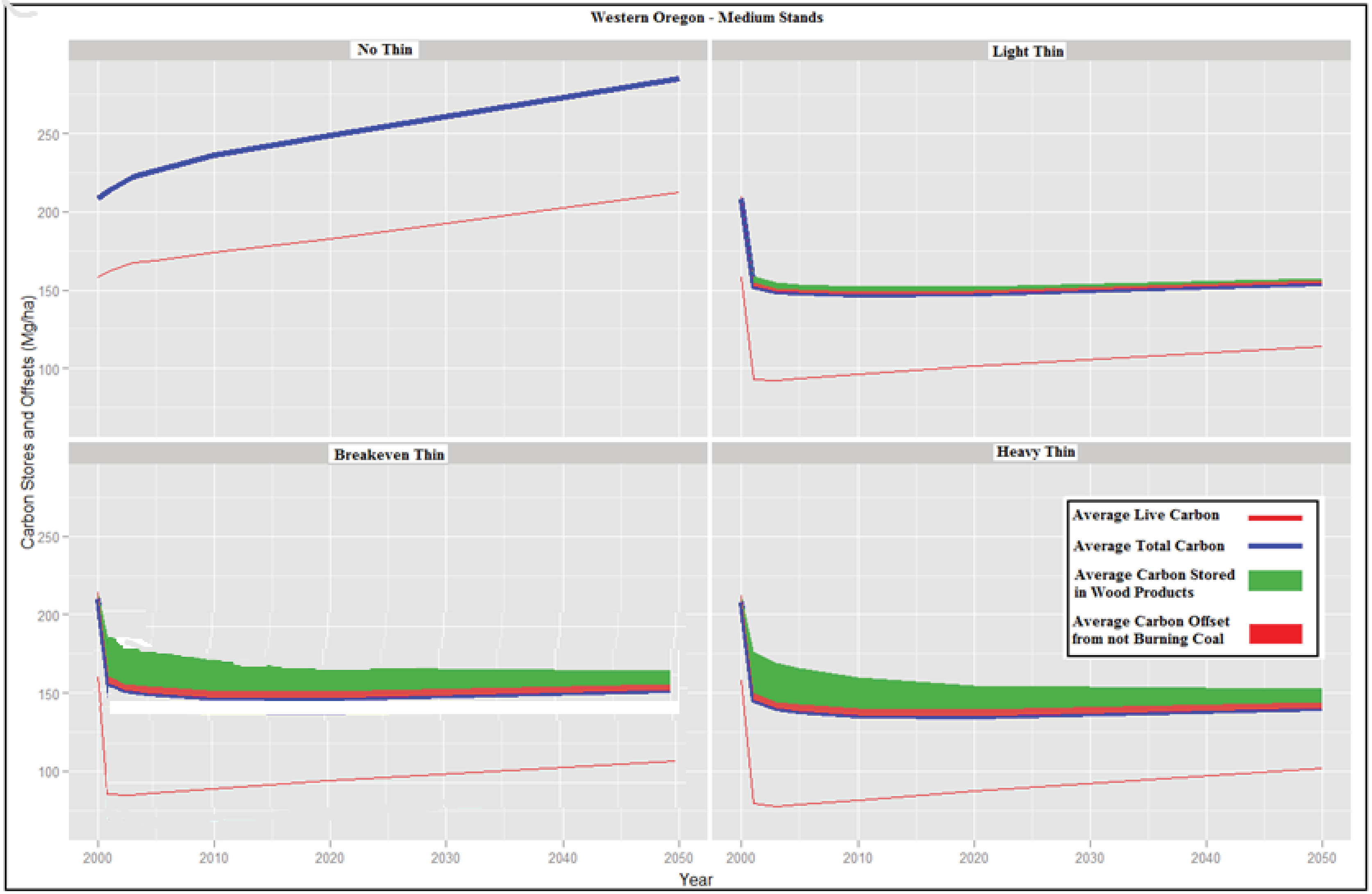


Figure D5. Western Oregon – medium stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

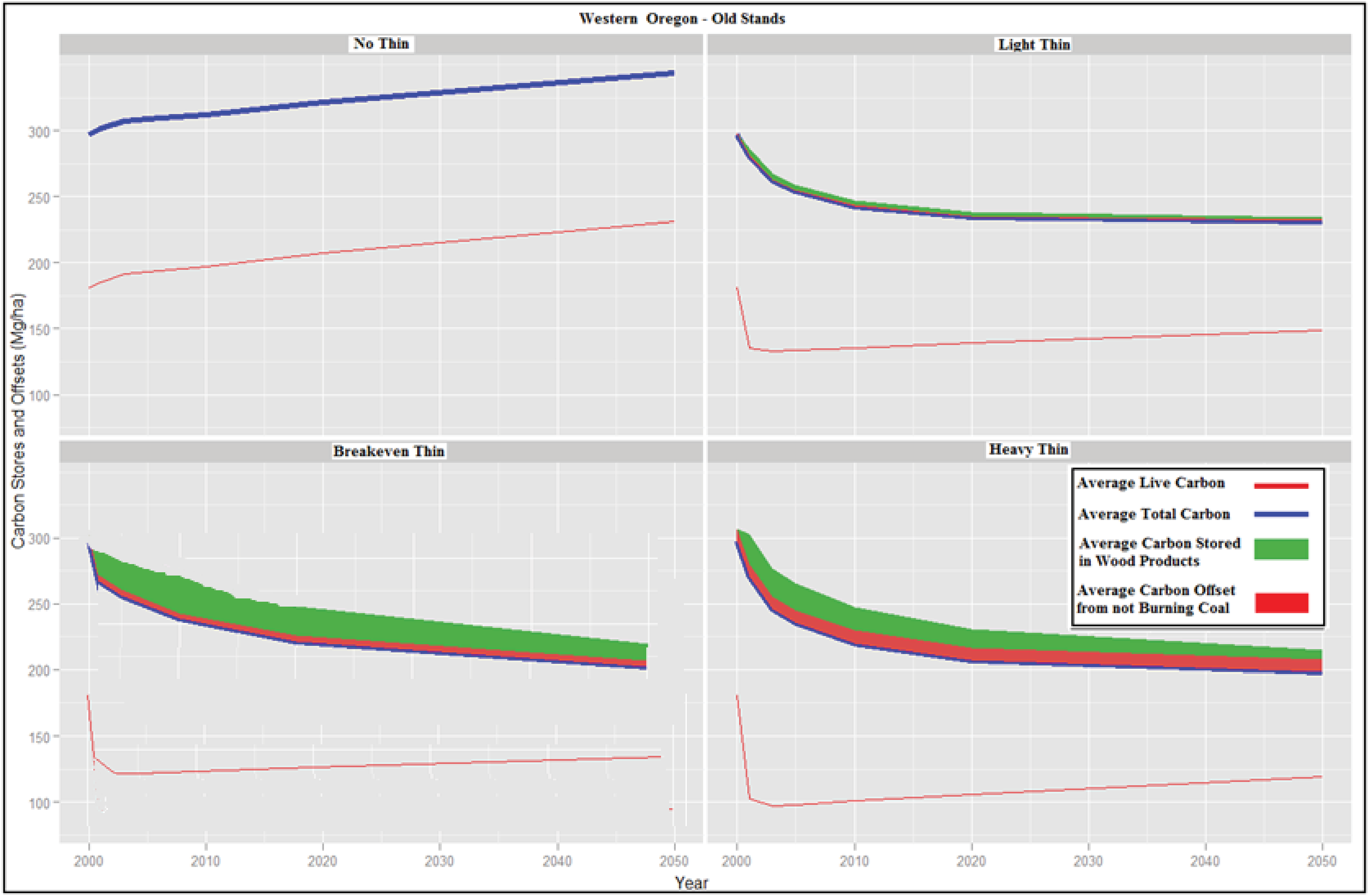


Figure D6. Western Oregon – old stands. Simulation of carbon pools for the forest stand over a 50 year period. Biomass for energy is not included in wood product sequestration – it is assumed utilized within the first year.

Appendix E. Carbon Accounting Methodology Using FVS and other Tools

There are several different methods and tools available to estimate tree-level and/or plot-level carbon. In this analysis, the primary source for biomass and carbon estimates is the FVS-FFE extension. Two components (1-ft stump and bole bark) were estimated manually. All plot carbon stores are included in estimates except for soil carbon. In all components, carbon weight is estimated as 50% of bone-dry biomass weight (see page 325 Penman et al. 2003) except for litter and duff, which is estimated as 37% carbon (Smith and Heath 2002).

Most of the detailed information about FVS calculations was taken from FVS user manuals or from personal communication with developers.

The carbon pools for each plot were estimated as follows:

Aboveground Standing Live (FVS and Allometric Estimates):

Bole Biomass (FVS):

- Bole volume (green) is estimated using equations from the National Volume Estimator Library, based on region, species, diameter at breast height (DBH), height, and other tree-level measurements.
- Specific gravity for the tree species/region is used (Reinhardt et al. 2009) to estimate bole biomass from green volume by the equation Bole Biomass = Green Volume * Specific gravity (Forest Products Laboratory 1999).
- Defects can be accounted for using total volume estimates in FVS, and 15% defect is included for wood products estimates, as in a previous study (Adams and Latta 2003).
- Bole bark is not included in the FVS estimate, but was included manually (see below).
- Stump biomass (from ground to 1 foot height) were not included in FVS-FFE estimates, but they were included manually (see below for stump calculations).

Bole Bark Biomass (Allometric Estimate):

- Estimate from regional biomass estimates (Gholz et al. 1979). Estimates are based on species and DBH.

Stump Biomass (Allometric Estimate):

- The stump not accounted for in the FFE-FVS measurement is 1 ft high. The part of the stump above 0.5 ft is considered part of the bole when harvested, and the part of the stump below 0.5 ft is assumed aboveground biomass left behind if the tree is cut.
- Diameter estimates for stumps are taken from allometric equations (Wensel and Olson 1995). Function of species, DBH, height of DBH measurement.
- The assumed cut height for stumps was 0.5 ft. Stump volume was estimated by dividing the 1-ft stump into 2 frustums, each 0.5 ft high.
- Density is assumed to be a constant (not height dependent) for each species (Bouffier et al. 2003; Megraw 1985). Biomass is calculated as Density*Volume. Carbon is assumed to be 50% of bone-dry weight.

Crown Biomass (FVS):

- Equations based on tree parameters (species, DBH, height, relative dominance in the plot). (Brown and Johnston 1976).
- Crown biomass estimates also based on crown ratio, tree height in stand.
  - If ≥ 60th percentile, then assumed dominant/co-dominant.
  - If < 60th percentile, then assumed suppressed/intermediate.
- The crown is divided into dead/live and material size by diameter (foliage, <0.25", 0.25"-1", 1"-3", >3").

Aboveground Standing Dead (FVS):

- This component was modeled based on several factors (details in Rebain 2008). Parameters modeled include snag fall (and associated height loss) and decay rates based on several parameters, including regional temperature, moisture class by plant association, years before hard snags become soft snags, soil moisture, soil depth, and soil position.
- All plots have dead trees that are measured and included. Snags are classified as recent mortality or not recent mortality.
- The 0.5 ft stumps left after thinning are also included. A study of decomposition rates of stumps in an old-growth stand of Douglas-fir and western hemlock (Janisch et al. 2005) suggested that log decay rates can be substituted for stump decay rates. In this analysis, the FVS decay rates of course woody debris are applied to stumps. Annual decay rates are 2.5% for stumps less than 3" diameter, and 1.25% for stumps greater than 3" diameter.

Belowground Live (FVS):

- Includes all coarse roots >2mm (0.079 in) in diameter.
- Fine roots are assumed to be part of soil carbon (e.g., Jenkins 2003), and are not estimated.

Belowground Dead (FVS):

- Includes coarse roots >2mm (0.0079 in) in diameter. Smaller roots are not estimated.
- The default root decay rate of 0.0425 is used (Ludovci et al. 2002).

Forest Floor (FVS):

- Includes duff and leaf litter.
- Annual litterfall uses estimates based on Keane et al. 1989, and is a function of species, foliage weight, and leaf lifespan.

Downed Dead Wood (FVS):

- For this pool, the default value is used initially (Reinhardt et al. 2009).

Shrubs and Herbs (FVS):

- Does not dynamically simulate weight of shrubs and herbs, and is assumed roughly constant in a stand, given the understory vegetation associated with a plot.
- Biomass estimates are based on the First Order Fire Effects Model (FOFEM) (Reinhardt et al. 1997).

Conversion from bone-dry biomass weight to carbon
The conversion from bone-dry biomass to carbon is simple. Carbon content for all biomass is assumed to be 50% of bone-dry biomass except for litter and duff which is estimated as 37% of bone-dry biomass.

Moisture Content

All moisture content estimates are made using wet basis. This basis estimates water content as a fraction of green weight.

$$MC\ \%_{wet} = \frac{green\ weight - dry\ weight}{green\ weight} \text{ x } 100.$$

Appendix F. Summary Tables of Carbon Stores, Fluxes, Relative Carbon and Fuel Reduction Measurement.

Table F1. Estimated mean carbon stores with associated standard error for each region/age category. Initial growing stock volume is total volume, not merchantable volume. Carbon pools for each plot are separated into three categories: (1) Live (Aboveground Standing, Belowground Live, Shrubs and Herbs); (2) Dead (Belowground Dead, Standing Dead, Downed Dead Wood); and (3) Forest Floor. Data is presented as: carbon mean [Mg/hectare] (carbon standard error) [Mg/hectare].

Plot Description		Number of plots	Initial Growing stock volume (m^3/ha)	Estimated Carbon Pools on site (Mg C/ha)		
Region	Group (by Plot Age)			Live	Dead	Total
Eastern Oregon	Young (21-69 years)	18	151.5 (21.2)	68.3 (8.5)	16.5 (2.4)	84.8 (10.4)
	Medium (71-118 years)	26	229.8 (21.4)	101.0 (8.9)	22.8 (2.9)	123.9 (10.7)
	Old (123-269 years)	20	363.9 (33.6)	150.9 (11.4)	27.5 (4.8)	178.4 (12.7)
Western Oregon	Young (10-60 years)	19	248.3 (46.5)	115.1 (21.7)	54.0 (7.5)	169.1 (25.0)
	Medium (64-105 years)	12	227.5 (86.0)	158.1 (28.7)	50.4 (8.0)	208.6 (28.4)
	Old (122-220 years)	5	362.3 (15.7)	180.9 (17.0)	116.4 (19.3)	297.3 (29.3)

Table F2. Carbon pool estimate relative to initial carbon store where 100% represents the initial mean carbon store of a region/plot age combination before thinning.

Carbon Pool Estimate Relative to Initial Carbon Pool																				
Plot Description			Live						Dead						Total					
Region	Plot Age	Thinning Scenario	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50
Eastern Oregon	Young	No Thinning	102%	107%	111%	123%	146%	215%	101%	103%	105%	108%	115%	149%	102%	106%	110%	120%	140%	203%
		Light Thinning	74%	72%	74%	79%	88%	115%	94%	92%	90%	85%	80%	78%	78%	76%	77%	80%	86%	92%
		"Break-even" Thinning	53%	48%	50%	52%	58%	75%	107%	102%	98%	90%	78%	64%	63%	60%	59%	60%	62%	73%
		Heavy Thinning	66%	63%	65%	69%	76%	100%	100%	96%	93%	87%	80%	72%	73%	70%	70%	72%	77%	95%
	Medium	No Thinning	102%	105%	109%	118%	135%	183%	101%	104%	106%	112%	124%	165%	102%	105%	109%	117%	133%	179%
		Light Thinning	84%	86%	91%	91%	102%	131%	110%	108%	107%	104%	102%	104%	89%	88%	90%	94%	102%	126%
		"Break-even" Thinning	60%	57%	59%	62%	70%	90%	122%	117%	113%	106%	97%	85%	72%	68%	69%	70%	74%	90%
		Heavy Thinning	51%	53%	55%	58%	64%	83%	149%	146%	141%	131%	115%	95%	68%	70%	70%	71%	73%	85%
	Old	No Thinning	101%	103%	104%	109%	117%	138%	103%	109%	114%	126%	148%	205%	101%	104%	106%	111%	122%	148%
		Light Thinning	84%	83%	84%	87%	93%	107%	108%	106%	105%	103%	101%	106%	87%	87%	87%	90%	94%	107%
		"Break-even" Thinning	71%	69%	70%	73%	77%	90%	130%	127%	124%	118%	111%	106%	80%	78%	78%	79%	82%	92%
		Heavy Thinning	50%	46%	47%	48%	51%	62%	229%	218%	210%	191%	165%	128%	78%	72%	72%	70%	69%	72%
Western Oregon	Young	No Thinning	104%	110%	114%	122%	139%	179%	100%	100%	100%	102%	107%	124%	103%	107%	110%	116%	128%	161%
		Light Thinning	60%	62%	64%	69%	78%	102%	93%	88%	85%	71%	68%	67%	76%	77%	67%	69%	75%	90%
		"Break-even" Thinning	38%	37%	38%	41%	46%	61%	110%	103%	97%	87%	74%	56%	61%	58%	57%	56%	55%	59%
		Heavy Thinning	55%	56%	58%	62%	70%	91%	92%	87%	84%	78%	71%	64%	67%	66%	66%	67%	70%	82%
	Medium	No Thinning	102%	106%	107%	110%	116%	134%	103%	109%	114%	123%	131%	144%	103%	106%	109%	113%	119%	137%
		Light Thinning	59%	58%	59%	61%	64%	72%	118%	113%	109%	101%	93%	80%	73%	72%	71%	71%	71%	74%
		"Break-even" Thinning	51%	50%	52%	55%	59%	69%	126%	122%	114%	105%	96%	79%	72%	70%	70%	72%	74%	77%
		Heavy Thinning	50%	49%	50%	52%	55%	64%	132%	124%	118%	108%	95%	76%	70%	67%	66%	65%	65%	67%
	Old	No Thinning	103%	106%	107%	109%	115%	128%	100%	99%	99%	99%	98%	97%	102%	103%	104%	105%	108%	116%
		Light Thinning	75%	73%	74%	75%	77%	82%	124%	111%	103%	92%	82%	71%	94%	88%	85%	82%	79%	78%
		"Break-even" Thinning	66%	68%	70%	71%	72%	76%	135%	122%	112%	97%	84%	70%	93%	85%	81%	74%	74%	75%
		Heavy Thinning	57%	54%	54%	56%	59%	66%	144%	128%	118%	103%	87%	68%	91%	83%	79%	74%	70%	67%

Table F3. Torching Index and Crowning Index estimates.

Torching Index is the wind speed at which crown fire is expected to initiate (based on Rothermel (1972) surface fire model and Van Wagner (1977) crown fire initiation criteria. Crowning Index is the wind speed at which active crowning fires are possible (based on Rothermel (1991) crown fire spread rate model and Van Wagner (1977) criterion for active crown fire spread). Wind speed refers to speed of wind measured 20 ft above the canopy. Lower values indicate higher susceptibility. Data is presented as: crowning/torching index [mi/hr] (standard deviation) [mi/hr] for select years. Red indicates that plots would benefit from a thinning using the criteria in this study. Orange indicates the average was still below criteria following the thinning, and green indicates that the average index for plots was above the minimum criteria used in this study.

Plot Description			Crowning Index (mi/hr)							Torching Index (mi/hr)						
Region	Plot Age	Thinning Scenario	Year 0 After Thinning	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50	Year 0 After Thinning	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50
Eastern Oregon	Young	No Thinning	39.3 (24.7)	39.3 (24.7)	38.7 (23.8)	37 (20.9)	30.5 (16.5)	28.7 (15.3)	25.1 (12.0)	52.2 (55.6)	48.3 (47.2)	49 (46.4)	51.3 (44.1)	54 (37.9)	84.4 (82.2)	117 (96.4)
		Light Thinning	78 (45.6)	78 (45.6)	89.1 (54.9)	88.6 (55.0)	75.3 (46.6)	72 (44.7)	65.7 (45.7)	38.6 (23.2)	38.5 (23.1)	43.8 (24.0)	45.6 (25.2)	44 (22.4)	50.7 (26.6)	67.4 (44.8)
		"Break-even" Thinning	89 (23.0)	89 (23.0)	102 (32.1)	103.4 (35.4)	87.6 (30.3)	83.6 (30.7)	76.4 (33.0)	29.4 (14.0)	29.4 (14.0)	33.5 (13.6)	35.9 (13.1)	37.2 (20.1)	39.6 (22.4)	47.1 (15.5)
		Heavy Thinning	66.4 (18.1)	66.4 (18.1)	77.7 (25.0)	78.2 (25.0)	65.9 (21.0)	63.2 (21.1)	57.5 (22.8)	35.4 (16.9)	35.4 (16.8)	45.1 (23.3)	48 (25.3)	44.4 (25.0)	44.6 (19.1)	55.1 (22.0)
	Medium	No Thinning	35.1 (13.1)	35.1 (13.1)	35.1 (13.5)	34.4 (13.1)	29.7 (11.1)	27.3 (9.8)	24.3 (9.8)	155.9 (184.4)	143.3 (180.7)	143.8 (183.7)	137.3 (145.9)	122.6 (133.3)	120.6 (127.2)	118 (108.6)
		Light Thinning	45.7 (19.6)	45.7 (19.6)	50.7 (23.3)	49.9 (22.9)	43.6 (20.5)	41.1 (19.5)	37.6 (19.6)	113.3 (197.3)	107.5 (176.2)	114.5 (174.4)	118.1 (179.9)	100.2 (126.0)	115.6 (125.8)	140.4 (145.3)
		"Break-even" Thinning	65.9 (26.4)	65.9 (26.4)	72.1 (30.2)	71.7 (30.4)	63.1 (27.1)	60.6 (28.0)	55 (28.4)	40.6 (19.8)	41 (20.4)	45.9 (22.2)	49 (22.5)	49.9 (25.5)	60.2 (36.8)	77.2 (64.9)
		Heavy Thinning	71.6 (34.5)	71.6 (34.5)	74.9 (34.7)	73.6 (33.2)	64.3 (30.8)	60 (27.9)	53.8 (24.9)	43.9 (26.9)	44.3 (27.6)	50.1 (31.2)	54.7 (34.1)	54.3 (33.6)	66.3 (54.7)	85.2 (73.4)
	Old	No Thinning	39.9 (17.3)	39.9 (17.3)	40 (17.0)	39.5 (16.7)	37.4 (17.1)	34.9 (15.5)	33.7 (15.4)	155.4 (163.7)	154.4 (165.1)	162.8 (173.0)	171.1 (181.3)	158.3 (156.8)	146 (155.7)	98.1 (95.7)
		Light Thinning	59.1 (25.9)	59.1 (25.9)	69.2 (31.1)	69.2 (31.8)	66.8 (34.0)	66 (35.8)	63.4 (37.0)	98.8 (110.9)	101.8 (114.5)	106.4 (115.5)	109.7 (121.4)	116.9 (139.3)	132.8 (155.9)	76.5 (66.2)
		"Break-even" Thinning	68.1 (26.1)	68.1 (26.1)	79.4 (32.3)	79.1 (32.6)	76.1 (35.1)	74.7 (36.7)	71.1 (38.2)	81.5 (87.8)	80.7 (88.0)	95.3 (93.9)	98.7 (99.1)	105.7 (113.7)	122.2 (132.8)	149.3 (158.8)
		Heavy Thinning	128.8 (93.1)	128.8 (93.1)	130.4 (89.5)	127.2 (87.2)	115.8 (82.3)	104.2 (72.6)	100.8 (79.4)	48.7 (29.7)	49 (29.6)	56.3 (35.6)	59 (35.0)	61.8 (31.1)	68.7 (31.0)	89.5 (42.2)
Western Oregon	Young	No Thinning	36.1 (40.3)	36.1 (40.3)	37 (43.6)	37.1 (44.6)	27.3 (29.9)	24.1 (20.6)	21.6 (14.2)	92.2 (98.4)	87.5 (106.3)	84 (128.6)	69 (75.8)	58.9 (72.9)	90.2 (148.8)	141.5 (218.3)
		Light Thinning	56.5 (58.8)	56.5 (58.8)	56.3 (58.6)	56.2 (58.3)	46.3 (48.9)	45.9 (48.3)	45.4 (47.0)	98.7 (81.6)	96.5 (80.6)	95 (79.2)	92 (75.7)	66.8 (52.9)	73.9 (53.9)	92.3 (58.4)
		"Break-even" Thinning	82.6 (96.2)	82.6 (96.2)	82.2 (95.6)	81.8 (95.1)	66.5 (76.9)	65.2 (75.3)	63.7 (72.5)	75.6 (70.0)	75.4 (68.6)	74.7 (66.2)	76.1 (65.0)	54.1 (44.4)	60.3 (41.3)	78.2 (43.3)
		Heavy Thinning	59.3 (57.0)	59.3 (57.0)	58.9 (56.9)	58.9 (56.6)	48.5 (47.4)	48.1 (46.8)	47.8 (45.5)	88.7 (78.8)	87.2 (77.1)	87.2 (74.7)	85.9 (69.7)	62.7 (47.3)	69.9 (48.2)	90.5 (53.2)
	Medium	No Thinning	20.3 (19.0)	20.3 (19.0)	20.2 (18.3)	20.2 (17.8)	15.3 (14.2)	15.1 (13.5)	15.5 (12.6)	56.7 (175.0)	61.8 (199.9)	73.6 (247.5)	72.6 (247.7)	67.5 (248.7)	69.3 (248.3)	85.2 (253.6)
		Light Thinning	26.4 (32.4)	26.4 (32.4)	27.5 (33.3)	27.4 (33.0)	21.3 (26.6)	20.9 (25.5)	20.8 (23.7)	64.1 (163.1)	64.1 (163.1)	71 (184.3)	84.1 (231.2)	76.7 (239.9)	79.7 (240.1)	93 (237.0)
		"Break-even" Thinning	38.7 (60.0)	38.7 (60.0)	40.3 (60.3)	40.6 (59.9)	31.8 (47.5)	31.2 (45.3)	30.8 (41.7)	61.8 (168.5)	62.3 (168.1)	65.1 (167.9)	65.8 (169.6)	41.4 (109.6)	46.1 (114.4)	60.3 (120.9)
		Heavy Thinning	30.7 (33.0)	30.7 (33.0)	32 (34.1)	32.1 (33.7)	25.4 (27.0)	25.8 (25.7)	27.2 (23.0)	59.5 (161.5)	59.5 (161.6)	68.6 (183.1)	80.1 (229.8)	75.2 (239.5)	78.7 (239.8)	88 (237.6)
	Old	No Thinning	14.7 (4.8)	14.7 (4.8)	15.8 (5.9)	15.5 (5.1)	12.5 (5.3)	13.7 (6.4)	15.2 (5.5)	7.9 (14.0)	8.4 (14.8)	12.3 (22.4)	14.9 (24.7)	9.2 (16.9)	14.4 (21.2)	36 (25.8)
		Light Thinning	26.9 (20.0)	26.9 (20.0)	28.8 (21.8)	28.3 (21.9)	22.5 (17.6)	23.1 (17.1)	23.3 (15.7)	31 (41.1)	33.5 (43.5)	51.4 (70.4)	55.8 (71.0)	35.7 (46.1)	45 (49.4)	63.1 (55.4)
		"Break-even" Thinning	42.6 (31.1)	42.6 (31.1)	46.9 (36.3)	46.4 (36.2)	36.7 (28.8)	37 (26.8)	36.1 (23.7)	25.3 (39.0)	26.4 (41.2)	30 (46.0)	31.6 (48.8)	18.4 (30.3)	26.3 (35.3)	49.1 (41.5)
		Heavy Thinning	33 (18.7)	33 (18.7)	35.1 (20.6)	34.7 (20.3)	27.8 (16.2)	29 (15.8)	28.8 (13.2)	27.3 (43.0)	29 (45.8)	44.3 (73.9)	46.3 (75.6)	28.8 (48.9)	38.6 (51.0)	61.7 (56.4)

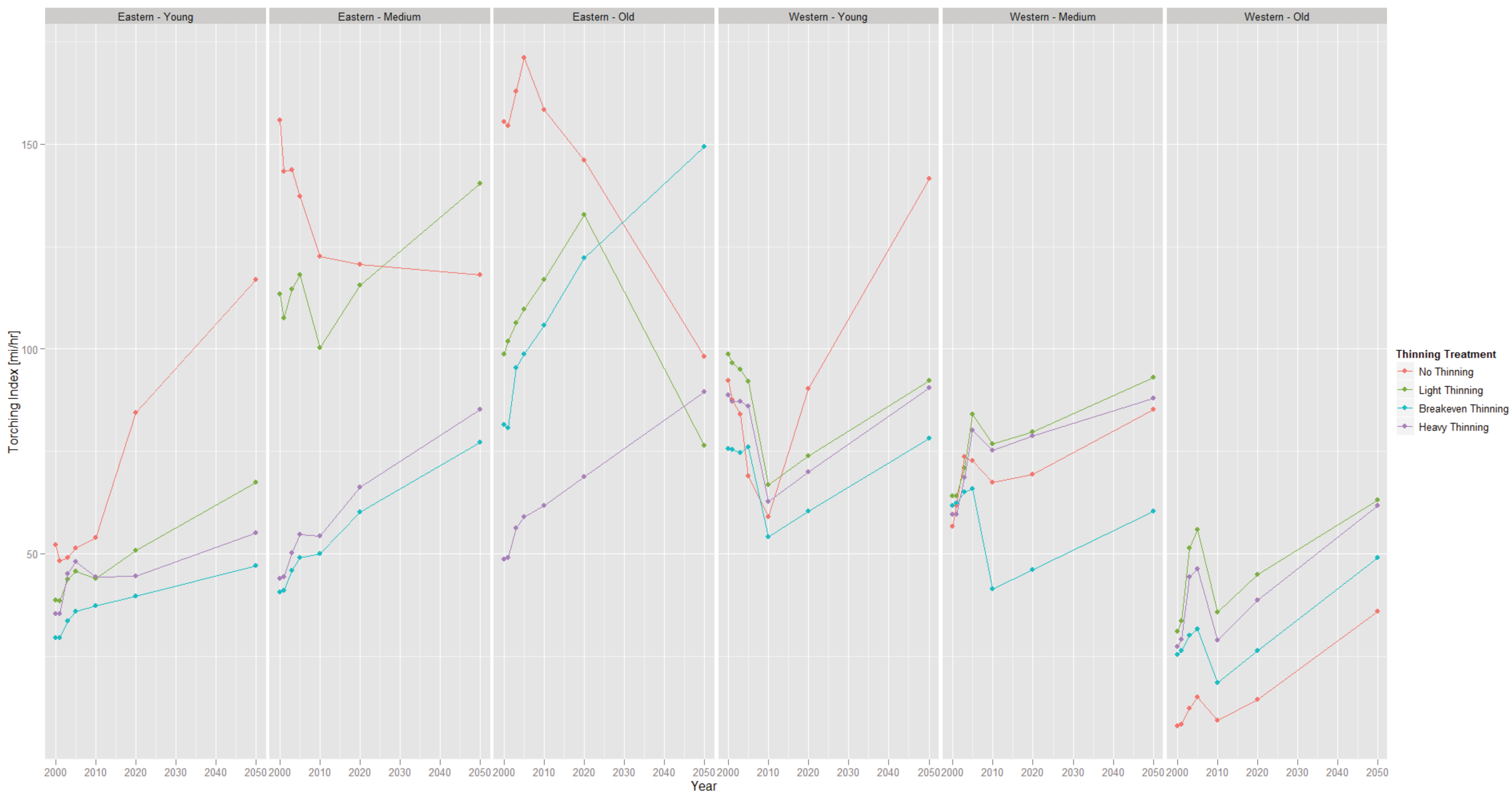


Figure F1. Torching Index (mi/hr) over a 50 year period – comparison is for different treatments for region/age combinations. This is a graphical representation of the means (averages) from Table F3, and does not include variance, which is relatively high compared to the mean.

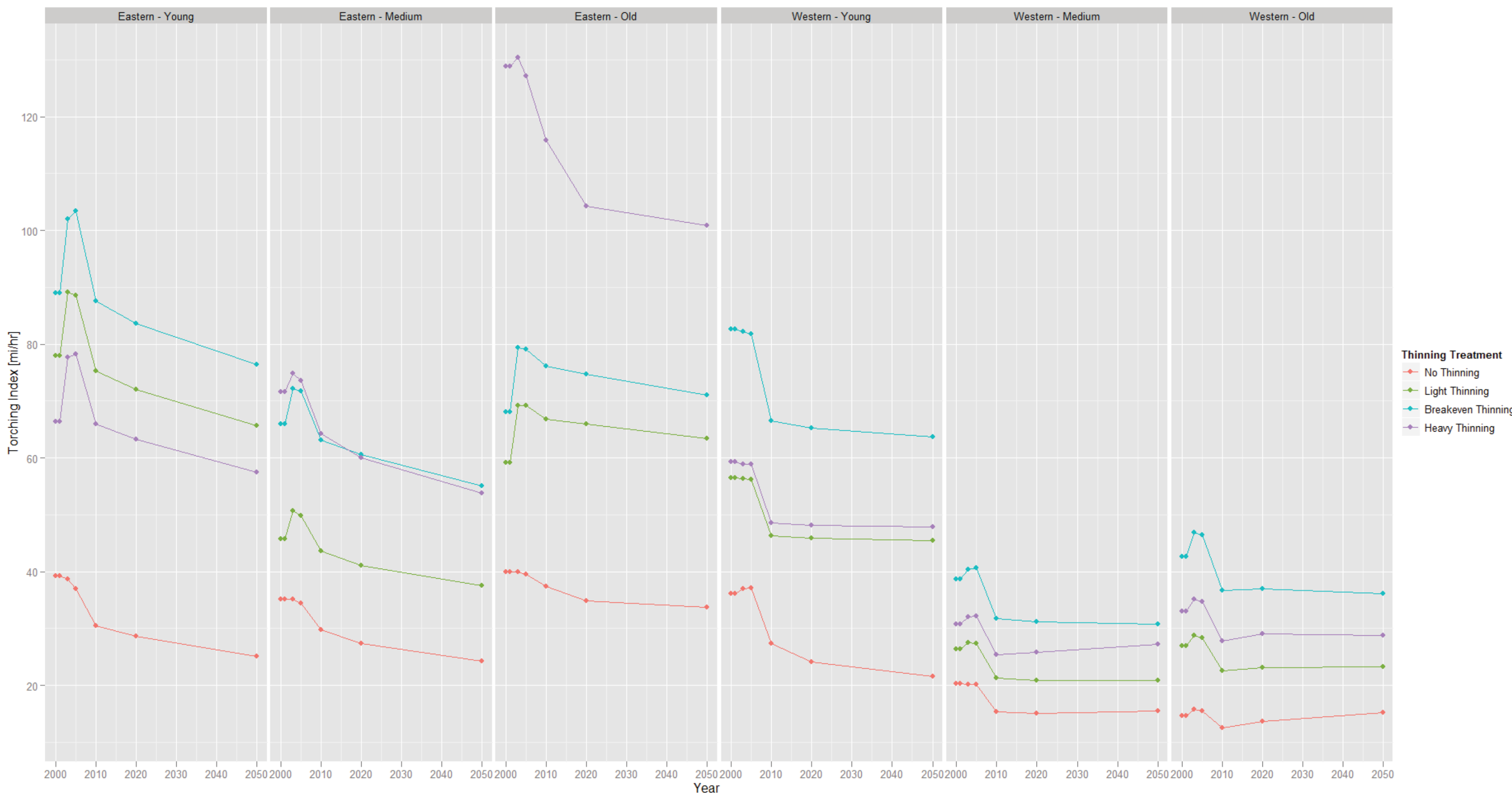


Figure F2. Crowning Index (mi/hr) over a 50 year period – comparison is for different treatments for region/age combinations. This is a graphical representation of the means (averages) from Table F3, and does not include variance, which is high relative to the mean.

Table F4. Number of plots within each region and age group with the greatest amount of Live, Dead, and Total Carbon stores for each thinning scenario vs. no thinning scenario. As seen in this table, carbon stores in a plot following a thinning are always lower for every plot used in this analysis.

Number of plots for each scenario with the largest carbon store																				
Plot Description			Live						Dead						Total					
Region	Plot Age	Thinning Scenario	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50	Year 1	Year 3	Year 5	Year 10	Year 20	Year 50
Eastern Oregon	Young	No Thinning	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18	18
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Medium	No Thinning	26	26	26	26	26	26	26	26	26	26	26	26	26	26	26	26	26	26
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Old	No Thinning	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20	20
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Western Oregon	Young	No Thinning	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19	19
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Medium	No Thinning	12	12	12	12	12	12	12	12	12	12	12	12	12	12	12	12	12	12
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Old	No Thinning	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
		Light Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		"Break-even" Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Heavy Thinning	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix G. Conversion Units and Definitions.

Conversion factors from metric to imperial units.

1 hectare (ha)	= 2.471 acre (ac)
1 meter (m)	= 3.281 feet (ft)
1 square meter (m^2)	= 10.764 square feet (ft^2)
1 cubic meter (m^3)	= 35.315 cubic feet (ft^3)
1 megagram (Mg)	= 1000 kilograms (kg) = 1 metric tonne = 1.102 short tons
1 short ton	= 2000 pounds (lbs)
1 kilowatt-hr (kWh)	= 3413 British Thermal Units (BTU)

Definitions from IPCC FAR used in this report (IPCC 2007):

- Reservoir – "a component of the climate system other than the atmosphere which has the capacity to store, accumulate or release… greenhouse gas..."
- Sink – "any process, activity or mechanism that removes a greenhouse gas … from the atmosphere."
- Source – "any process, activity or mechanism that releases a greenhouse gas… into the atmosphere."

California

13.2% emissions rate reduction

126 lbs CO2 / MWh reduction

DRAFT RULE REDUCTION: 23.1% (161 lbs CO2 / MWh)

Editor's note: The following summary represents state and utility stances after the Supreme Court stayed the Clean Power Plan in February 2016.

California was the first in the nation to release a **draft proposal** for complying with U.S. EPA's Clean Power Plan mandate.

The Golden State in a draft released in early August said it will rely on its cap-and-trade program for carbon emissions and proposed amendments to extend that system in order to meet its targets under EPA's rule.

"The Cap-and-Trade Program establishes a declining limit on major sources of GHG emissions, and it creates a powerful economic incentive for major investment in cleaner, more efficient technologies," the California Air Resources Board said in the proposal. Other state rules will provide assistance. Those include California's energy efficiency standards and a mandated level of renewable power for electricity generation.

The ARB is choosing to comply with the CPP via the "state measures" option, which gives states the choice to develop and use their own rules to achieve required cuts to greenhouse gas emissions.

California said it's proceeding despite the Supreme Court's stay as "an insurance policy" and to make sure the CPP requirements sync up with cap and trade in the later years of the program. It also wants to provide "a proof of concept for other states, to demonstrate that this is a program that can be adapted to each state and that can be set up in a way that we can form a regional association," said ARB spokesman Stanley Young.

California under the CPP must cut its emissions rate 13.2 percent below the 2012 level by 2030.

It looks likely to hit that number. As the only state with an economywide carbon cap, California is already on track to reduce its greenhouse gases to 1990 levels by 2020, and is writing regulations to reach 40 percent below that by 2030.

Even in a high-emissions scenario, which could come about through higher-than-expected electricity demand or a drought that limits hydropower production, the state expects to be about 2 million tons below EPA's 2030 target.

"There's no reason for us to delay," said Mary Nichols, head of the ARB. "Obviously, we were surprised and disappointed — as were many other people — by the [Supreme Court] decision, but as we look at where we are and what we need to do, first of all, we still believe very strongly that EPA will prevail, that the Clean Power Plan will be upheld at the end of the day, so it would be foolish to slack off in our efforts to develop approvable plans right now."

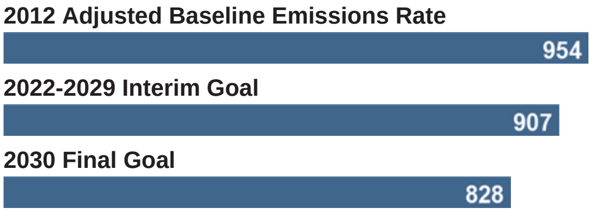
The state has also been in discussions with other Western states about the potential for multistate carbon trading. Nichols said she has been engaged in talks with all other Western states through former Colorado Gov. Bill Ritter's (D) clean energy center at Colorado State University, although she has not put forth a formal proposal yet.

Multistate collaboration also could come through California's grid operator, the California Independent System Operator, which manages the electricity grid for the majority of California and a small portion of Nevada but also has connections to nine other Western states.

"We see the Clean Power Plan as creating opportunities to further modernize the Western grid into a flexible, resilient system that will meet or exceed state and federal environmental goals," said California ISO President and CEO Steve Berberich.

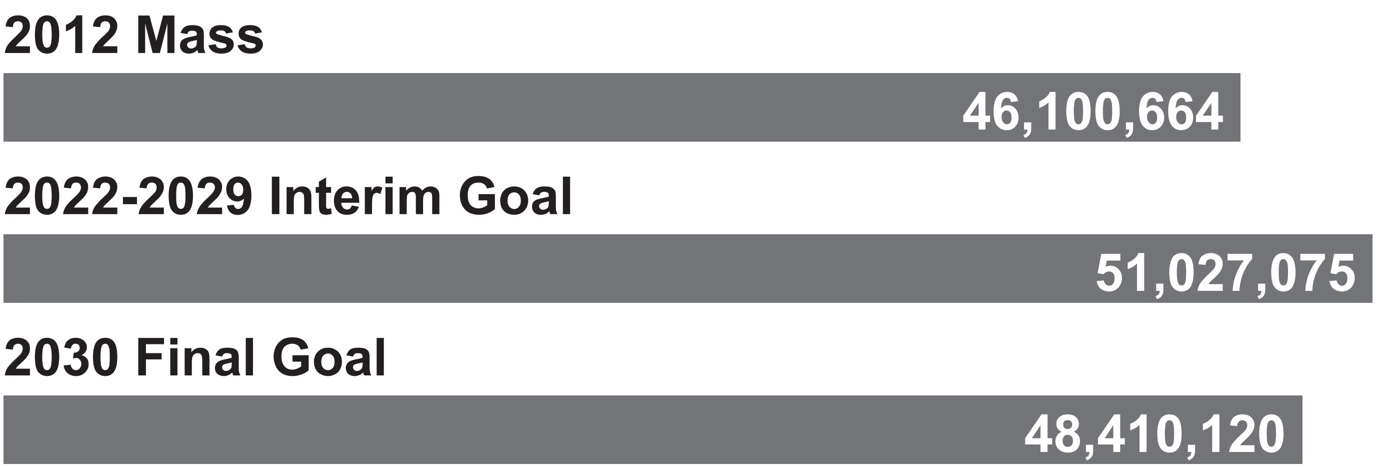
Rate-based emissions

(lbs CO2 / MWh)



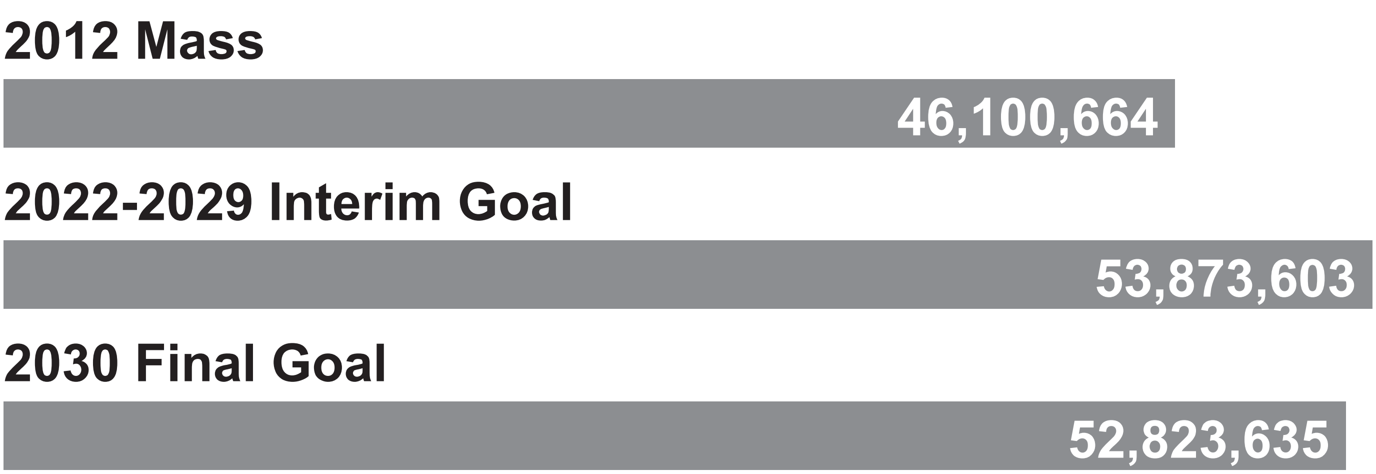
Mass-based emissions

(short tons CO2)



Mass-based (+ new source) emissions

(short tons CO2)



California Air Resources Board
Posted: December 16, 2015

DOCUMENTS

Proposed Compliance Plan
Posted: August 02, 2016

Western States Petroleum Association Comments to CARB
Posted: January 04, 2016

CARB Modeling Analysis
Posted: December 14, 2015

CARB Affected EGU List
Posted: December 14, 2015

CARB Regional and Linkage Considerations Presentation
Posted: December 14, 2015

CARB RPS Adjustment Presentation
Posted: December 14, 2015

CARB Discussion Paper
Posted: September 01, 2015

Testimony to Senate Environment Committee by CARB Chairman Mary Nichols
Posted: March 11, 2015

Statement to FERC by California Public Utilities Commissioner Carla Peterman
Posted: February 19, 2015

State Agencies Scoping Meeting Presentation
Posted: September 09, 2014

DRAFT RULE PUBLIC COMMENTS

Sempra Energy
Posted: December 13, 2014

Los Angeles Department of Water & Power (LADWP)
Posted: December 12, 2014

14 States
Posted: December 10, 2014

California Air Resources Board
Posted: December 08, 2014

Attorney General of New York et al.
Posted: December 06, 2014

Calpine
Posted: December 06, 2014

Pacific Gas and Electric Co.

Southern California Public Power Authority (SCPPA)

Posted: December 06, 2014

California Air Pollution Control Officers Association

Posted: December 03, 2014

Western Electricity Coordinating Council (WECC)

Posted: December 02, 2014

"Western States"

Posted: November 18, 2014

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NATURAL CAPITAL INITIATIVE AT MANOMET REPORT

BIOMASS SUSTAINABILITY AND CARBON POLICY STUDY

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EXECUTIVE SUMMARY

BIOMASS SUSTAINABILITY AND CARBON POLICY

INTRODUCTION

This study addresses a wide array of scientific, economic and technological issues related to the use of forest biomass for generating energy in Massachusetts. The study team, assembled and directed by the Manomet Center for Conservation Sciences, was composed of experts in forest ecosystems management and policy; natural resource economics; and energy technology and policy. The Commonwealth of Massachusetts Department of Energy Resources (DOER) commissioned and funded the study.

The study provides analysis of three key energy and environmental policy questions that are being asked as the state develops its policies on the use of forest biomass.

1. What are the atmospheric greenhouse gas implications of shifting energy production from fossil fuel sources to forest biomass?
2. How much wood is available from forests to support biomass energy development in Massachusetts?
3. What are the potential ecological impacts of increased biomass harvests on forests in the Commonwealth, and what if any policies are needed to ensure these harvests are sustainable?

The goal of the report is to inform the development of DOER's biomass policies by providing up-to-date information and analysis on the scientific and economic issues raised by these questions. We have not been asked to propose specific policies except in the case where new approaches may be needed to protect the ecological functioning of forests. We do not consider non-forest sources of wood biomass (e.g., tree care and landscaping, mill residues, construction debris), which are potentially available in significant quantities but which have very different greenhouse gas (GHG) implications.

This Executive Summary highlights key results from our research and the implications for the development of biomass energy policies in Massachusetts. While certain of the study's insights are broadly applicable across the region (e.g., estimates of excess lifecycle emissions from combustion of biomass compared to fossil fuels), it is also important to recognize that many other conclusions are specific to the situation in Massachusetts—particularly greenhouse gas accounting outcomes that depend on the forest management practices of the state's landowners, which likely differ considerably from those in neighboring states. Nonetheless, the framework and approach that we have developed for assessing the impacts of wood biomass energy have wide applicability for other regions and countries.

SUMMARY OF KEY FINDINGS

Greenhouse Gases and Forest Biomass: At the state, national, and international level, policies encouraging the development of forest biomass energy have generally adopted a view of biomass as a *carbon neutral* energy source because the carbon emissions were considered part of a natural cycle in which growing forests over time would re-capture the carbon emitted by wood-burning energy facilities. Beginning in the 1990s, however, researchers began conducting studies that reflect a more complex understanding of carbon cycle implications of biomass combustion. Our study, which is based on a comprehensive lifecycle carbon accounting framework, explores this more complex picture in the context of biomass energy development in Massachusetts.

The atmospheric greenhouse gas implications of burning forest biomass for energy vary depending on the characteristics of the bioenergy combustion technology, the fossil fuel technology it replaces, and the biophysical and forest management characteristics of the forests from which the biomass is harvested. Forest biomass generally emits more greenhouse gases than fossil fuels per unit of energy produced. We define these excess emissions as the biomass *carbon debt*. Over time, however, re-growth of the harvested forest removes this carbon from the atmosphere, reducing the carbon debt. After the point at which the debt is paid off, biomass begins yielding *carbon dividends* in the form of atmospheric greenhouse gas levels that are lower than would have occurred from the use of fossil fuels to produce the same amount of energy (Figure 1). The full recovery of the biomass carbon debt and the magnitude of the carbon dividend benefits also depend on future forest management actions and natural disturbance events allowing that recovery to occur.

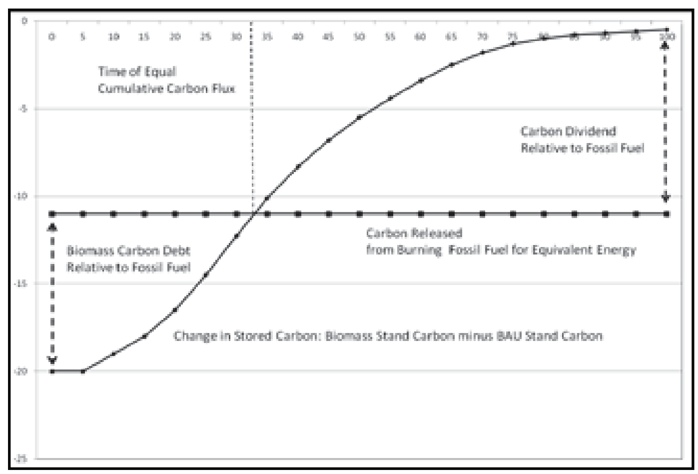

Figure 1 (tonnes of carbon). The schematic above represents the incremental carbon storage over time of a stand harvested for biomass energy wood relative to a typically harvested stand (BAU). The initial *carbon debt* (9 tonnes) is shown as the difference between the total carbon harvested for biomass (20 tonnes) and the carbon released by fossil fuel burning (11 tonnes) that produces an equivalent amount of energy. The *carbon dividend* is defined in the graph as the portion of the fossil fuel emissions (11 tonnes) that are offset by forest growth at a particular point in time. In the example, after the 9 tonnes biomass carbon debt is recovered by forest growth (year 32), atmospheric GHG levels fall below what they would have been had an equivalent amount of energy been generated from fossil fuels. This is the point at which the benefits of burning biomass begin to accrue, rising over time as the forest sequesters greater amounts of carbon relative to the typical harvest.

The initial level of the carbon debt is an important determinant of the desirability of producing energy from forest biomass. Figure 2 provides a summary of carbon debts, expressed as the percentage of total biomass emissions that are in excess of what would have been emitted from fossil fuel energy generation. Replacement of fossil fuels in thermal or combined heat and power (CHP) applications typically has lower initial carbon debts than is the case for utility-scale biomass electric plants because the thermal and CHP technologies achieve greater relative efficiency in converting biomass to useable energy. As a result, the time needed to pay off the carbon debt and begin accruing the benefits of biomass energy will be shorter for thermal and CHP technologies when the same forest management approaches are used in harvesting wood.

Figure 2: Carbon Debt Summary Table

Excess Biomass Emissions as % of Total Biomass Emissions				
Scenarios	**Coal**	**Oil (#6)**	**Oil (#2)**	**Natural Gas**
Electric	31%			66%
Thermal/ CHP		2%-8%	9%-15%	33%-37%

The absolute magnitude and timing of the carbon debts and dividends, however, is sensitive to how landowners decide to manage their forests. Since future landowner responses to increased demand for forest biomass are highly uncertain, we modeled the recovery of carbon in growing forests under a number of alternative management scenarios.

For a scenario that results in relatively rapid realization of greenhouse gas benefits, the switch to biomass yields benefits within the first decade when oil-fired thermal and CHP capacity is replaced, and between 20 and 30 years when natural gas thermal is replaced (Figure 3). Under comparable forest management assumptions, dividends from biomass replacement of coal-fired electric capacity begin at approximately 20 years. When biomass is assumed to replace natural gas electric capacity, carbon debts are still not paid off after 90 years.

Figure 3: Carbon Debt Payoff

Fossil Fuel Technology	Carbon Debt Payoff (yr)
Oil (#6), Thermal/CHP	5
Coal, Electric	21
Gas, Thermal	24
Gas, Electric	>90

Another way to consider greenhouse gas impacts of biomass energy is to evaluate at some future point in time the cumulative carbon emissions of biomass (net of forest recapture of carbon) relative to continued burning of fossil fuels. The Massachusetts Global Warming Solutions Act establishes 2050 as an important reference year for demonstrating progress in reducing greenhouse gas emissions. Figure 4, comparing 40 years of biomass emissions with 40 years of continued fossil fuel burning, shows that replacement of oil-fired thermal/CHP capacity with biomass thermal/CHP fully offsets the carbon debt and lowers greenhouse gas levels compared to what would have been the case if fossil fuels had been used over the same period—approximately 25% lower over the period under a rapid recovery scenario. For biomass replacement of coal-fired power plants, the net cumulative emissions in 2050 are approximately equal to what they would have been burning coal; and for replacement of natural gas cumulative total emissions are substantially higher with biomass electricity generation.

Figure 4: Cumulative Carbon Dividends from Biomass Replacement of Fossil Fuel

Biomass Cumulative % Reduction in Carbon Emissions (Net of Forest Carbon Sequestration)				
Year	**Oil (#6) Thermal/ CHP**	**Coal, Electric**	**Gas, Thermal**	**Gas, Electric**
2050	25%	-3%	-13%	-110%
2100	42%	19%	12%	-63%

Forest Biomass Supply: Future new supplies of forest biomass available for energy generation in Massachusetts depend heavily on the prices that bioenergy facilities are able to pay for wood. At present, landowners in the region typically receive between $1 and $2 per green ton of biomass, resulting in delivered prices at large-scale electricity facilities of around $30 per green ton. Under current policies that are influenced by the competitive dynamics of the electricity sector, we do not expect that utility-scale purchasers of biomass will be able to significantly increase the prices paid to landowners for biomass. Consequently, if future forest biomass demand comes primarily from large-scale electric facilities, we estimate the total "new" biomass that could be harvested annually from forest lands in Massachusetts would be between 150,000 and 250,000 green tons—an amount sufficient to support 20 MW of electric power capacity—with these estimates potentially increasing by 50%–100% when out-of-state forest biomass sources are taken into account (these estimates do not include biomass from land clearing or other non-forest sources such as tree work and landscaping). This is the amount of incremental biomass that would be economically available and reflects the costs of harvesting, processing and transporting this material as well as our expectations about the area of land where harvest intensity is likely to increase. Thermal, CHP, and other bioenergy plants can also compete for this same wood—which could support 16 typically sized thermal facilities or 4 typical CHP plants—and have the ability to pay much higher prices on a delivered basis; thus, they have more options for harvesting and processing forest biomass and can outbid electric power if necessary.

Paying higher prices to landowners for forest biomass could potentially increase forest biomass supplies significantly. For this to occur, electricity prices would need to rise, due to substantially higher fossil fuel prices or significant policy shifts. Thermal, CHP, and pellet facilities can already pay much higher prices for biomass at current energy prices, and would remain competitive if prices paid to landowners were to rise significantly. If these prices were to increase to $20 per green ton, we estimate that supplies of forest biomass from combined in-state and out-of-state sources could be as high as 1.2 to 1.5 million green tons per year. However, this high-price scenario is unlikely given current expectations of fossil fuel prices and existing renewable energy incentives.

Figure 5 shows the potential bioenergy capacity that could be supported from these estimated volumes of "new" forest biomass in Massachusetts. The upper end of the range for Massachusetts forest biomass supplies under our high-price scenario is approximately 885,000 green tons per year—this is close to the annual quantity of biomass that can be harvested without exceeding the annual net growth of the forest on the operable private land base. If additional forest biomass supplies that would be potentially available from out-of-state sources are taken into account, the biomass quantity and number of bioenergy facilities that could be furnished would be 50%–100% higher than shown in this table.

Figure 5: Potential Bioenergy Capacity from "New" Forest Biomass Sources in Massachusetts

	Green Tons per Year
Current Massachusetts Harvest *	325,000
Potential Forest Biomass Supply (Massachusetts only) **	
Current Biomass Prices	200,000
High-Price Scenario	800,000
	Number of Facilities
Electric Power Capacity: Number of 50 MW Plants	
Current Biomass Prices	0.4
High-Price Scenario	1.6
Thermal Capacity: Number of 50 MMBtu/hr Plants ***	
Current Biomass Prices	16
High-Price Scenario	62
CHP Capacity: Number of 5 MW/34 MMBtu/hr Plants ***	
Current Biomass Prices	4
High-Price Scenario	15

Notes: * *Average of industrial roundwood for 2001–2009.*
** *Based on mid-point of the range of volumes estimated for new biomass in Massachusetts.*
*** *Thermal plants are assumed to operate 1800 hours per year, while CHP plants operate 7200 hours per year.*

Forest Sustainability and Biomass Harvests: In Massachusetts, the possibility of increased harvesting of biomass for energy has raised a number of sustainability issues at both the landscape and stand levels. At the landscape scale, potential impacts to a broad range of societal values arise with increases in biomass harvesting. However, in our low-price scenario for biomass, we anticipate that harvested acreage will not increase from current levels—biomass will come from removal of logging residues and poor quality trees at sites that would be harvested for timber under a business-as-usual scenario. Furthermore, in this scenario the combined volume of timber and biomass harvests represents less than half of the annual net forest growth across the state's operable private forest land base. Under our high-price biomass supply scenario, although harvests still represent annual cutting on only about 1% of the forested lands in the state, the total harvest levels approach the total amount of wood grown each year on the operable private forest land base.

Under either price scenario, however, harvests for bioenergy facilities could have more significant local or regional impacts on the landscape. These might include aesthetic impacts of locally heavy harvesting as well as potential impacts on recreation and tourism and the longer-term health of the wood products sector of the economy. We have outlined four general options encompassing a wide range of non-regulatory and regulatory approaches that the state may wish to consider if it determines that further actions are needed to protect public values at the landscape scale.

- Option 1: Establish a transparent self-monitoring, self-reporting process for bioenergy facilities designed to foster sustainable wood procurement practices.
- Option 2: Require bioenergy facilities to purchase wood from forests with approved forest management plans.
- Option 3: Require bioenergy facilities to submit wood supply impact assessments.
- Option 4: Establish formal criteria for approval of wood supply impact assessments—possible criteria might include limits on the amount of harvests relative to anticipated forest growth in the wood basket zone.

At the stand level, the most significant sustainability concerns associated with increased biomass harvests are maintenance of soil productivity and biodiversity. Current Chapter 132 Massachusetts forest cutting practices regulations provide generally strong protection for Massachusetts forests, especially water quality; however, they are not currently adequate to ensure that biomass harvesting is protective of ecological values across the full range of site conditions in Massachusetts. Other states and countries have recently adopted biomass harvesting guidelines to address these types of concerns, typically through new standards that ensure (1) enough coarse woody debris is left on the ground, particularly at nutrient poor sites, to ensure continued soil productivity and (2) enough standing dead wildlife trees remain to promote biodiversity. While the scientific literature does not provide definitive advice on the appropriate practices for Massachusetts' forests, recent guidance from the Forest Guild and other states provides the State Forestry Committee with a useful starting point for developing additional stand level standards that ensure continued protection of ecological values in Massachusetts forests.

CHAPTER 1

INTERNATIONAL AND U.S. FOREST BIOMASS ENERGY POLICIES

1.1 OVERVIEW

International and U.S. domestic forest biomass energy policies form a critical backdrop to the analyses presented in this report. The purpose of this introductory chapter is to provide a general understanding of (1) the development of policies that have driven the growth of the biomass energy sector; (2) the key policy instruments that have been relied upon to promote this development; and (3) a summary of recent discussions about the greenhouse gas (GHG) implications of forest biomass energy.

The chapter is organized into two major sections. The first reviews international biomass energy policies—focusing on the historical development of these policies, discussing the policy instruments in place that promote biomass development, and summarizing recent concerns about the impact on GHG of emissions from biomass energy facilities. The second section provides a more detailed review of U.S. energy policies affecting forest biomass both at the federal and state levels, with a particular focus on policies in Massachusetts.

1.2 INTERNATIONAL FOREST BIOMASS ENERGY POLICIES

1.2.1 Historical Context

The late 20th century development of forest biomass energy facilities originated from energy security concerns triggered by the 1973–1974 oil crisis. The International Energy Agency (IEA) was founded at this time primarily to address the security issue.

> *Energy Security can be described as "the uninterrupted physical availability at a price which is affordable, while respecting environment concerns." The need to increase "energy security" was the main objective underpinning the establishment of the IEA. With particular emphasis on oil security, the Agency was created in order to establish effective mechanisms for the implementation of policies on a broad spectrum of energy issues: mechanisms that were workable and reliable, and could be implemented on a co-operative basis (International Energy Agency, 2010).*

Although IEA's original founding agreements did not explicitly address forest biomass, the agency created IEA Bioenergy in 1978 with:

> *...the aim of improving cooperation and information exchange between countries that have national programmes in bioenergy research, development and deployment (IEA Bioenergy, 2010).*

Our review of available documents suggests that prior to IEA Bioenergy's 1998–2002 Strategic Plan (IEA Bioenergy, NA), the greenhouse gas implications of forest biomass combustion were not a primary area of research for the organization (IEA Bioenergy, 1995). Moreover, recent IEA policies have continued to reflect the view that biomass combustion is "close to carbon neutral in most instances" (International Energy Agency, 2007).

In fact, from a climate change perspective, the desirability of biomass energy appears to have been the prevailing wisdom of international bioenergy policies over most of the past ten or fifteen years. These policies have generally equated burning of biomass from renewable sources with "climate friendly" outcomes. The presumption has been that as long as the harvested areas grow back as forests, the emitted CO_2 emissions will be recaptured in the growing trees, resulting in lower net CO_2 emissions over time across the entire energy generation sector. For example, in a 2000 study of forestry and land use, the Intergovernmental Panel on Climate Change (IPCC), the lead international organization charged with assessing impacts of greenhouse gas emissions, stated that:

> *Biomass energy can be used to avoid greenhouse gas emissions from fossil fuels by providing equivalent energy services: electricity, transportation fuels, and heat. The avoided fossil fuel CO_2 emissions of a biomass energy system are equal to the fossil fuels substituted by biomass energy services minus the fossil fuels used in the biomass energy system. These quantities can be estimated with a full fuel-cycle analysis of the system. The net effect on fossil fuel CO_2 emissions is evident as a reduction in fossil fuel consumption (IPCC, 2000).*

In its most recent 2007 assessment, IPCC noted that:

> *In the long term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fibre or energy from the forest, will generate the largest sustained mitigation benefit.*
>
> *For the purpose of this discussion, the options available to reduce emissions by sources and/or to increase removals by sinks in the forest sector are grouped into four general categories (1)...(4) increasing the use of biomass-derived energy to substitute fossil fuels (IPCC, 2007).*

European Union policies also promote the use of forest biomass energy, as embodied in the EU's 2006 Forest Action Plan:

> *The EU has adopted an ambitious energy and climate policy which aims by 2020 to reduce energy consumption by 20%, with a similar cut in CO_2 emissions, while raising the share of renewables in the EU's energy mix to 20%.*
>
> *More than half of the EU's renewable energy already comes from biomass, 80% of which is wood biomass. Wood can play an important role as a provider of biomass energy to offset fossil fuel emissions, and as an environmentally friendly material. There has recently been higher demand for wood from the energy sector in addition to*

rising demand from the established wood-processing industries. Many experts consider that significantly more wood could be mobilised from EU forests than is currently the case. However, the cost at which this can be done is the key factor (EU, 2006).

In approving the Forest Action Plan, the Commission of European Communities identified a variety of key actions, including:

Key action 4: Promote the use of forest biomass for energy generation

Using wood as an energy source can help to mitigate climate change by substituting fossil fuel, improving energy self-sufficiency, enhancing security of supply and providing job opportunities in rural areas.

The Standing Forestry Committee will support the implementation of the Biomass Action Plan (Commission of European Communities, 2005) in particular concerning the development of markets for pellets and chips and information to forest owners about the opportunities of energy feedstock production.

The Commission will facilitate investigation and dissemination of experience on mobilisation of low-value timber, small-sized wood and wood residues for energy production. The Member States will assess the availability of wood and wood residues and the feasibility of using them for energy production at national and regional levels, in order to consider further actions in support of the use of wood for energy generation. The 7th Research Framework Programme and the IEE-CIP provide the necessary possibilities to facilitate such activities.

The Commission will continue to support research and development of technologies for the production of heat, cooling, electricity and fuels from forest resources in the energy theme of the 7th Research Framework Programme's cooperation specific programme, and to encourage the development of the biofuel technology platform and support the implementation of its research agenda through the 7th Research Framework Programme (Commission of European Communities, 2006).

1.2.2 Policy Instruments

Energy policies for forest biomass are embedded in a broader system of policies promoting the development of renewable energy sources. These policies are typically implemented through incentive schemes such as feed-in tariffs that guarantee favorable purchase prices for renewables and through Renewable Portfolio Standards (RPS) requiring that renewable sources constitute a certain minimum percentage of energy generation. A 2009 status report from the Renewable Energy Policy Network for the 21st Century (REN21) provides summary data characterizing the renewable energy policies of countries around the globe. According to REN21:

By early 2009, policy targets existed in at least 73 countries, and at least 64 countries had policies to promote renewable power generation, including 45 countries and 18 states/provinces/territories with feed-in tariffs (many of these recently updated). The number of countries/states/provinces with renewable portfolio standards increased to 49. Policy targets for renewable energy were added, supplemented, revised, or clarified in a large number of countries in 2008 (Renewable Energy Policy Network for the 21st Century, 2009).[1]

By allowing projects to qualify for feed-in tariffs and be counted towards RPS goals, designation of forest biomass as a renewable energy source has been an important driver of biomass energy project development. The REN21 status report indicates that by the end of 2008, 52 GW of biomass power capacity existed worldwide, about evenly split between developed and developing countries. The European Union and United States accounted for 15 GW and 8 GW of this capacity, respectively. About 2 GW of this total were added in 2008, an annual increase of approximately 4 percent.

Within the broad context of biomass energy policies, individual countries have emphasized different policy instruments. A variety of researchers have conducted assessments of country-specific impacts of biomass policies—for an excellent summary see (Junginger, 2007). Faaij (2006) points out that:

All EU-15 countries implemented policies for supporting bioenergy. These include the deployment of compensation schemes, tax deduction (in some cases specifically aimed at biofuels), feed-in tariffs, tax incentives, energy tax exemption, bidding schemes, CO_2-tax and quota. Precise targets on the national level differ strongly however and are hard to compare because of differences in definitions and fuels in or excluded (such as MSW and peat).The same is true for the level of (financial) support provided through the various programs and instruments. The different countries clearly have chosen very different approaches in developing and deploying various bio-energy options. Partly this is caused by the natural conditions (type of resources and crops, climate) and the structure of the energy system, and also by the specific political priorities linked to the agricultural and forestry sectors in those countries.

A general conclusion of these studies is that higher rates of biomass energy development are typically a function not of any single factor but instead result from the combined effects of a variety of policy instruments, in the context of a country's existing mix of energy sources and the degree of development of its forestry sector (Kautto, 2007; Junginger, 2007). For example, Sweden is one of the European countries that have most rapidly adopted biomass energy systems. Two key factors have been identified as

[1] For an extensive list of countries and their policies, see Table 2, pages 23–24, www.ren21.net/pdf/RE2007_Global_Status_Report.pdf, and pages 17–18 of www.ren21.net/pdf/RE_GSR_2009_Update.pdf

the basis for this growth. First is the presence of a large and well-developed forest products sector. Second, the design of Sweden's tax system has strongly encouraged biomass development through a range of mutually reinforcing policies.

> *Overall it appears that taxation has been a very effective policy instrument in increasing biomass utilisation in Sweden throughout the 1990's. This has particularly been the case in the heat sector, but, following market liberalisation, significant increases in the electricity sector have also been noted. It should be noted in this respect that the Swedish tax regime is long established and comprises multiple layers of VAT, energy and CO_2 taxes, increasing the effectiveness of tax increases. There is also a complex and frequently modified system of allocating rebates to certain industries that has enabled the tax to be augmented as required to encourage biomass use at the expense of fossil fuels, while maintaining competitive industrial advantage (Cooper & Thornley, 2007).*

On the other hand, Faaij (2006) points out that France's focus on biofuels and heat is primarily a function of excess capacity in its nuclear electricity production sector, making electrical generation from biomass unattractive.

The government policies of non-European countries also could dramatically increase biomass energy generation. For example, China has established a variety of policy goals that will promote biomass energy development (Roberts, 2010). By 2020, China is proposing to build 24 GW of biomass power capacity, equivalent to more than eight 25 MW plants per month over the next decade, although Roberts notes this is overly ambitious and likely to be downgraded to 10 GW. Although most of China's biomass appears to be based on agricultural wastes, plans do include increasing wood pellet production from two million tons per year in 2010 to 50 million tons per year by 2020 and developing 13.3 million hectares of forests to produce biomass feedstock. According to Roberts (2010), China has accounted for 23 percent of recent worldwide investment in biomass energy (compared with Europe's 44 percent share). Policies in large forested countries like Canada are also aimed at promoting biomass energy development, although Roberts notes that Canada has been slow in developing its bioenergy resources and that most "meaningful" biomass policies are being put in place at the provincial level, for example Ontario's feed-in tariffs and British Columbia's carbon tax.

Overall, growth of the biomass sector internationally could have important implications for the U.S. and Massachusetts. In Britain, two 300 MW biomass power plants are currently in the planning stages. These plants are projected to consume six million green tons of wood chips annually, purchased from around the globe, with New England identified as a possible source of woodchips (MGT Power, 2010). Given the potential for such increased international trade in biomass, Massachusetts forests could become suppliers of biomass regardless of whether any biomass plants are actually built in the state.

1.2.3 Sustainability Concerns

Although mainstream policies continue to promote biomass as a renewable and carbon friendly fuel, the international policy framework is beginning to require more detailed assessments of the carbon implications of bioenergy development. This more sophisticated approach to understanding the greenhouse gas implications of climate policy dates from the 1990s when researchers began building formal models to explore the impacts of biomass combustion on greenhouse gas levels, for example studies by Marland and Schlamadinger (1995).[2] Work along these lines became a prominent feature of research conducted IEA Bioenergy Task 38, which is focused directly on the climate change implications of biomass combustion for energy. Researchers contributing to Task 38 have pointed out the difficulty of generalizing about the climate benefits of biomass combustion. This view was expressed in a December 2009 status report from IEA Bioenergy issued to coincide with the Copenhagen conference on climate change. This report provided a clearly articulated summary of the current, and in our view state-of-the-art, thinking on the impacts of forest biomass combustion on greenhouse gases.

> *Ranking of land use options based on their contribution to climate change mitigation is also complicated by the fact that the performance of the different options is site-specific and is determined by many parameters. Among the more critical parameters are:*
>
> - *Biomass productivity and the efficiency with which the harvested material is used—high productivity and efficiency in use favour the bioenergy option. Low productivity land may be better used for carbon sinks, given that this can be accomplished without displacing land users to other areas where their activities lead to indirect CO_2 emissions. Local acceptance is also a prerequisite for the long-term integrity of sink projects.*
> - *The fossil fuel system to be displaced—the GHG emissions reduction is for instance higher when bioenergy replaces coal that is used with low efficiency and lower when it replaces efficient natural gas-based electricity or gasoline/diesel for transport.*
> - *The initial state of the land converted to carbon sinks or bioenergy plantations (and of land elsewhere possibly impacted indirectly)—conversion of land with large carbon stocks in soils and vegetation can completely negate the climate benefit of the sink/bioenergy establishment.*
> - *The relative attractiveness of the bioenergy and carbon sink options is also dependent on the timescale that is used for the evaluation. A short timeframe (a few*

[2] For a more complete list of Task 38 background papers from the 1990s, see www.ieabioenergy-task38.org/publications/backgroundpapers/backgroundpapers.htm#marland1

decades) tends to favour the sink option, while a longer timeframe favours the bioenergy option. The reason is that the accumulation of carbon in forests and soils cannot continue endlessly—the forest eventually matures and reaches a steady state condition. This is also the case for soils. In contrast, bioenergy can be produced repeatedly and continue to deliver greenhouse gas emissions reduction by substituting fossil fuels.

The bioenergy and carbon sink options obviously differ in their influence on the energy and transport systems. Bioenergy promotion induces system changes as the use of biofuels for heat, power, and transport increases. In contrast, the carbon sink option reduces the need for system change in relation to a given climate target since it has the same effect as shifting to a less ambitious climate target. The lock-in character of the sink option is one disadvantage: mature forests that have ceased to serve as carbon sinks can in principle be managed in a conventional manner to produce timber and other forest products, offering a relatively low GHG reduction per hectare. Alternatively, they could be converted to higher yielding energy plantations (or to food production) but this would involve the release of at least part of the carbon store created. On the other hand, carbon sinks can be viewed as a way to buy time for the advancement of climate-friendly energy technologies other than bioenergy. Thus, from an energy and transport systems transformation perspective, the merits of the two options are highly dependent on expectations about other energy technologies (IEA Bioenergy, 2009).

Growing concerns about greenhouse gas impacts of forest biomass policies also surfaced recently in journal articles by Johnson (2008) and by Searchinger, et al. (2009). The Searchinger article, appearing in Science and titled "Fixing a Critical Climate Accounting Error," points out that rules for applying the Kyoto Protocol and national cap-and-trade laws contain a major flaw in that the CO_2 emissions from biomass energy are not properly taken into account because they embody the implicit assumption that all biomass energy is carbon neutral. Consistent with the recent IEA report discussed above, Searchinger's critique states:

The potential of bioenergy to reduce greenhouse gas emissions inherently depends on the source of the biomass and its net land-use effects. Replacing fossil fuels with bioenergy does not by itself reduce carbon emissions, because CO_2 released by tailpipes and smokestacks is roughly the same per unit of energy regardless of the source. Bioenergy therefore reduces greenhouse gases only if the growth and harvesting of the biomass for energy capture carbon above and beyond what would be sequestered anyway and thereby offset emissions from energy use. This additional carbon may result from land management changes that increase plant uptake or from the use of biomass that would otherwise decompose rapidly.

In on-line supporting material for the Science article, Searchinger et al. note that:

Use of forests for electricity on additional carbon: *Roughly a quarter of anthropogenic emissions of carbon dioxide are removed from the atmosphere by the terrestrial carbon sink, of which the re-growth of forests cut in previous decades plays a major role. Any gain in carbon stored in regenerating forests contributes to the sink, so activities that keep otherwise regenerating forests to constant levels of carbon reduces that sink relative to what would have occurred without those activities.*

The net effect of harvesting wood for bioenergy is complicated and requires more analysis. Each ton of wood consumed in a boiler instead of coal does not significantly alter combustion emissions. However, some of the wood in standing timber is typically not utilized and is left to decay in the forest or nearby, causing additional emissions. Much of the carbon in roots will also decompose. Replanting may accelerate release of carbon from forest soils. As the forest regenerates following cutting, it may sequester carbon faster or slower than would have occurred in the absence of the harvesting, depending on the previous forest's age, site quality and forest type. Over long periods, the carbon stocks of the forests with and without the harvest for biofuels may be equal. For this reason, how different emissions are valued over time plays an important role in estimating the net carbon effects of harvesting wood for use as a bioenergy.

In Europe, policies towards biomass may be beginning to reflect this more complex view of potential greenhouse gas impacts. A 2009 EU policy directive recognizes the need to demonstrate the sustainability of biomass energy, and specifies that the European Commission complete such a study.

Section 75: The requirements for a sustainability scheme for energy uses of biomass, other than bioliquids and biofuels, should be analysed by the Commission in 2009, taking into account the need for biomass resources to be managed in a sustainable manner (European Parliament and Council, 2009).

However, the results of this recently completed study of biomass sustainability take as a starting point the presumption of biomass carbon neutrality—adopting the long-term view that CO_2 emissions from combusted biomass eventually will be recaptured as long as the forests are regenerated. In this context, the report goes on to discuss a variety of recommended policy options including ones to ensure that all biomass is sourced from certified sustainable supplies. To the extent that this new report becomes the basis for future EU policies, such policies would appear to adopt a very long-term view of the relevant timeframe for biomass policies, one that does not place great emphasis on the potential for shorter term increases in CO_2 flux that likely result from forest biomass energy generation.

At the broader international level, the IPCC is also in the processing of preparing a new report on renewable energy that is expected to be published in 2011. Initial indications are that this report will provide more detailed considerations of the carbon issue for forest biomass.

1.3 U.S. FEDERAL FOREST BIOMASS ENERGY POLICIES

1.3.1 Most Significant Federal Programs & Incentives for Biomass Energy

Federal incentives for renewable energy (including forest biomass) have taken many forms over the past four decades. The focus of most of these programs has been on encouraging renewable electricity generation and, more recently, production of renewable transportation fuels, such as ethanol. The third area of energy use—thermal applications for heat, cooling and industrial process heat—has not been a focus of federal energy programs until very recently. A summary of the full scope of existing federal programs and incentives related to the development of biomass energy facilities is included as Appendix 1-A to this report.

Federal policy initially encouraged renewable electricity generation by requiring utilities to purchase electricity from renewable energy generators at a fixed cost through the Public Utility Regulatory Policy Act (PURPA). More recently, federal policy has shifted towards encouraging renewable energy through tax incentives and direct grants—with the primary focus on renewable transportation fuels and renewable electricity generation.

The thrust of current federal investment in renewable energy is summarized in a recent report by the Environmental Law Institute (Environmental Law Institute, 2009). From 2002 through 2008 the U.S. Government spent approximately $29 billion on renewable energy subsidies (compared to $72 billion spent on fossil fuels). Of this $29 billion, most was dedicated to transportation fuels or electricity generation through a combination of tax programs and direct grants and loans.

- **Transportation fuels** via corn-based ethanol production received more than half of the total subsidies ($16 billion), primarily through the Volumetric Ethanol Excise Tax Credit Program (VEETC) ($11 billion) and the corn-based ethanol grant program ($5 billion).
- **Renewable electricity generation** projects received approximately $6 billion in subsidies during this seven-year period, principally through the Production Tax Credit ($5 billion), the Investment Tax Credit ($250 million), the Modified Accelerated Cost Recovery System ($200 million), and the Clean Renewable Energy Bond program ($85 million).
- **Thermal energy** as a sector received no significant subsidies.

Within the electric power sector biomass facilities are eligible for funding under these four primary renewable electricity generation incentives (the Production Tax Credit, Investment Tax Credit, Modified Accelerated Cost Recovery System, and Clean Renewable Energy Bond program); however they have received a relatively small share of the total funding. The U.S. Energy Information Administration (EIA) estimates that in fiscal year 2007, open-loop biomass facilities received approximately $4 million in tax credits under the production tax credit program, compared to approximately $600 million for wind facilities. Funding for combined heat and power or purely thermal facilities is also negligible compared to expenditures on other renewable resources (EIA, 2008). And many of the biomass-specific grant programs have total annual allocations in the $1 to $5 million range, with individual projects often capped in the $50,000 to $500,000 range.

The primary federal subsidy or incentive to biomass electric power production is the Renewable Electricity Production Tax Credit which provides $0.011 per kWh or approximately $10 per MWh.[3] As discussed more fully below, while smaller in value than state Renewable Energy Credits (REC's), which currently average between $20–$35 per MWh, the PTC does provide a significant and stable incentive for the development of biomass power over time. The American Recovery and Reinvestment Act of 2009 allows taxpayers eligible for the federal renewable electricity production tax credit (PTC) to take the federal business energy investment tax credit (ITC) or to receive a grant from the U.S. Treasury Department instead of taking the PTC for new installations for up to 30% of capital costs following the beginning of commercial production. The new law also allows taxpayers eligible for the business ITC to receive a grant from the U.S. Treasury instead of taking the business ITC for new installations. Grants are available to eligible properties placed in service in 2009 or 2010, or if completed by 2013.

Within federal subsidies specific to biomass energy, there is an even greater emphasis on transportation fuels, a very limited focus on biomass power, and no historic public policy support for biomass thermal applications.

In addition to the federal Production Tax Credit, the Biomass Crop Assistance Program (BCAP) has provided significant subsidies over the past year to the biomass supply sector. However, it is considered unlikely that the current high level of subsidies will continue. Created in the 2008 Farm Bill, BCAP (sec. 9011) is an innovative program intended to support establishment and production of eligible crops for conversion to bio-energy, and to assist agricultural and forest landowners with collection, harvest, storage, and transportation (CHST) of these eligible materials to approved biomass conversion facilities (BCF).

3 The federal renewable electricity production tax credit (PTC) is a per-kilowatt-hour tax credit for electricity generated by qualified energy resources and sold by the taxpayer to an unrelated person during the taxable year. Originally enacted in 1992, the PTC has been renewed and expanded numerous times, most recently by H.R. 1424 (Div. B, Sec. 101 & 102) in October 2008 and again by H.R. 1 (Div. B, Section 1101 & 1102) in February 2009. Efforts to again renew the PTC are currently underway in the US Congress.

The program pays for up to 75% of establishment costs of new energy crops. In addition, farmers participating in a selected BCAP project area surrounding a qualifying BCF can collect five years of payments (15 years for woody biomass) for the establishment of new energy crops. An additional matching payment of up to $45/ton (on a $1 to $1 basis) to assist with collection, harvest, storage and transportation (CHST) of an eligible material to a BCF will also be available for a period of two years.

The launch of this new program has resulted in a substantial new subsidy for the existing wood market with significant market impact. Large numbers of existing biomass conversion facilities (led by lumber, pellet and paper mills currently burning wood for their own energy use without a federal subsidy) submitted applications to USDA to be approved as qualifying facilities. Consequently, funds obligated (though not yet spent) for BCAP through the end of March 2010 soared to over $500 million, more than seven times BCAP's estimated budget of $70 million in the 2008 Farm Bill. The USDA now estimates BCAP costs at $2.1 billion on CHST from 2010 through 2013.

USDA has allocated $2.1 million to Massachusetts for BCAP payments and $500,000 has been dispersed to date. Despite broad outreach (11 public meetings and other efforts), BCAP enrollment has been limited in the state, probably due to the limited array of biomass facilities. In Massachusetts, there are two qualifying biomass conversion facilities (BCF): Pinetree Power (17 MW electric generation facility) and LaSalle Florists, a very small greenhouse operation (USDA, 2010). Pinetree Power has about 20–25 suppliers that are approved eligible material owners (EMO). Based on interviews with procurement personnel at the Pinetree facility, the long-term impact of BCAP is unknown at this point. Overall, it is perceived to have created instability in the supply sector, potentially cutting costs for the electric power industry, but increasing costs for other competing industries that are not enrolled in the program. In Pinetree's view, it also might encourage overcutting in response to the short-term subsidy to suppliers. The lack of forest management requirements for the program was also noted. [4]

Based on interviews with Cousineau Forest Products, a leader in the wood brokerage industry for pulp, chips and biomass supplies across New England and the east, approximately 50% of the BCAP subsidy is being passed onto qualifying facilities from suppliers in the form of lower prices paid for fuel. Consequently, as currently structured, the BCAP program is significantly lowering fuel costs for the biomass power sector. Where landholdings are small, such as in Massachusetts, these savings generally accrue to loggers and the biomass consumers. In areas with larger landholdings, more of these savings go to landowners.

4 Pinetree Power information based on interviews with Tim Haley who prepared their BCAP application and Jamie Damman (M.S.) forester and wood buyer for North Country Procurement, consultant to Pinetree Power.

The Commodity Credit Corporation (CCC) has issued a draft rule to implement BCAP specifying the requirements for eligible participants, biomass conversion facilities, and biomass crops and materials. Public comment on the draft rule closed on April 9, 2010. Comments on the rule address a diversity of issues ranging from overall support for the continuation of the program to concern that the initial focus on CHST payments has resulted in a substantial new subsidy for the existing woody-biomass market, creating market distortions and instability in the supply sector, cutting costs for some users (e.g., biomass power plants) and increasing costs for other competing industries (OSB manufacturers and other users of bark and chips). In addition, some comments have raised the issue of the absence of forest management requirements in BCAP could encourage overcutting in response to the short term subsidy to suppliers. Others have spoken to the need to focus BCAP on directing more resources towards the establishment and production of new energy crops, so the program can fulfill its purpose of expanding the amount of biomass available for alternative energy.

1.3.2 Environmental Protection Agency Position on Biomass Energy and Carbon Accounting [5]

As determined by the Environmental Protection Agency in their final rule on Mandatory Reporting of Greenhouse Gases, electric generation and thermal facilities are not required to count emissions associated with biomass combustion when determining whether they meet or exceed the threshold for reporting (emission of 25,000 metric tons per year for all aggregated sources at a facility). But if the threshold is exceeded, facilities are required to separately report emissions associated with the biomass combustion. Thus, facilities that rely primarily on biomass fuels are not be required to report under the rule (EPA, 2009).

This approach is consistent with IPCC Guidelines for National Greenhouse Gas Inventories, which require the separate reporting of CO_2 emissions from biomass combustion, and the approach taken in the U.S. Inventory of Greenhouse Gas Emissions and Sinks. Separate reporting of emissions from biomass combustion is also consistent with some State and regional GHG programs, such as California's mandatory GHG reporting program, the Western Climate Initiative, and The Climate Registry, all of which require reporting of biogenic emissions from stationary fuel combustion sources. While this reporting requirement does not imply whether emissions from combustion of biomass will or will not be regulated in the future, the data collected will improve EPA's understanding of the extent of biomass combustion and the sectors of the economy where biomass fuels are used. It will also allow EPA to improve methods for quantifying emissions through testing of biomass fuels.

5 Much of this section is drawn directly and/or quoted verbatim from the EPA's Response to Public Comments Volume No.: 1 Selection of Source Categories to Report and Level of Reporting, September 2009

This rule is based on the EPA's basic premise that burning biomass for energy is considered to be carbon-neutral when considered in the context of natural carbon cycling:

> *Although the burning of biomass also produces carbon dioxide, the primary greenhouse gas, it is considered to be part of the natural carbon cycle of the earth. The plants take up carbon dioxide from the air while they are growing and then return it to the air when they are burned, thereby causing no net increase. Biomass contains much less sulfur and nitrogen than coal; therefore, when biomass is co-fired with coal, sulfur dioxide and nitrogen oxides emissions are lower than when coal is burned alone. When the role of renewable biomass in the carbon cycle is considered, the carbon dioxide emissions that result from co-firing biomass with coal are lower than those from burning coal alone (EPA, 2010).*

Regarding consideration of life-cycle emissions, the EPA has stated that preparation of a complete life cycle analysis is beyond the scope of this rule:

> *With respect to emissions and sequestration from agricultural sources and other land uses, the rule does not require reporting of emissions or sequestration associated with deforestation, carbon storage in living biomass or harvested wood products. These categories were excluded because currently available, practical reporting methods to calculate facility-level emissions for these sources can be difficult to implement and can yield uncertain results. Currently, there are no direct GHG emission measurement methods available except for research methods that are very expensive and require sophisticated equipment (EPA, 2009).*

Regarding biomass-derived transportation fuels, the Energy Independence and Security Act of 2007 (EISA) (P.L. 110–140) required EPA to establish a rule for mandatory lifecycle GHG reduction thresholds for various renewable liquid transportation fuel production pathways, including those using wood as a feedstock. Each qualifying renewable fuel must demonstrate that net GHG emissions are less than the lifecycle GHG emissions of the 2005 baseline average for the fossil fuel that it replaces. For non-agricultural feedstocks, renewable fuel producers can comply with the regulation by: (1) collecting and maintaining appropriate records from their feedstock suppliers in order to demonstrate that feedstocks are produced in a manner that is consistent with the renewable biomass requirements outlined in the ruling, or (2) fund an independent third party to conduct annual renewable biomass quality-assurance audits based on an a framework approved by EPA.

1.3.3 Pending Federal Climate and Energy Legislation

Pending federal climate and energy legislation continues to be in flux, with an uncertain future and significantly evolving content. Overall, these bills focus primarily on the production of renewable electricity and transportation fuels rather than production of thermal energy. In all of the various versions of these bills, energy produced from biomass is considered to be renewable and carbon neutral and generally excluded from proposed caps on carbon emissions and related proposals for carbon emission allowances. There is continuing debate about the definition of biomass from qualifying sources and various proposals to provide safeguards for natural resources on public and/or private lands. This debate also includes consideration of sustainability requirements or guidelines for biomass to qualify as a renewable fuel. There is concern that aggressive targets for increasing the use of biomass for production of renewable electricity and transportation fuels from the current Renewable Fuels Standard, a proposed Renewable Electricity Standard and a limit on carbon emissions would outstrip the capacity of our nation's forests to provide an economically and ecologically sustainable supply. To ensure sustainable harvesting levels and accurate accounting of carbon emissions and re-sequestration, there is discussion and debate about including emissions from renewable biomass energy under proposed carbon caps based on full lifecycle accounting. At this point, however, it is unclear what direction will emerge in this developing legislation.

1.4 MASSACHUSETTS FOREST BIOMASS ENERGY POLICIES

Massachusetts has implemented policies to increase the use of biomass to meet energy needs in the electricity sector, the transportation sector, and the building heating sector, although as is the case at the federal level, state policies have been focused primarily on using biomass to replace fossil fuels in the electricity and transportation sectors. Combined with the state's regulatory structure for implementing the Regional Greenhouse Gas Initiative (RGGI) (which sets an emissions cap on fossil fuel electrical generation systems of 25 megawatts or greater), this has created significant incentives driving the state towards greater reliance on biomass electric generation capacity. A recent exception to this trend is the Massachusetts Green Communities Act of 2008, which established new Renewable and Alternative Energy Portfolio Standards (RPS and APS) that allow eligible CHP units to receive credits for useful thermal energy. This program promotes the installation and effective operation of new CHP units for residential, commercial, industrial, and institutional applications. Overall, the bill significantly reforms the state's energy policy, and makes large new commitments to electric and natural gas energy efficiency programs, renewables, and clean fossil fuels like combined heat and power (Environment Northeast, 2008).

Massachusetts has two regulatory programs that directly impact the incentives for developing biomass-fueled electricity in the state. The first is the Massachusetts Renewable Portfolio Standard (RPS), which is administered by the Department of Energy Resources (DOER), and the second is the implementation of the state's membership in the Regional Greenhouse Gas Initiative (RGGI), which is administered by the Department of Environmental Protection (DEP).

1.4.1 Massachusetts Renewable Portfolio Standard

The Massachusetts RPS program currently mandates that all retail electricity suppliers must include minimum percentages of RPS Class I Renewable Generation, RPS Class II Renewable Generation, and RPS Class II Waste Energy in the retail electricity they sell to consumers. For 2010, the Class I requirement is 5%, the Class II Renewable requirement is 3.6%, and the Class II Waste requirement is 3.5%. The definition of "eligible biomass fuel" under the RPS program is:

> *Fuel sources including brush, stumps, lumber ends and trimmings, wood pallets, bark, wood chips, shavings, slash and other clean wood that are not mixed with other unsorted solid wastes; by-products or waste from animals or agricultural crops; food or vegetative material; energy crops; algae; organic refuse-derived fuel; anaerobic digester gas and other biogases that are derived from such resources; and neat Eligible Liquid Biofuel that is derived from such fuel sources.*

It is notable that this definition contains no "sustainability" requirement. The RGGI definition, by contrast, does contain such a requirement, though the criteria for sustainability in that definition are not fleshed out at this time. This definition also includes liquid biofuels, which are expressly excluded from the definition of "eligible biomass" for purposes of the Massachusetts RGGI program.

Biomass facilities may qualify as RPS Class I or Class II generation units as long as they are classified as "low-emission, advanced biomass Power Conversion Technologies using an Eligible Biomass Fuel." Both the Class I and Class II RPS regulations also allow generators that co-fire to qualify as RPS Renewable Generation as long as certain requirements are met. This provision in the RPS program is analogous to the biomass exemption from carbon dioxide emissions accounting in the RGGI program.

In 2008, the Massachusetts Green Communities Act established new Renewable and Alternative Energy Portfolio Standards (RPS and APS) allowing Combined Heat and Power facilities to be included as an eligible technology, provided the thermal output of a CHP unit is used in Massachusetts. APS eligible CHP units receive credits for the useful thermal energy of a CHP unit delivered to Massachusetts end-uses, subject to the formula included in the regulations. The DOER rules issued for this program will, for the first time in the Commonwealth, promote the installation of new CHP units for residential, commercial, industrial, and institutional applications.

A central component of the Massachusetts RPS program is the issuance of Renewable Energy Credits (REC's) for biomass-fueled electric power generation, providing a significant incentive and market driver for large-scale biomass electric power generation. While the market price for REC's varies significantly based on state RPS requirements, the available pool of qualifying renewable energy sources, and overall demand for electricity, they are a very significant factor in the economics of biomass power generation and a significant factor in negotiating Power Purchase Agreements. The current market price for REC's is between \$20–\$40 per MWh and the average monthly price for electricity in the ISO New England region from March 2003—February 2010 is \$62/MWh (ISO New England, 2010). At these rates (which have been even higher in past years with REC's bringing up to \$50/MWh) REC's are clearly a major, though variable, factor in a biomass power plant's return on investment.

1.4.2 Massachusetts RGGI Implementation

As a member of the Regional Greenhouse Gas Initiative (RGGI), Massachusetts has agreed with ten other states to cap carbon dioxide emissions from large (i.e. > 25 MWe) fossil fuel-fired electric power plants in the ten-state region, and to lower this cap over time. Each individual state has adopted regulations to create allowances corresponding to their share of the cap, and to implement accounting, trading, and monitoring regulations necessary to control emissions. Any allowance can be used for compliance with any state's RGGI regulation. The RGGI Model Rule provides a template on which all state regulations are based.

The RGGI Model Rule includes three provisions related to the combustion of biomass fuels. The first exempts facilities whose fuel composition is 95% or greater biomass from the program. The second allows projects that achieve emissions reductions by switching to certain biomass-derived fuels for heating to apply to create offset allowances. The third applies to regulated facilities that co-fire biomass fuels with fossil fuels, or switch completely from fossil to biomass fuel. In such cases, emissions that result from the combustion of "eligible biomass" fuels are not counted toward compliance obligations. Massachusetts' RGGI regulation includes all three of these provisions, but no power plant or offset project in the state has yet applied to take advantage of the co-firing or offset provisions. The definition of below is from Massachusetts' RGGI regulation:

> ***Eligible biomass.*** *Eligible biomass includes sustainably harvested woody and herbaceous fuel sources that are available on a renewable or recurring basis (excluding old-growth timber), including dedicated energy crops and trees, agricultural food and feed crop residues, aquatic plants, unadulterated wood and wood residues, animal wastes, other clean organic wastes not mixed with other solid wastes, and biogas derived from such fuel sources. Liquid biofuels do not qualify as eligible biomass. Sustainably harvested shall be determined by the Department [of Environmental Protection].*

In addition to the complete exemption from the RGGI system for generators whose fuel composition is 95 percent or greater biomass, the RGGI Model Rule and all participating states except for Maine and Vermont provide partial exemptions for facilities that co-fire with smaller percentages of biomass. This partial exemption provides that any carbon dioxide emissions attributable to "eligible biomass" may be deducted from a facility's total carbon dioxide emissions when calculating whether the facility's emissions are within its carbon-allowance budget.

Regarding the impact of the Regional Greenhouse Gas Initiative (RGGI) as an incentive for biomass electric power generation, since RGGI defines biomass power as carbon neutral and exempt from participation in the carbon allowance program and categorically excludes biomass power from allowable offsets qualifying for carbon allowances, biomass energy receives no direct incentives through the carbon allowance auction program central to RGGI implementation. It might be incentivized, however, through state investments in clean energy from auction revenues allocated to consumer benefit and renewable energy and efficiency programs. In Massachusetts, these revenues are allocated to five uses, as follows, based on the recently passed 2008 Green Communities Act: promotion of energy efficiency and demand response (minimum of 80% of revenue); reimbursement of municipalities in which tax receipts decrease due to RGGI (limited to 3 years); green communities (not to exceed $10 million per year); zero-interest loans to some municipalities for efficiency projects; and, state administration of the cap and trade program (Green Communities Act, 2008).

In terms of the impact of the RGGI program on the development of biomass generating facilities, should auction prices rise sufficiently, they could provide an incentive for generating facilities to switch to biomass as a power source, or for the construction of new biomass-fired power plants. However, at current allowance prices of approximately $2 per ton of carbon dioxide, there is insufficient price pressure to incentivize such a shift at this time (RGGI, Inc, 2010).

A summary of the range of statutory and regulatory provisions that directly address biomass in Massachusetts, with an emphasis on biomass policy within the electricity sector, is included in Appendix 1-A to this report.

1.5 BIOMASS ENERGY POLICIES IN OTHER STATES

Based on a review of eleven states' policies regarding biomass (Arizona, California, Connecticut, Maryland, Minnesota, Missouri, Oregon, Pennsylvania, Vermont, Washington, and Wisconsin), the thrust of state policies promoting biomass and/or biofuels is focused on electric generation and less so on transportation and thermal. All surveyed states have numerous policies, programs and/or incentives to promote electric generation from renewable sources of energy, including biomass. A few states have policies to support the use of biomass/biofuels for transportation (California, Minnesota, Oregon, Pennsylvania, Washington, and Wisconsin) and/or for thermal production (Arizona, Connecticut, Missouri, Oregon, Pennsylvania, Vermont, Washington, and Wisconsin).

Typically, states include biomass as one of a number of sources of renewable energy in a variety of policies and programs aimed at increasing electric generation from renewable energy such as renewable portfolio standards. Other common state policies supportive of biomass electric generation are net metering programs; public benefits funds; other grant and/or loan programs; power purchasing programs at the state and/or local level; and a variety of tax incentives. [6]

States with large sources of biomass supply—Minnesota, Missouri, Oregon, Washington and Wisconsin—also tend to have biomass-specific policies or programs in addition to general programs such as renewable portfolio standards. These states are also likely to have biomass working groups or a biomass program (Connecticut, Minnesota, Oregon, Pennsylvania, and Vermont). Some have produced biomass reports, including woody biomass supply assessments. (Arizona, California, Minnesota, Oregon, Vermont, Washington, and Wisconsin). These reports typically focus more on biomass promotion and less on sustainability, and some discuss the linkage between biomass utilization and climate change. Finally, some states have produced woody biomass harvesting guidelines that focus on best management practices for harvesting woody biomass in an ecologically sensitive and sustainable manner (Minnesota, Missouri, Pennsylvania, and Wisconsin). All such harvesting guidelines are voluntary guidance only.

1.6 OVERALL STATE AND FEDERAL POLICY DRIVERS FOR BIOMASS POWER IN MASSACHUSETTS

While conclusive data on the cumulative amounts and impacts of the suite of state and federal policies relevant to biomass power are not available, interviews with plant managers and experts in the field of electric power regulation and development[7] and analyses of federal subsidies indicate that, generally, the most important federal subsidy is the Production Tax credit ($10 per MWh) and most important state incentives are Renewable Portfolio Standards and the related sale of Renewable Energy Credits (currently $25–$35 per MWh). While the value of a REC is higher, the price varies significantly in the marketplace with the cycling of RPS requirements, emergence of new technologies, construction of new renewable energy facilities, the state of the economy and demand for electric power. While less valuable at only $10/MWh, the federal PTC is a more stable source of income for biomass plants over time.

Overall, the economics of individual biomass power plants are determined by the Power Purchase Agreement (PPA), which defines a long-term contract for the purchase of power from a generating facility to utilities or other buyers in the electric power market. PPA's include some or all of the power produced by the generating facility and can also include some or all of the REC's held by a facility in long term contracts. Overall, banks and other investors need confidence in a credible investment stream stemming from a contract including an adequate price (for power and

6 For a description of the range of tax incentive programs, see the public policy program appendix to this report

7 Synapse Energy Economics, Cambridge, Massachusetts; Innovative Natural Resource Solutions, Portland, ME; Mc Neill Generating Station, Burlington VT; Schiller Station, Portsmouth, NH; Ryegate Power Station, East Ryegate, VT.

possibly REC's) over a sufficiently long period of time to satisfy the debt service for the facility. It is worth noting that only one new biomass power plant has been built in the region since the advent of REC's (Schiller) and that RECs are considered to be an important feature in its financial picture.

After the Power Purchase Agreement, the second largest cost variable involved in the finances of a biomass power plant is fuel supply and pricing. For example, the Ryegate plant in Vermont and Schiller plant in New Hampshire spend between 60% and 70% of their operating costs on fuel purchases and generally, costs in excess of $30–$35 per ton are considered the maximums if biomass power is to remain competitive with other fossil fuel capacity.[8] Given the relative importance of fuel purchases on operating costs, BCAP payments could play a significant role in incentivizing power plants over other non-energy biomass uses in Massachusetts if a continued high level of subsidy to suppliers of biomass to qualifying electric generation facilities lowers fuel supply costs for the power sector. However, given current Congressional review of the BCAP program and the USDA rulemaking process, it is considered unlikely that current levels of subsidies will continue.

Regarding relative incentives for the construction and location of biomass power plants in Massachusetts versus other New England states, it does not appear that there are significant subsidies or incentives in existing public policy that make Massachusetts more or less likely to attract new biomass power plant proposals. While Massachusetts does have a strong market for REC's due to their well-established and aggressive RPS program, this does not provide any particular incentive for building qualifying plants in Massachusetts versus surrounding states. Furthermore, Massachusetts is not unique in having a number of current biomass power plant proposals. Vermont currently has 5 to 8 proposals in varying stages of discussion; New Hampshire has two major projects that have come and gone over the past few years; etc.[9] To further illustrate the scale and scope of biomass power plant proposals across the region, over the past ten years, there have been 243 biomass power plant proposals in the ISO New England region, with only one new plant constructed Schiller Power Plant, NH).

Overall, federal and state policies and incentives are responsible for the trend within the biomass industry to propose large-scale electric generation facilities in Massachusetts and elsewhere in the country.

8 $30–$35 per ton for wood purchase is the breaking point as reported in interviews with the Ryegate and Schiller power plants and is also consistent with independent research conducted by the Biomass Energy Resource Center.

9 Recent Vermont biomass power plant proposals include: 20–25 MW plant in Ludlow, 20–30 MW plant in Rutland, two 20–30MW combined pellet mill/biomass plants in Pownal and Fair Haven, 20–30MW plant in North Springfield. Recent New Hampshire biomass power plant proposals include: 70MW power plant in Berlin, 50–70 MW power plant (in combination with a celluslosic ethanol plant) in Groveton.

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CHAPTER 2
TECHNOLOGY PATHWAYS

2.1 INTRODUCTION TO TECHNOLOGY OPTIONS

Biomass in various forms can be used for a range of energy options, through a variety of technologies, to achieve various end purposes. In this chapter, we are looking at several pathways to give the reader an understanding of this range, but also to inform and model potential demand for fuel supply in the future (Chapter 3), and to understand the carbon implications for these choices (Chapter 6). This assessment looks exclusively at the use of existing low-grade forest resources in Massachusetts and surrounding counties in neighboring states, as opposed to agricultural crops or residues or plantation trees and crops which can also provide biomass for energy. Sources of non-forest based biomass, such as wood waste from construction debris, or other sources sometimes considered as biomass, such as municipal waste, were not considered.

With respect to the forest's low-grade wood resource potentially used for energy, the end products can be solid—such as cordwood, wood chips, or wood pellets—liquid, such as pyrolysis oil or cellulosic ethanol, or gas—synthetic or producer gas made through "gasification" and "bio-char" technologies. Finally, the end uses can range from residential to industrial applications, and fall into three general categories: electricity power production, thermal applications for heating (and cooling), or emerging technologies such as cellulosic ethanol or gasification. Between the first two categories, is combined heat and power (CHP), which in turn can be thermally led (optimizing heat production with some electricity produced) or electricity-led (sizing the plant for optimal electricity production and using some of the heat).

Some of these technologies and applications are well established and have been in place for years and others are pre-commercial or still under development. In the sections that follow, we describe two main currently available applications for electricity and thermal production, with CHP discussed in a subsequent section. This discussion focuses on those technologies and applications that are already well established, or are technologically available in the immediate future should policies wish to guide additional biomass in these directions. These are the applications most likely to place demands on Massachusetts' forest resources in the short term. Still, because of the amount of federal investment for research and development in some of the emerging technologies, which, if realized, have the potential to significantly affect demand for forest resources (such as cellulosic ethanol), a third category of applications is discussed in Section 2.5, entitled "Emerging Technologies." All of the liquid biofuels options for producing transportation fuels fall into this category, as does gasification and bio-char production.

Among these application areas, we selected 12 technology pathways to describe how biomass might be used, and compared them to their six fossil fuel equivalent applications. These are described in Appendix 2-A, and summarized in Appendix 2-B.

2.2 ELECTRICITY GENERATION

2.2.1 CURRENT SOURCES OF ELECTRICAL SUPPLY

Massachusetts uses about 55.8 million Megawatt hours (MWH) of electricity (Energy Information Administration—EIA, 2010) and produces about 47.1 million MWH (EIA, 2007). Massachusetts is a member of ISO New England, which is responsible for wheeling power throughout the region and bringing in power from other regions as needed. Of the power the state produces, renewables account for about two million MWH (4.3 percent), with biomass power generation accounting for 119,000 MWH, or six percent of the renewable portfolio and 0.3 percent of total production (EIA, 2007). Ten natural gas-fired power plants are now the state's leading power producers, accounting for over half of net generation. Coal, primarily from Colorado and West Virginia, is the state's second leading generation fuel; it is used in four plants and accounts for about 25 percent of net electricity production. Massachusetts also uses oil-fired systems (seven existing plants—although oil has been increasingly replaced by natural gas over the past decade) and nuclear from the Pilgrim plant to round out the remaining percentages of its profile. Of the renewables, landfill gas is the largest contributor, accounting for about 1.1 million MW followed by hydroelectric generation at 797,000 MWH (EIA, 2010).

The nuclear facility, all of the fossil fuel based power, and solid-fuel biomass power plants all use steam turbine technology, which has the common attribute of being approximately 25 to 32 percent efficient at converting the energy value of the fuel to electricity. Unused heat in these systems is released through cooling towers, or through heat exchanged in Cape Cod Bay in the case of the Pilgrim Nuclear facility (Entergy, 2008). The four coal facilities use 382,000 tons of coal each year (EIA, 2007), and the wood facilities[1], at full operation, would use approximately 215,000 green tons annually (INRS, 2007).

2.2.2 ELECTRICAL GENERATION PATHWAYS

Pathways 1–4 describe the range of power facilities used now, and for the foreseeable future, to produce electricity. Pathway #1 assumes a 50 MW biomass powered facility, and enables comparison to two fossil fuel options for coal (Pathway #3) and natural gas (Pathway #4) as well as a co-firing option where wood is substituted for 20 percent of the coal at a coal-fired unit (Pathway #2).

All pathways assume advanced pollution controls as needed to ensure the units are performing to meet expected pollution control objectives, but the efficiency is an average based on present performance of units in use today. Generally, this is 32 percent for coal, 20–25 percent for woody biomass, and 33 percent for natural gas (Appendix 2-B).

[1] There are two wood-fired electrical facilities in Massachusetts: Pinetree-Fitchburg (14 MW) which is operating and Ware Co-Gen (8.6 MW) which is idle (INRS at 40).

The following chart (Exhibit 2-1) presents the CO_2 emissions for the four electrical generation pathways.

Exhibit 2-1
Electrical Generation Pathway CO_2 Emissions

Electrical Generation Pathway	CO_2 Emissions (lbs/MMBtu)	CO_2 Emissions (lbs/MWH)
Coal (Pathway #3)	642	2,189
Modern Woody Biomass (Pathway #1)	863	2,945
Natural Gas (Pathway #4)	355	1,211
Co-firing with 20% wood* (Pathway #2)	684	2,334

*Total emissions for coal and wood combined.

These pathways are used to evaluate and compare different scenarios for forest management and carbon impacts if policies are directing biomass use toward stand-alone electrical generation, and to enable comparison to the most likely fossil fuel alternatives. Of all the fuels considered, natural gas is the cleanest and the lowest carbon emitting due to its ability to generate power using a direct combustion turbine at higher efficiency than traditional steam turbine technologies, and the fact that it has less carbon per unit of energy.

2.3 THERMAL PRODUCTION

Roughly one-third of the nation's energy demands are thermal demands for heat, hot water, cooling, and industrial process heat (EIA,2008),. In the Northeast, this percentage is even higher, with the region using 82 percent of the nation's home heating oil (EIA, 2009). In Massachusetts, 42 percent of the households and businesses use #2 heating oil or propane as their primary source of heat (EIA, 2007).

At the residential and community scale, biomass can be an effective means of using local wood resources and displacing fossil fuels efficiently. Generally, these thermal systems are between 75 percent and 85 percent efficient (See Appendix 2-B).

2.3.1 CURRENT SOURCES OF THERMAL SUPPLY

2.3.1.1 Residential Biomass Forms and Uses

Biomass has been used to heat homes for millennia. The amount of biomass used to heat Massachusetts' homes is not known, but is estimated at between one and two million green tons annually (Personal Communications, MADOER, 2010). Residential applications use biomass in fireplaces; wood stoves, furnaces, and boilers[2]; pellet stoves furnaces and boilers; and outdoor wood boilers. These applications decrease in efficiency (California Air Resources Board-CARB, 2005) and increase in emissions as one moves from pellet stoves and boilers to wood stoves and boilers to outdoor wood boilers to fireplaces.

Exhibit 2-2 presents efficiency, particulate, and CO2 emissions associated with these residential applications.

Exhibit 2-2
Residential Wood Appliance Efficiencies and Emissions

Wood Appliance	Efficiency	Particulate Emissions	CO_2 Emissions (lbs/MMBtu)
Masonry Fireplace	-10% to 10%	50 g/hr	2,157.0
Outdoor Wood Boilers	28% to 55%	55 g/hr to 143 g/hr (Pre-2007) 15 g/hr (Post-2007, Voluntary)	359.5
Fireplace Insert	35% to 50%	.94 to 3.9 g/kg	507.5
Airtight Stove	40% to 50%	10-20 g/hr (estimate based on Cert .3 of old wood stoves)	479.3
EPA-Certified Stoves and Inserts	60% to 80%	2.5 to 7.5 g/hr (EPA, 2/22/10)	317.2
Residential Pellet Stoves	75% to 90%	<1 to 2 g/hr (EPA, 2/22/10)	269.6
Residential Pellet Boilers	80% to 90%	<1 to 2 g/hr (EPA, 2/22/10)	239.7

2.3.1.2 Institutional Biomass Forms and Uses

Use of biomass for heat and hot water in community buildings, institutions, etc. has had limited application in Massachusetts. Two examples are: Quabbin Reservoir Administrative Building in Belchertown, and Mount Wachusett Community College in Gardner. The Quabbin system was installed in 2008 and uses 350 tons of wood per year to displace 22,000 gallons of #2 heating oil (Biomass Energy Resource Center-BERC, 2010). It is 2.0 MMBtu/hr in size. The Mount Wachusett system is 8.0 MMBtu/hr in size, was installed in 2002 and uses between 1,200 and 1,400 tons of wood each year (BERC, 2010). This system replaced electric heating, and the college estimates it has saved 30 million kWh of electricity in the eight years of operation (BERC, 2010). The technology for these systems uses centralized hot water-based boilers and underground insulated pipe distribution systems.

Other applications of this scale of system are used in several schools. Several colleges are considering conversion to biomass, including UMASS Amherst, and the VA hospital in Northampton.

2.3.2 THERMAL PRODUCTION PATHWAYS

Pathways 5–10 describe the range of applications that may be used for thermal production, beginning with cordwood systems that would serve a typical home (Pathways #5 and #6). These boilers represent small systems that, at 100,000 Btu/hr, would be used to serve a small business or residence. The difference between these two pathways is that Pathway #6 represents an EPA-certified boiler that is more efficient and therefore has fewer carbon emissions per energy output than Pathway #5.

Pathway #7 describes a pellet system, separated into two parts in order to compare effectively with other sources of thermal energy presented—pellet manufacturing is Pathway 7A and covers the process of using green wood chips to produce pellets, and Pathway 7B describes the use of these pellets in a typical commercial or institutional setting, sized at 5.0 MMBtu/hr. When considering pellets and comparing to other fuels with respect to harvesting needs and carbon impacts, it is important to consider both pathways.

[2] A stove is considered to be a stand-alone space-heating device, a furnace is a central hot air system, and a boiler is a central hydronic (hot water pipe and radiator) system.

Pathway #8 is a wood chip system sized at 50 MMBtu/hr, which would serve a community in a district energy system of the kind commonly used in Europe. Pathways #9 and #10 provide information about the fossil fuel equivalent versions of this system, using #6 heating oil and natural gas, respectively.

Exhibit 2-3 presents the CO_2 emissions from these thermal pathways[3]:

Exhibit 2-3 Thermal Generation Pathway CO_2 Emissions	
Thermal Generation Pathway	**CO_2 Emissions (lbs/MMBtu)**
Wood chip-fired District Energy (Pathway #8)	288
Non EPA-Certified Residential Wood Boiler (Pathway #5)	360
#6 Heating Oil (Pathway #9)	217
Natural Gas (Pathway #10)	138

2.4 COMBINED HEAT AND POWER OPTIONS

All electrical production from combustion of fuels creates excess heat that is often wasted. In the case of power plants, excess heat is often released through cooling towers, as steam from the turbine is condensed and returns to the boiler. Combined heat and power systems (CHP) seek to utilize some or all of this excess heat. As this excess heat is made into useful energy, the efficiency of the generating system increases with the proportion of heat it uses. Generally, using conventional technology, for each unit of electricity produced, three units of thermal energy are released.

Electricity-led CHP is an option where power production is near a thermal demand. A 20 MW power plant produces enough heat to heat approximately 1,100 homes[4]. However, to date, the economics, incentives and siting preferences have not resulted in power plants choosing this route. As a result, regardless of the fuel source producing the electricity, approximately 75 percent of the energy value of the fuel has been wasted as lost heat. Taking advantage of this energy value requires planning, intentional siting, and either financial or regulatory incentives that promote power producers deciding to increase the complexity of their systems by the addition of steam or hot water as a salable output. This is not the business model that has been pursued to date. Recently, with the increased understanding of efficiency and concern about efficient use of resources, biomass power facilities are beginning to incorporate some CHP in their proposals, though because of the large amount of heat available relative to potential nearby uses, these projects often make use of only a small percentage of the available heat (10–15 percent).

[3] As with the other exhibits which follow, the source of data for these charts is presented in Appendix 2-B

[4] 20 MW electric produces approximately 136 MMBtu/hr of heat. Residential heating typically uses 40 Btu's/sq ft. Based on a 3,000 square foot house, heating requirement is 120,000 Btu's/hr, or 1,137 homes.

Thermally led CHP maximizes the demand for heat, but produces relatively little electricity. At the community scale, a typical CHP facility might produce 1–5 MW of electricity while heating a college campus or small community district of 200–500 homes and businesses.

An important point to note is that the efficient scale of producing electricity alone leads to plants in the 20–50 MW size range. At this scale, it is more cost-effective to produce the power, and any CHP component is a complicating factor that tends to reduce the overall cost-effectiveness of the project under current policies. At smaller scale thermal-led CHP systems, the opposite is true—production of heat alone maximizes cost-effectiveness of the project, and adding an electrical component reduced the overall economics of the project, i.e. the savings in heat help subsidize the electrical generation components.

Conventional technology requires the production of steam to produce electricity, but European commercial technologies include gasification where the produced gas is combusted directly in a combustion turbine, or Organic Rankine Cycle (ORC) thermal oil technology which uses a thermal oil to gain temperature gradients necessary to produce electricity without steam, so that the thermal system can be designed around hot water, and at low pressure. The ORC system, while more easily incorporated into a hot-water based thermal application and therefore of greater potential in smaller CHP systems (see below), the ORC process is still only approximately 20% efficient on its own in the production of electricity, but would be expected to be between 75% and 85% efficient in heat-led applications. Heat-led gasification can be expected to be approximately 75% efficient. (See Appendix 2-B for sources of efficiency information).

2.4.1 CHP PATHWAYS

Pathways #11 and #12 describe moderate-sized CHP systems capable of producing 5.0 MW of electricity. The first uses conventional technology, producing steam to run a turbine, and fully utilizes the 34 MMBtu/hr of heat generated to heat facilities on the order of magnitude of a college campus, a hospital, or small community. As such, the overall efficiency is rated at 75 percent. The second pathway uses gasification technology, which is just an emerging technology here in the United States. Still, there is an example of a commercial system operating since 2000 in the Town of Harboøre, Jutland, Denmark that produces 1.6 MW of electricity and heats 900 homes (BERC, 2010). The efficiency rating for this system is also 75 percent.

Pathways #13 and #14 are the fossil fueled equivalent of the biomass CHP systems for oil and natural gas.

Exhibit 2-4 below presents CO_2 emissions for the four CHP pathways considered.

Exhibit 2-4 CHP Generation Pathway CO_2 Emissions	
CHP Generation Pathway	**CO_2 Emissions (lbs/MMBtu)**
Wood chip Steam System (Pathway #13)	287
Wood chip Gasifier (Pathway #14)	287
Oil System (Pathway #15)	232
Natural Gas System (Pathway #16)	146

2.5 EMERGING TECHNOLOGIES

There are several emerging technologies for using biomass that have the potential to change the demand for low-grade wood over time. Most of these are transportation sector related. The US Department of Energy has invested hundreds of millions of dollars over the last decade to augment the ethanol production of agricultural crops (corn primarily) with ethanol derived from woody-biomass sources (cellulosic ethanol). To date, they have sponsored both research and development, funding six pilot scale plants throughout the country. While not yet commercially viable, our transportation fuel demands are so high and this is another area, like heating oil, directly related to our importation of fossil fuels, that the issue is an important one to consider in the context of making policies to support the sustainable use of the low-grade wood resource. To put it in context, the Range Fuels plant near Soperton, Georgia will begin at pilot scale producing 20 million gallons of cellulosic ethanol a year, using 250,000 tons of wood. At its commercial scale of 100 million gallons per year, the wood demand will be over 1.2 million tons of green wood per year for this one plant (Range Fuels, 2010).

Smaller scale work in bio-oil (pyrolysis oil) and bio-char (torrefaction) are emerging technologies that can help with both transportation fuel alternatives to gasoline and diesel, as well as, in the case of bio-char, potentially sequester portions of the wood carbon for long periods of time (Laird, 2008). These systems are operational at very small scales at the moment, but have a potential to contribute positively to the biofuel equation.

There are other technologies of similar scale to the bio-oil that use biomass to produce a range of products, including fertilizers, plastics, and glues. All of these products are relatively limited in demand, so source material from forests will not be significant relative to energy demands or other forest product uses.

2.5.1 EMERGING TECHNOLOGY PATHWAYS

The emerging technologies represented here all use some of the heat for other aspects of their processes, so their efficiencies are generally in the 40–45 percent range. Pathway #15 provides an example of a commercial-scale cellulosic ethanol plant, making 100 million gallons of cellulosic ethanol per year. In this process, the cellulose in the wood is converted to sugars that are fermented into alcohol. The lignin part of the wood is combusted directly to produce steam and electricity. Pathway #18 is a variation on this whereby the by-product of pyrolysis is used to produce other products, such as plastics, glues, organic fertilizers, and fuel additives instead of electricity. Pathway #16 represents a bio-oil and bio-char system, producing 15 million gallons/year of bio-oil, and approximately 21,575 tons of bio-char (charcoal), having heating value of 11,000 btu/lb (dry basis), that can be used as a soil amendment for carbon storage. Pathway #17 is of similar size, producing a syngas that is used to make liquid fuels, with lignin used to produce steam-based electricity. The following chart summarizes the CO_2 implications of these pathways:

Exhibit 2-5
Emerging Technology Pathway CO_2 Emissions

Emerging Technologies Pathway	CO_2 Emissions (lbs/MMBtu)
Cellulosic Ethanol (Pathways #15 and #18)	255
Bio-products Pathway (Pathways #16 and 17)	119

2.6 GENERAL DISCUSSION AND SUMMARY

2.6.1 THE FUTURE ROLE OF BIOMASS UNDER PRESENT POLICIES

Electricity demand is expected to increase by approximately 1.2 percent annually, with a peak demand increase of 1.3 percent due to increased cooling demand in the summer (ISO New England Inc., 2009). Air pollution goals, as well as cost and projected supplies, will continue to drive new power production toward natural gas, but for the state's RPS. In an attempt to reach 15 percent by 2020, Massachusetts is looking to alternatives to fossil fuels to reach its goals. There are several significant wind projects in place and in planning, as well as solar projects, but as biomass power is "base load," the trend has been to look to it to supply an increased share of the electricity portfolio.

Over the next five to 10 years, barring a change in policy or incentives, or a dramatic change in the price of fossil fuel or electricity, we would expect the current pattern of incremental proposal and construction of stand-alone biomass power plants between 20 MW and 50 MW to continue to be the major focus of the use of biomass. As described elsewhere, the pattern has been for many to be proposed (214 throughout New England over the past decade, with one constructed), and there are currently four proposals in Massachusetts. In part, the low ratio of "proposed" to "constructed" reflects the marginal economics of constructing plants based on the present cost of electricity, and the desire for investors to recoup costs of capital investment within a relatively short period of time—most private investors look for a return on investment of 20 percent within two to five years[5].

Events that can speed this up are if the wholesale rates of electricity increase substantially while the policy direction for renewables is maintained. In 2008, Massachusetts paid an average of 16.27 cents/kWh retail for electricity, the fourth highest in the nation and highest in New England. It is doubtful that electricity prices will increase dramatically in the face of the downward regional and nationwide pressure on prices. If Renewable Electricity Credits (REC's) rise in value and are stabile over a period of several years, this too would encourage construction of more power plants.

[5] It also reflects the tendency for proposers to announce projects at a very early stage of project development as a relatively easy means of assessing public acceptance of a given project, so the public announcements are not a good gauge of projects that are truly in advanced development and are likely to be built.

Factors that can make power plant investment slow down are low value of REC's coupled with only an inflationary increase in the price of electricity. Also, if the availability of fuel supply is restricted, or if it is only available at a cost higher than what plants can afford to pay, biomass power will be discouraged. We consider this scenario to be possible, but unlikely in the immediate future.

While incentives and policies may promote biomass electric plant construction, the pace and penetration of biomass power plants are controlled most significantly by the fuel supply; it is such a large portion of the cost of operations that it is looked at very carefully by investors. This is why multiple proposals may be vetted at a given time, but if one is built, the others in the wood-basket are significantly adversely affected and are less likely to go forward. If there are reasonable harvesting and procurement standards in place regarding overall sustainability, this factor is likely to increase the due diligence on available fuel supply and prevent over-development of biomass power facilities.

If policies are changed to require CHP or a minimum annual net efficiency standard, as some states have done in certain circumstances and as DOE encouraged in recent procurements, more CHP can be expected. But under current conditions, siting constraints, the required scale for economically viable power production and lack of large centralized demand for thermal at the scale produced by a 20–50 MW power plant will all limit the desirability of power developers to include heat, as well as the amount of heat that can be effectively used by an electricity-led CHP system. We do not see electricity-led CHP as growing in the absence of policies or incentives to encourage that direction.

Residential conversions are very dependent on oil and propane prices. In the absence of policies that would encourage large-scale switchover to biomass in residences, such as a substantial increase in the residential tax credit, or a change in building codes or insurance standards (to not require a conventional fossil fuel-based system in the home), the trend is expected to remain about the same. Although the use of biomass for home heating is significant, and currently not well-quantified, dramatic changes in the trend are not expected, though as explained below, residences can react quickly to rapid oil and propane price increases.

At this scale, residential use will not be a significant driver in determining Massachusetts' forest resource capacity for increased biomass use or the overall sustainability of the resource. Accordingly, the analyses in subsequent sections of this report assume residential use (and all existing uses for that matter) remains about the same as they are. That said, things which weigh in on people's decisions to burn wood in the home primarily relate to cost of the fossil fuel alternative, and while this consideration may be at the forefront individual preferences regarding energy security and price stability, ease of operation and maintenance, degree of automation and convenience, cleanliness, availability of the wood fuel, heating effectiveness and comfort all play a role. Other factors such as emissions, environmental benefit, energy independence, space, and cumulative impacts are of lesser importance to the individual decision.

Biomass options in the home most closely able to substitute for oil are pellet boiler and furnace systems, and these systems are very popular in Europe and increasingly so here. The obstacles preventing large conversion of homes are primarily related to price. A conventional central heating system costs between $2,500 and $4,000 for a typical home. A comparable pellet system would be between $5,000 and $8,500. Even though the fuel is cheaper than oil, its availability in bulk is presently limited, and the cost disparity in systems cannot be made up for by the present 30 percent tax credit that has a cap of $1,500 per home.

If one wishes to promote advanced biomass technologies for the home, incentives such as tax credits, change-out programs, and programs that allow homeowners to offset the additional costs of choosing a biomass system either through credits or ability to finance costs through low or no cost options all work to overcome the cost implications. Proposals are pending in Congress to raise or eliminate the tax credit cap, and to develop a Homestar program that among other things supports pellet system installations. Similarly, New Hampshire and Maine each have programs to encourage an expanded residential market. A reliable bulk delivery option and convenient storage and automated delivery to the boiler or furnace are also necessary for the residential use of pellets to increase significantly and displace oil and propane.

Cordwood use is limited in growth to those capable of handling and tolerating the storage, handling, and messiness of cordwood. Outdoor wood boilers avoid some of the indoor mess of handling cordwood, but the low efficiency and high emissions from them are of increasing concern to states in the Northeast, even when compared to conventional wood stoves. Though they are improving, some of the cost-attractiveness of these systems will be lost as their technology improves.

One hears periodically about home-based CHP systems, but with regard to biomass systems these are not commercially available, and developing products are very expensive relative to either conventional fossil fuel or biomass thermal systems. There are some demonstration projects using a Stirling Engine design, but these are still experimental or unique applications (Obernberger, et. al, 2003). We conclude from this that electrical generation from wood at the residential scale is not commercially available.

With respect to residential heating, it is important to recognize the individual residential component and fuel price sensitivity of the cordwood market when considering net available low-grade wood for sustainable biomass use. Although each homeowner's use is relatively small—perhaps five to 10 tons per season (2-5 cords)—cumulatively, it can be significant, and often the hardest sector to quantify. In Vermont for example, cordwood is estimated to account for between 30 and 40 percent of all biomass use in the state (BERC, 2007). It increased by 20–30 percent in the single season of 2008 when oil approached $150/barrel.

There will also likely be small, incremental increase in thermal applications of biomass at colleges, institutions, and other facilities that have the capital to invest in longer-term payback projects, as the economics are compelling at current or slightly higher than current heating oil prices. These are not going to be common or numerous, as few institutions have the capital to make the change-over, and the payback period of generally between seven and 12 years is too long for private investment interest. To increase thermal applications dramatically, if that is a policy direction Massachusetts wishes to pursue, state and federal incentive programs to provide capital, such as through a revolving loan fund, would be needed .

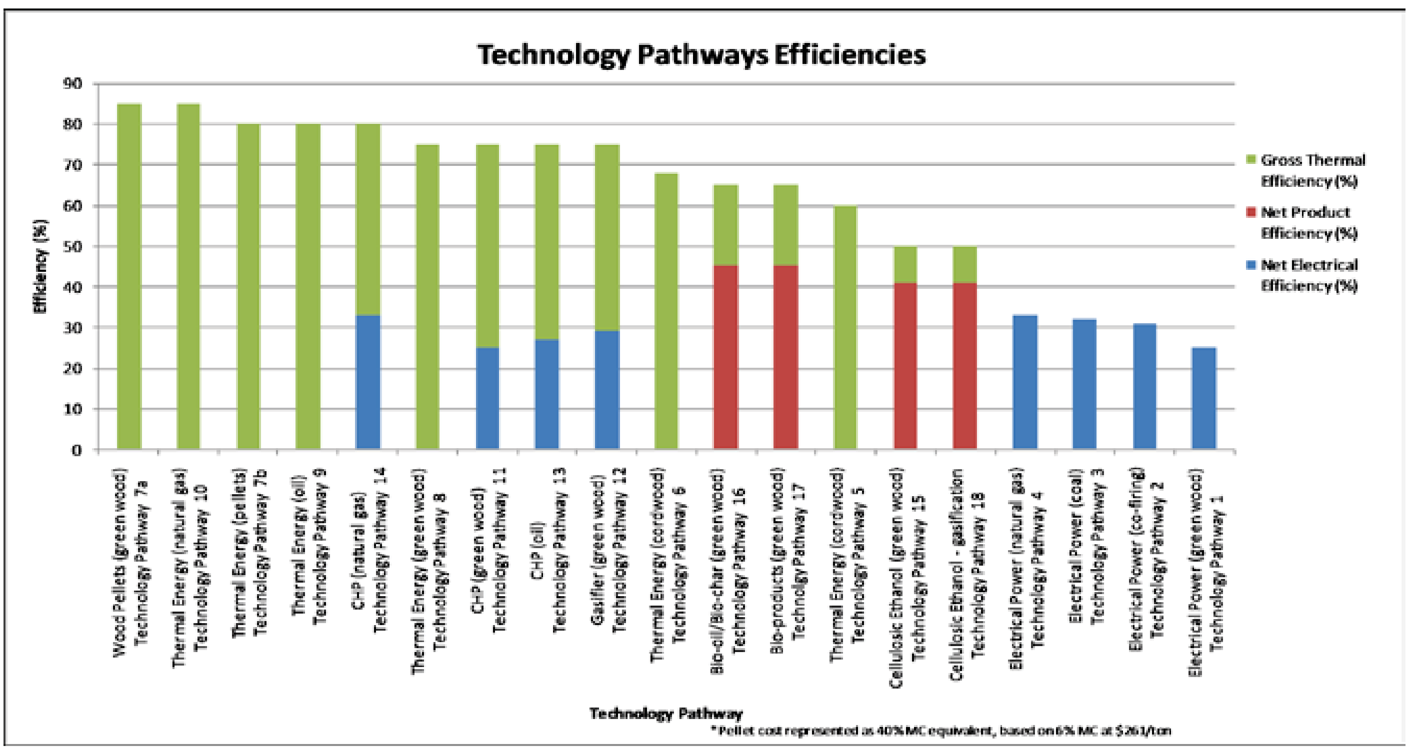
Finally, cellulosic ethanol production has the potential to completely usurp power production at a comparable scale if electricity prices remain low, and oil (gasoline) prices increase markedly. However, the pilot projects under way and supported by the US DOE must prove out, and as such, we consider this scenario to be worthy of watching, but unlikely—especially in the near five to 10 year timeframe.

2.6.2 EFFICIENCY

As has been discussed throughout, converting biomass into different energy pathways and products yields varying ranges of efficiency for extracting the energy value of that biomass resource. Exhibits 2-6 and 2-7 on the following pages show the range of efficiencies for the different applications and pathways selected from most efficient to least efficient.

It is important to recognize that what is presented is just the efficiency of the process to produce energy or fuel or product from the biomass. This does not include up-front processes to get the biomass to the facility, or additional losses incurred through the use of the end product. For example, for electricity, these efficiencies do not include line losses or the efficiency of a given appliance to turn remaining electricity into useful work. Similarly, for the transportation fuels, this does not include the relative inefficient (18 percent) ability of your car to take the energy value of the fuel and convert it into the work of moving you down the road. Finally, for the thermal applications, it does not include the loss of heat exchange from the thermal system to a home, or the efficiency of a home to retain heat. These examples show that further down the process more losses of the energy value of the original biomass will be incurred. They may be smaller or they may be quite large, depending on the end use.

Exhibit 2-6: Graph of Efficiency of 18 Technology Pathway Options[6]



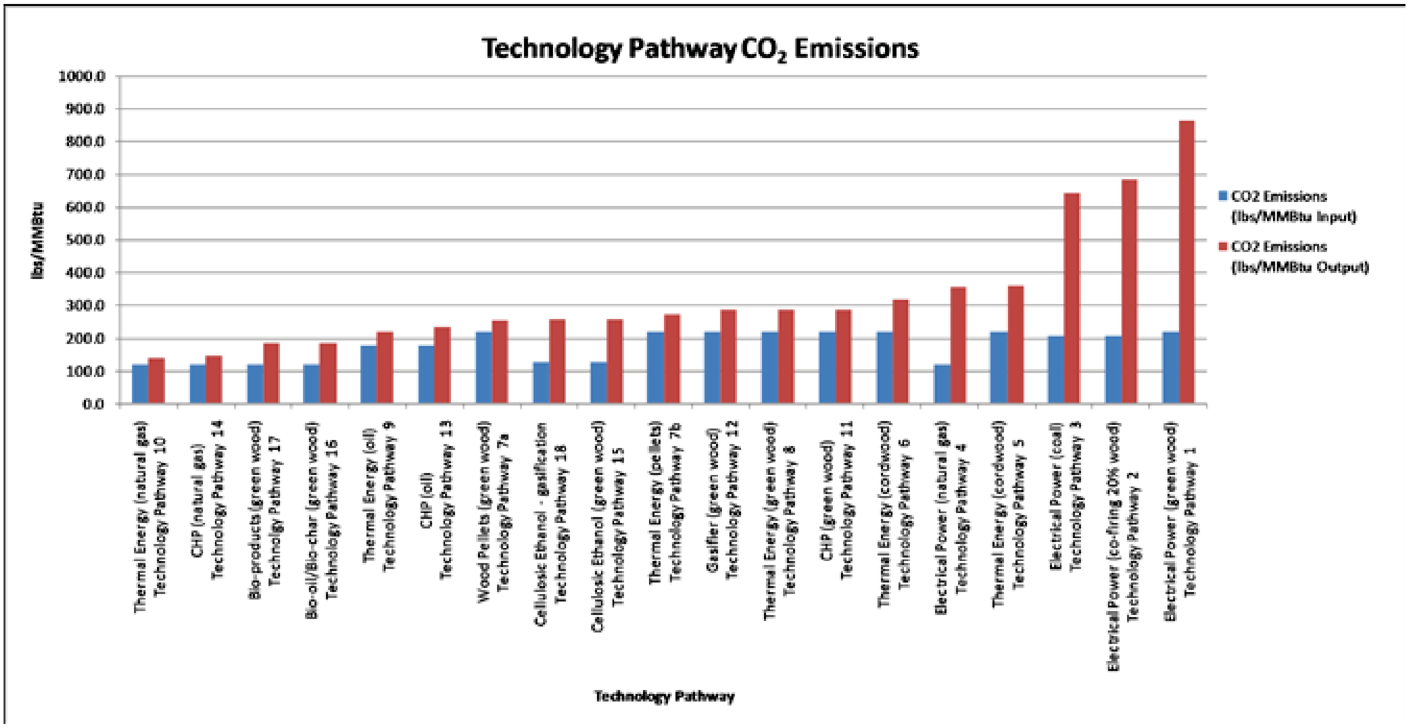
6 Graph information is derived from Appendix 2-B. See that Appendix for data and sources.

Exhibit 2-7: Chart of Efficiency of 18 Technology Pathway Options[7]

Technology Pathway	Net Electrical Efficiency (%)	Net Product Efficiency (%)	Gross Thermal Efficiency (%)
Wood Pellets (green wood) Technology Pathway 7a			85
Thermal Energy (natural gas) Technology Pathway 10			85
Thermal Energy (pellets) Technology Pathway 7b			80
Thermal Energy (oil) Technology Pathway 9			80
CHP (natural gas) Technology Pathway 14	33		47
Thermal Energy (green wood) Technology Pathway 8			75
CHP (green wood) Technology Pathway 11	25		50
CHP (oil) Technology Pathway 13	27		48
Gasifier (green wood) Technology Pathway 12	29		46
Thermal Energy (cordwood) Technology Pathway 6			68
Bio-oil/Bio-char (green wood) Technology Pathway 16		45	20
Bio-products (green wood) Technolgy Pathway 17		45	20
Thermal Energy (cordwood) Technology Pathway 5			60
Cellulosic Ethanol (green wood) Technology Pathway 15		41	9
Cellulosic Ethanol - gasification Technology Pathway 18		41	9
Electrical Power (natural gas) Technology Pathway 4	33		
Electrical Power (coal) Technology Pathway 3	32		
Electrical Power (co-firing) Technology Pathway 2	30.6		
Electrical Power (green wood) Technology Pathway 1	25		

7 Chart information is derived from Appendix 2-B. See that Appendix for sources.

Exhibit 2-8: Graph of CO_2 Emissions of 18 Technology Pathways[11]



2.6.3 CARBON IMPACTS

The CO2 emissions from each of the pathways vary depending on the fuel and the efficiency of the product made. Generally, the CO2 emissions expressed as "input" energy reflect the fuel the process is based on, and the CO2 emissions based on "output" energy reflect the efficiency of the biomass-product conversion, be that electricity, thermal, or fuel. Exhibits 2-8 and 2-9 on the following pages reflect the different pathways from least CO2 emissions based on energy output to the most emitting pathways.

As with the efficiency discussion, it is very important to note this is not a life-cycle analysis of these technology pathways. The carbon aspects of mining coal, harvesting biomass, or drilling and transporting natural gas or oil are not shown here. Nor, except for the electricity and thermal applications, are the emissions of the ultimate use accounted for—that is, the fuels combusted will further release CO_2 associated with that product. While full carbon life-cycle accounting for all pathways is beyond the scope of this work, lifecycle estimates of carbon emissions for the technological options considered in Chapter 6 are provided there.

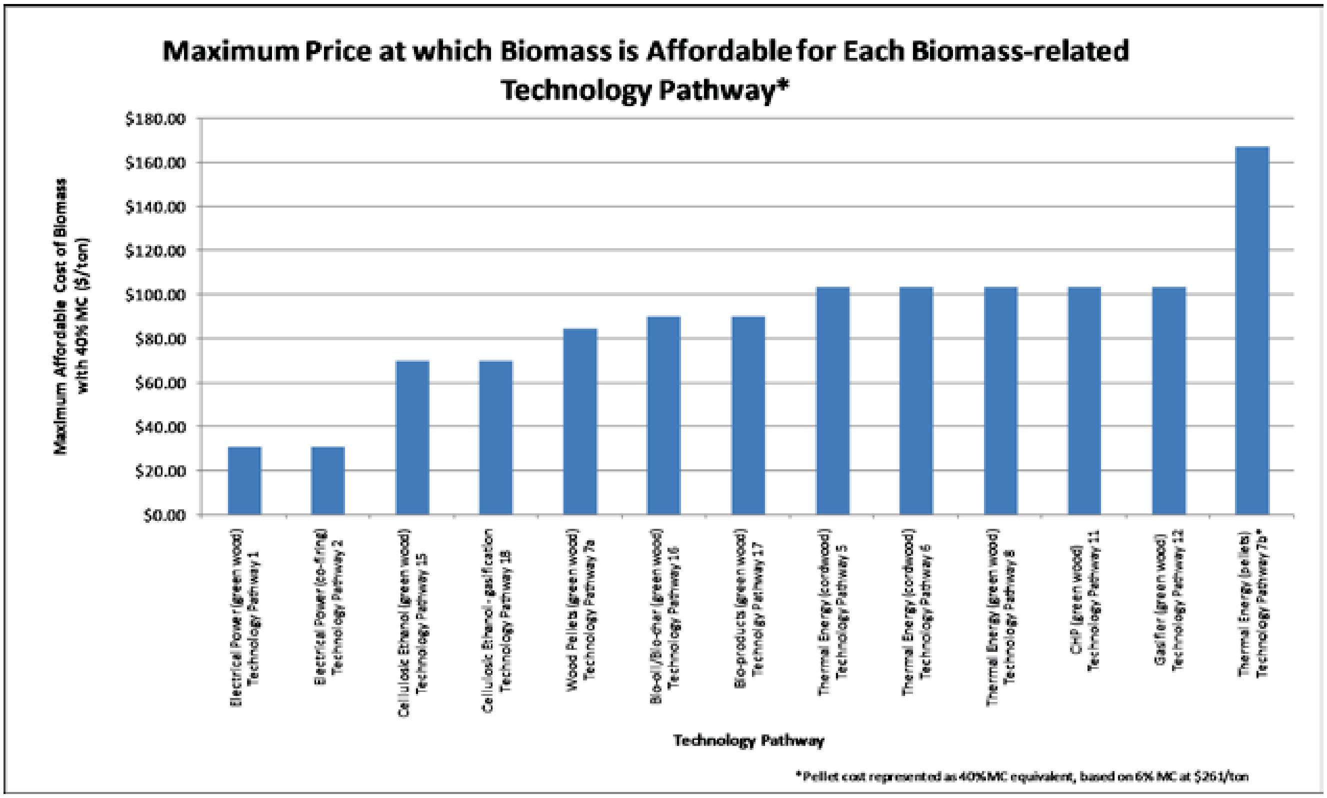
Exhibit 2-9: Chart of CO2 Emissions of 18 Technology Pathways[12]

Technology Pathway	CO_2 Emissions (lbs/MMBtu Input)	CO_2 Emissions (lbs/MMBtu Output)
Thermal Energy (natural gas) Technology Pathway 10	117.0	137.6
CHP (natural gas) Technology Pathway 14	117.0	146.3
Bio-products (green wood) Technolgy Pathway 17	118.6	182.5
Bio-oil/Bio-char (green wood) Technology Pathway 16	118.6	182.5
Thermal Energy (oil) Technology Pathway 9	173.9	217.4
CHP (oil) Technology Pathway 13	173.9	231.9
Wood Pellets (green wood) Technology Pathway 7a	215.7	253.7
Cellulosic Ethanol - gasification Technology Pathway 18	127.3	254.5
Cellulosic Ethanol (green wood) Technology Pathway 15	127.3	254.5
Thermal Energy (pellets) Technology Pathway 7b	215.7	269.6
Gasifier (green wood) Technology Pathway 12	215.7	287.6
Thermal Energy (green wood) Technology Pathway 8	215.7	287.6
CHP (green wood) Technology Pathway 11	215.7	287.6
Thermal Energy (cordwood) Technology Pathway 6	215.7	317.2
Electrical Power (natural gas) Technology Pathway 4	117.0	354.5
Thermal Energy (cordwood) Technology Pathway 5	215.7	359.5
Electrical Power (coal) Technology Pathway 3	205.3	641.6
Electrical Power (co-firing 20% wood) Technology Pathway 2	207.4	684.3
Electrical Power (green wood) Technology Pathway 1	215.7	862.7

8 Graph information is derived from Appendix 2-B. See that Appendix for data and sources.

9 Chart information is derived from Appendix 2-B. See that Appendix for sources.

Exhibit 2-10: (below) Maximum Price at which Biomass is Affordable for Each Biomass-Related Technology Pathway[13]



2.6.4 AFFORDABLE COST FOR BIOMASS SOURCE MATERIAL

Finally, for the purposes of conducting sensitivity analyses of the demand for forest products and how demand might affect cost paid for biomass, and how, in turn, that affects harvesting methods, intensity and options, we have looked at what the maximum affordable price is for each pathway to pay for biomass from the forests. The following Exhibits 2-10 and 2-11 illustrate these prices.

The maximum affordable price for power generation has been calculated based on the wholesale price of 12.5 cents per kWh including REC benefits, the cost of biomass fuel as 33 percent of sale price, higher heating value of wood chips as 17 MMBtu/ton, and moisture content of wood chips as 40 percent. The maximum affordable price for thermal applications has been calculated based on the price of #2 oil as $3 per gallon, higher heating value of 138,000 Btu/gallon, combustion efficiency of 80 percent for oil boiler, affordable price of wood chips as percent of price of oil on $/MMBtu basis as 50 percent and the combustion efficiency of wood chips boiler as 75 percent. The maximum affordable price of wood pellets for thermal energy has been calculated based on e f wood pellets with six percent moisture content as percent of price of oil on $/MMBtu basis as 75 percent and the combustion efficiency of wood pellet boiler at 80 percent. The maximum affordable price of wood chips for manufacturing wood pellets have been calculated based on maximum affordable price of wood pellets for thermal energy at $261 per ton, efficiency of conversion of wood chips to wood pellets as 85 percent, requirements of wood chips per ton of wood pellets as 1.575 tons, and the affordable price of wood chips as 60 percent of the price of wood pellets. The maximum affordable price for other technology pathways has been estimated in proportion of the net efficiencies for the products.

The maximum affordable price is important as the price one is willing and able to pay for biomass determines the type of equipment and treatments that can be applied to the forest, and which uses may get preference over others with respect to biomass product. Higher affordable prices may enable better management, landowner commitment to sustainable forestry, and enhancement of logging infrastructure and methods. The pathways constraining the electricity related biomass prices are based on an electricity wholesale price of 12.5 cents/ kWh, which assumes a wholesale price to the grid plus any value of REC's. Thermal applications are based on a $3.00 per gallon oil equivalent. Obviously, if the price of either goes up, then the ability to pay more for biomass (and still have the project "break even") goes up as well. All of the assumptions for this and the other analyses are shown in the attached Appendix 2-C.

10 Graph information is derived from Appendix 2-B. See that appendix for data and sources. Methodology for calculations is presented in Section 2.6.4.

Exhibit 2-11: Maximum Price at which Biomass is Affordable for Each Biomass-Related Technology Pathway[11]

Technology Pathway	Maximum Affordable Cost of Biomass with 40% MC ($/ton)
Electrical Power (green wood) Technology Pathway 1	$31.00
Electrical Power (co-firing) Technology Pathway 2	$31.00
Cellulosic Ethanol (green wood) Technology Pathway 15	$70.00
Cellulosic Ethanol - gasification Technology Pathway 18	$70.00
Wood Pellets (green wood) Technology Pathway 7a	$85.00
Bio-oil/Bio-char (green wood) Technology Pathway 16	$90.00
Bio-products (green wood) Technolgy Pathway 17	$90.00
Thermal Energy (cordwood) Technology Pathway 5	$104.00
Thermal Energy (cordwood) Technology Pathway 6	$104.00
Thermal Energy (green wood) Technology Pathway 8	$104.00
CHP (green wood) Technology Pathway 11	$104.00
Gasifier (green wood) Technology Pathway 12	$104.00
Thermal Energy (pellets) Technology Pathway 7b*	$167.00

[11] Chart information is derived from Appendix 2-B. See that appendix for sources.

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CHAPTER 3
FOREST BIOMASS SUPPLY

3.1 INTRODUCTION AND MAJOR FINDINGS

Massachusetts has attracted the attention of bioenergy proponents and investors, in part due to a substantial rise in timber inventories over the last several decades. Recent studies on the availability of biomass to support new bioenergy plants have focused on incremental forest growth—implicitly treating inventory accumulation as potential supply—and confirmed expectations that inventories will continue to rise significantly. These studies thus concluded that available biomass is more than adequate to furnish several large-scale electric power plants without reducing timber inventories below current levels.

At this juncture, state policymakers require a better understanding of biomass supply, looking at factors beyond forest growth. Policymakers need to know whether the objectives of different energy policies are consistent with available wood supply, and how forest biomass harvests might respond to different economic realities that may be driven policy choices. With this perspective, we have crafted this analysis of forest biomass supplies in 2010–2025 around two central questions:

- How much forest biomass would be supplied at current biomass stumpage prices if there is an increase in demand from bioenergy plants?
- How much would forest biomass supplies increase if bioenergy plants pay higher prices for wood?

Another goal of this supply analysis is to better understand the implications of potential biomass harvest levels for forest health and forest harvesting guidelines.

3.1.1 CONCEPTUAL FRAMEWORK FOR FOREST BIOMASS SUPPLY ANALYSIS

Key Study Features

Our approach focuses on economic issues and landowner behavior and has been developed with an eye toward the availability and quality of relevant data. Unlike previous forest-growth-based studies,[1] this study of forest biomass supply in Massachusetts has several features that are different: 1) it is explicitly linked to energy prices; 2) it incorporates data on biomass harvesting and production costs; 3) it provides a detailed analysis of historical harvesting patterns on private lands, thus recognizing landowner willingness to harvest along with harvest intensity; 4) it considers the effect of stumpage prices and per-acre income on landowner behavior; 5) it is closely linked to available timber inventory in terms of accessible areas, mature volumes on private lands, and stocks of low-value trees; 6) it treats public lands separately and utilizes information on historical harvest levels, new Forest Resource Management Plans, and the Forest Futures Visioning Process; and 7) it incorporates sustainability criteria that have been developed and presented in Chapter 4.

We define forest biomass as wood supplied from forest management activities on private lands and public lands. These two ownership categories are considered separately in our analysis because they differ in several important ways: 1) the factors that determine the decision to harvest; 2) forest management objectives on private and public lands, and thus silvicultural prescriptions and harvesting techniques; and 3) harvest intensity and timber yields. In terms of area harvested in Massachusetts each year, private lands dominate with an average of about 22,000 acres harvested annually in 2000–2009.[2] In contrast, only about 4,000 acres of public land were harvested annually in the same time period. Note that we do not include land clearing as a source of forest biomass, because it is not a forest management activity and there are issues related to definitions of renewability. Nevertheless, it is the source of a substantial volume of wood (the average area of land cleared for development in 1999–2005 was estimated to be almost 5,000 acres per year) and so we have provided a separate section on potential biomass volumes from this source.

Incremental Biomass Production

The purpose of this supply study is to evaluate how much forest biomass would be available to furnish the potential expansion of bioenergy capacity and production in Massachusetts. For this reason, our analysis and projections are focused on incremental biomass production, not total production. The volume of biomass chips that has been produced from forest sources historically is considered to be "utilized" and, since this wood is already accounted for, it is not available to meet the demand from new bioenergy plants. We sometimes refer to this incremental production as "new" biomass.

Two Biomass Price Scenarios Linked to Energy Prices

We have developed two biomass price scenarios—linked to energy prices—that are intended to provide DOER with guidance as to how much wood may be available to furnish new bioenergy plants. These scenarios recognize the importance of stumpage prices and income in influencing landowner behavior, and the important relationship between delivered biomass prices and harvesting systems/logging costs. This section discusses these scenarios with respect to electricity prices; thermal and CHP

[1] Recent studies using the forest-growth approach to assess biomass availability in Massachusetts are reviewed in Appendix 3-A. While these studies provide useful information on how much wood could be harvested on an ongoing basis without reducing inventories below current levels, they do not address the complex economic and social factors that will determine how much of this biomass would actually be available to furnish new biomass facilities. We have developed estimates of biomass availability using a forest-growth approach in Section 3.2.5 so that they may be compared with the results of the approach that we have developed.

[2] The data and information provided in this section are summarized from the main body of this chapter. Sources and references are contained in the relevant sections.

are addressed in the following section. Note that this assessment is intended to provide estimates of forest biomass potential over the medium term; in the near term, logging and infrastructure constraints (not addressed in this study) could be significant obstacles to harvest increases.

Our starting point is to estimate the potential of forest biomass to supply electric power plants in Massachusetts. This is an area of immediate concern for DOER given that they are now considering proposals for several facilities and the adequacy of wood supplies to furnish these plants is a central issue. In this scenario, our assumptions have been developed to reflect the current pricing environment for electricity and biomass: real electricity prices are assumed to remain near recent levels as are the price of renewable energy credits.[3, 4] Consistent with this assumption, real biomass prices are also assumed to remain near recent levels: delivered wood prices at power plants would be about $30 per green ton, and biomass stumpage prices would average $1–$2 per green ton. We refer to this scenario as the "Low-Price Biomass" scenario.

Our second scenario is intended to provide perspective on the upper bound for forest biomass production if bioenergy demand and prices increase beyond the level established in the Low-Price Biomass scenario. It is not reasonable to specify an absolute maximum for biomass supply since supply is an economic concept which depends on timber prices (and a host of other factors). Thus, we need to specify a "high" biomass stumpage price, and then consider how private landowner harvests might respond to this price level. Forest biomass volumes could still increase beyond this level, but it would be increasingly difficult to due to biophysical, economic, and social constraints and increasingly unlikely due to macroeconomic and energy constraints. We refer to this future outlook as the "High-Price Biomass" scenario.

How high should the biomass stumpage price be in this "limiting"case? For increased demand from new wood-fired electric power capacity, we have developed an upper-range electric price scenario that leads to real biomass stumpage prices of about $20 per green ton.[5] The significant increase in real electricity prices needed for power plants to purchase wood in this scenario could be triggered by either macroeconomic or policy shifts.[6] Also, policy initiatives (such as REC's) that provide higher income for utilities could be compatible with this level of biomass stumpage prices.[7] We should note that we think that the high level of electricity prices that would drive this scenario is unlikely on the basis of macroeconomic trends and projections of future escalation in coal and natural gas prices. Significant changes in government policies would probably be necessary for this scenario to unfold and could take the form of greater incentives for electric power, or policies that spur substantial investment in thermal, CHP plants, and pellet plants.

How much forest biomass would landowners be willing to supply in response to higher prices? As demand and prices increase, more wood can be supplied from private lands by increasing removals of low-value wood from sites that are already under harvest, diverting wood from other end-use markets (such as pulpwood) to biomass, and increasing the number of acres being harvested. The standard and most direct approach to answering this question would be to estimate the effect of price changes on harvest volumes directly (that is, the timber supply elasticity). We have presented some results from our analysis of this relationship in Massachusetts, but they are merely suggestive due to the poor quality of the data on both harvest volumes and prices.

A second approach would be to rely on the literature for estimates of timber supply elasticities that have been developed in other regions. Available studies generally show that timber supply is very inelastic (that is, price changes have little or no influence harvest volumes).[8] However, these results are not necessarily relevant in evaluating the biomass supply situation in Massachusetts because the characteristics of the landowners, timber inventory, and forest products industry are very different. Importantly, there are two issues not addressed in previous research that are likely to have a significant effect on forest biomass supply behavior in Massachusetts and call for an alternative approach.

The first issue relates to biomass prices and per-acre incomes. Studies which examine the relationship between harvests and prices generally focus on sawtimber prices (and sometimes pulpwood) because these dominate the value of a harvest in most regions.

[3] Reference case (or base case) forecasts of electricity prices suggest that real prices will remain relatively flat over the next 15 years, as they play off a projected declining trend in real natural gas prices and a slightly increasing trend in real coal prices (see for example, Annual Energy Outlook 2010: U.S. Energy Information Administration, 2009).

[4] The assumption about REC's is important since they provide a significant share of revenue for wood-fired power plants and they can be modified by state policy.

[5] The delivered wood and electricity prices consistent with this scenario are discussed later in this report.

[6] There are numerous policies under consideration that could lead to such changes (see U.S. Environmental Protection Agency, 2009: EPA Analysis of the American Clean Energy and Security Act of 2009).

[7] If electric power plant demand for wood increases but there are no increases in electricity prices that would allow power producers to pay the higher prices needed to generate more wood supply, then direct payments to landowners would be another policy that could lead to more biomass production.

[8] There are many issues with these studies that raise concerns, perhaps the most serious being data limitations and errors in measuring price and harvest variables. In addition, many studies estimate binary choice models and only address the question of whether or not price has an effect, not the magnitude of that effect.

However, if biomass prices rise significantly, they can make an important contribution to income and influence landowner decisions.[9] The second issue is the age structure of the inventory in Massachusetts. Many empirical studies consider inventory levels in a broad sense, but none directly consider the age structure of the inventory. A large percentage of the private forests in Massachusetts are now over 60 years old and are ready—if not overdue—to be thinned for landowners interested in commercial timber production[10]; financial incentives could have an important effect on the decisions of these landowners.

These concerns have led us to an approach for the High-Price Biomass scenario that recognizes landowner characteristics, the age structure of the inventory, and the importance of per-acre income levels. While we believe this method provides a better estimate of forest biomass supply than traditional economic approaches, a good deal of uncertainty concerning landowner responses cannot be eliminated since we are considering behavior that is well beyond our historical experience. As demand and prices increase, the confidence intervals grow wider and it is important to recognize and acknowledge this uncertainty.

Biomass Supplies for Thermal and CHP Plants

It is relatively straightforward to extend the above scenarios to evaluate the availability of forest biomass supplies for wood-fired thermal and CHP plants. The cost structure of thermal and CHP plants and their competition with facilities that use oil and natural gas allow them to pay much higher prices for wood than electric power plants. For example, in current markets (assuming oil prices of $3 per gallon), thermal and CHP plants could pay up to $85–$95 per green ton of wood (45% moisture content) and still cover their full cost of capital (based on the analysis in Chapter 2).

In terms of wood supply, one important difference between electric power and thermal/CHP plants is that the latter prefer higher-quality chips that are uniform in size and shape and have low ash content (Maker, 2004; P Squared Group and Biomass Energy Resource Center, 2008). Clean chips and chip specifications in general may add about $10–$15 per green ton to the cost of chip production. Thus, thermal and CHP plants would need to pay $40–$45 per delivered green ton compared to $30 for an electric power plants.[11] Importantly, in the same woodshed, thermal and CHP plants can pay this difference—and much more if necessary—and remain profitable.

At the high end of the supply curve, if the market price of delivered wood for electric power plants is $50–$60 per green ton, thermal and CHP plants would face wood prices in the range of $65–$75 per green ton. This price level is still below the range that these plants could afford to pay today and cover their full costs. Of course, if electric power prices increase due to macroeconomic factors and fuel costs, it is a safe bet that oil prices would be much higher as well; in fact, most forecasts indicate that oil prices will increase faster than electricity prices (which are tied more closely to the cost of coal and natural gas).

In sum, higher-quality chip specifications for thermal and CHP plants shift the supply curve for delivered wood chips upward relative to that of electric power plants. Under reasonable energy price scenarios, when these plants compete for the same wood supply, thermal and CHP plants will be able to outbid electric power plants due to their production economics and the competitive environment of the energy markets in which they operate.

Harvesting Systems and Logging Costs

We have conducted our assessment of wood biomass supply in Massachusetts with and without the harvesting restrictions—particularly with respect to the removal of tops and limbs—that are provided by the guidelines in Chapter 4 of this report.

Our assessment of biomass supply in Massachusetts suggests that if demand increases due to the expansion of electric power plants, it will almost certainly be accompanied by increases in whole-tree harvesting due to the limited supply of other forest biomass and the cost advantages of whole-tree methods. Generally, we assume that whole-tree harvesting can be used on private lands as long as it meets the forest practices standards required by the state. Given the uncertainty regarding the acceptance of whole-tree harvesting (particularly mechanical systems) in Massachusetts, our supply projections allow for the fact that many landowners, foresters, and loggers will still favor alternative harvesting methods.

Thermal and CHP plants are not constrained to use whole-tree harvesting methods because of their ability to pay higher prices for delivered wood chips. These facilities could buy wood procured with log-length methods, in which trees are delimbed and bucked at the stump and the logs are forwarded or skidded to the landing. Log-length methods may be selected over whole-tree methods if management plans call for leaving tops and limbs scattered on the site and/or there is concern about damage to soils or to the

[9] Landowners may also respond differently to an equivalent amount of income from harvesting biomass and sawtimber because the removal of low-value biomass may have a different impact on the value of non-timber amenities than the removal of large trees.

[10] Kelty et al. (2008) reference silvicultural research that indicates that 50 years is the recommended age for first thinning (cited from Hibbs and Bentley, 1983), but indicate that first thinnings in Massachusetts are commonly delayed until stands reach 70 years of age.

[11] While thermal and CHP plants will compete for bole chips, electric power plants can use whole-tree chips from tops and limbs. However, given the wood supply situation in Massachusetts, it appears that electric power plants would need to obtain most of their wood from whole trees and thus could face the prospect of competing directly with thermal and CHP plants for bolewood when operating in the same woodshed.

residual stand (Fight et al., 2006). As noted earlier, our estimates indicate that log-length harvesting methods would add about \$10–\$15 to the cost of a green ton of chips.

3.1.2 MAJOR FINDINGS AND CONCLUSIONS

Here we summarize the major findings of our wood supply assessment:

Forest Biomass Supply Available in Massachusetts with Low-Price Stumpage

- At current prices for biomass stumpage, we estimate that about 150,000–250,000 green tons of "new" biomass could be harvested annually from forest lands in Massachusetts.[12] Most of this material would be sourced from standing trees due to the small size of the forest industry in Massachusetts, and hence the limited supply of logging residues and limited opportunities for log merchandizing. This wood would be available to electric power, thermal, CHP or other bioenergy plants; however, if the wood is harvested as feedstock for electric power plants, whole-tree harvesting would be necessary to produce chips at \$30 per delivered green ton.

- We estimate that virtually all of the "new" forest biomass supply would be harvested from private lands. Given the low price of stumpage in this scenario, biomass producers would have economic access only to low-value wood and it would be harvested almost exclusively on sites that are already being harvested for sawtimber. If whole-tree harvesting operations are established for biomass production, it would also become economical to remove sawtimber logging residues from those same sites. Applying the ecological guidelines provided in Chapter 4 of this report, our projection shows that tops and limbs from industrial roundwood would account for about 15%–20% of the "new" biomass harvest from private lands.

- We find that there would likely be little or no increase in biomass production from public lands. Our review of Forest Resource Management Plans and anticipated forest policies leads us to conclude that the total volume of wood harvested on public lands in 2010–2025 will be about the same level that we have observed during the past decade. We have assumed that biomass fuel will not be diverted from other end uses (such as pulpwood) in this scenario. Logging residues are not projected to contribute to supply because of ecological restrictions and poor economics.

Forest Biomass Supply Available in Massachusetts with High-Price Stumpage

- Higher biomass stumpage prices could dramatically affect the supply of biomass by providing economic incentives that bring more private land into timber production, increase the harvest intensity on all lands that are harvested, and divert wood from pulpwood and other end-use markets to biomass. With our scenario of biomass stumpage prices at \$20 per green ton, per-acre income from wood sales could double and we estimate that about 685,000–885,000 green tons of "new" forest biomass could be produced annually in Massachusetts.

- Increased prices would not be expected to lead to higher harvest levels on public lands. However, at these higher stumpage prices, biomass supplies would increase as wood from public lands would likely be diverted from pulpwood to bioenergy plants. The volumes would be small, however, and would account for only about 5% of "new" statewide forest biomass production.

- We have estimated a "sustainable" level of biomass supply using the criteria that harvests do not exceed net growth and that biomass harvests can be maintained at the same level for the foreseeable future. Based on our estimates of operable private land area and our growth estimates in Chapter 5, we have calculated that average annual biomass supply could be 900,000 green tons per year. Thus, the high end of the range that we derived using our approach (885,000 green tons) would be considered "sustainable" by this definition. In addition, our analysis suggests that the "supply" estimates developed using forest-growth approaches would only be consistent with very high biomass stumpage prices.

Forest Biomass Supply Available from the Border Counties

- We evaluated supplies in the border counties (NH, VT, NY, CT, and RI) by considering timberland area, timber inventory, growth rates, ownership characteristics, and forest products production. There is no simple scheme to weight these factors, but our best estimate is that incremental forest biomass production in the border counties would be about 50% greater than that of Massachusetts. The logic of our two scenarios still applies: at low biomass stumpage prices, "new" volumes would be limited because they come primarily from the additional harvest of low-value wood on sites already being logged for other commercial timber; at high biomass stumpage prices, the harvested land base would increase considerably, as would the harvest intensity on these sites.

- Biomass produced in the border region could be consumed in the "local" market, shipped to Massachusetts, or shipped to the next ring of bordering counties and beyond. The eventual destination for this wood will depend on the location and timing of new capacity investment throughout the region and a variety of other factors such as transportation costs, infrastructure, and supply logistics. While this is a complex problem with a high degree of uncertainty, we think that as a general planning guide it would be prudent to assume that Massachusetts could successfully purchase only half of the available wood. Thus, in the Low-Price Biomass scenario, "new" forest biomass available from the border counties to

[12] The major uncertainty that accounts for this range is the average volume of biomass material removed from an acre. It is also possible that some pulpwood could be diverted to biomass fuel at relatively low biomass stumpage prices, but we have not introduced this potential shift in the Low-Price Biomass scenario.

furnish bioenergy plants in Massachusetts would be about 110,000–190,000 green tons per year. With the assumption of high biomass stumpage prices, forest biomass supplies from adjacent counties would increase to about 515,000–665,000 green tons annually.

Our projections for incremental forest biomass production in Massachusetts and the border counties are summarized in Exhibit 3-1. Although we have provided a range of estimates in this table, there are, of course, a wider set of possible outcomes for these scenarios. This uncertainty is largely due to our limited historical experience with biomass harvesting in Massachusetts, and this becomes a greater concern when we analyze the impact of much higher biomass prices. We have conducted sensitivity analysis of some of our key assumptions within this chapter. Perhaps the most significant source of uncertainty is how private landowners will respond to the prospect of earning higher income from biomass harvests. Another general issue is the acceptance and adoption of whole-tree harvesting by landowners, foresters, and loggers in Massachusetts—this is particularly important in scenarios involving electric power expansion since whole-tree harvesting would likely be necessary due to cost considerations. For the border counties, it is more difficult to address the issue of confidence intervals because our estimates were established relative to Massachusetts, and then scaled down to recognize that facilities outside of Massachusetts would compete in this same woodshed.

Exhibit 3-1: Summary of Forest Biomass Fuel Supplies for 2010–2025

Low- and High-Price Biomass Scenarios 000 Green Tons per Year		
	Low-Price	**High-Price**
Massachusetts		
Private Lands	150–250	650–850
Public Lands	0	35
Total	150–250	685–885
Border Counties	110–190	515–665
Combined Total	260–440	1,200–1,550

Note: Estimates have been rounded for this table.

We have focused on two price scenarios for forest biomass supply, with the high-price scenario intending to provide an approximate upper bound for incremental biomass harvests. Clearly, these two price levels represent only two points on a supply curve that embodies many price-harvest combinations. A few comments on the shape of this curve are appropriate. At current/low price levels, the supply curve for private owners is presumed to be flat suggesting that any volume of forest biomass up to the range of 150,000–250,000 green tons per year could be procured at these prices. At high-end prices, we would expect that the slope of the curve would be relatively steep reflecting landowner resistance to harvesting additional acres due to the greater value that owners at the margin may place on non-timber amenities. This nonlinearity suggests that if bioenergy capacity increases in Massachusetts, it may not be difficult to procure wood at affordable prices in the early stages of expansion, but it could become more problematic as prices rise nearer to the levels assumed in the High-Price Biomass scenario.

3.1.3 POTENTIAL WOOD BIOMASS SUPPLIES FROM OTHER SOURCES

This assessment has focused on the core issue of biomass production from forest sources. It is important to recognize that there are other biomass sources that could potentially make a substantial contribution to the supply of wood available for new bioenergy facilities in Massachusetts. These can be classified into three major categories: 1) wood from land clearing; 2) wood from mill residues and tree care/landscaping sources; and, 3) wood grown in short-rotation plantations.

Wood From Land Clearing

There is a high degree of uncertainty in estimating the area of land that is cleared each year in Massachusetts, the amount of wood removed from that land, and the current disposition of that wood. As a result, it is difficult to estimate the volume of incremental biomass supplies that could be generated from land clearing over the next 15 years. Holding the area of land cleared annually constant, we have calculated that a 10% increase in the recovery rate[13] would yield an additional 30,000 green tons per year of biomass that could furnish an expansion in bioenergy plants. Given current disposal costs for cleared wood and current potential uses for that wood, it would seem that an increase in recovery rates from 30% to 70% (at high biomass stumpage prices) would provide reasonable bounds for the potential supply from this source. This translates to a maximum volume of 120,000 green tons of "new" biomass given our assumptions on the area of land cleared and the expected diversion of high-quality wood to other end-use markets.

Wood Biomass From Mill Residues and Tree Care/ Landscaping Sources

Among these other sources, the most significant is wood from tree care/landscaping sources. This wood is often referred to as "urban wood" which is somewhat of a misnomer because it includes wood not only from tree care in urban areas, but also wood from tree care from sources such as county parks and recreation areas and maintenance of electric power lines. The term can also be confusing because it is not always clear whether it includes "urban waste" such as construction debris.

A literature review conducted in 2002 indicated that tree care/ landscaping sources accounted for 1.0 million tons (42%) out the total available supply of 2.5 million tons of non-forest wood biomass in Massachusetts (Fallon and Breger, 2002). However, given the difficulties in estimating this volume (noted in the report), this estimate is perhaps best used to suggest that the potential from

13 We define the recovery rate as the percentage of wood cleared that is used for industrial roundwood products or industrial and residential fuelwood.

these sources may be substantial and worthy of further investigation (importantly, the carbon profile of this material is generally similar to logging residues and thus very favorable compared to that of harvesting standing trees).

Two other important sources of wood biomass that should be noted are mill residues and urban waste (municipal solid waste, and construction and demolition debris). Although mill residues can be a valuable source because they are clean, dry and easily accessed, they are generally fully utilized. Moreover, mill residue supplies in Massachusetts have been declining in parallel with the contraction in lumber production. On the other hand, solid waste and C&D debris may be considered under-utilized, but are expensive to sort and can be difficult to recover due to contamination issues.

Short-Rotation Wood Plantations

DOER and DCR commissioned a study that included an evaluation of the potential of growing short-rotation willow crops in Massachusetts for bioenergy use (Timmons et al., 2008). In light of our forest biomass supply assessment, there are three reasons that the potential of this supply source on marginal agricultural lands may deserve more attention if DOER wishes to promote bioenergy development. First, our economic analysis has shown that the potential to produce forest biomass chips in the current pricing environment and with current policy incentives is significantly less than suggested by previous studies that were focused on forest growth. Second, although BCAP policies are now undergoing revision, the proposed rules offer significant subsidies for the establishment and development of wood energy crops (see policy review in Chapter 1). Third, if carbon emissions are an important consideration in state energy policies, closed-loop short-rotation crops have some obvious advantages when compared to natural forest biomass sources.

3.1.4 REPORT ORGANIZATION

This report is organized as follows. Section 3.2 provides an in-depth analysis of biomass supplies from private lands in Massachusetts. We begin with a review of historical levels of timber harvesting since we believe this is fundamental to understanding future biomass supplies—biomass production often makes economic sense only when integrated with sawtimber harvests. The forecast for low-price biomass supply requires the review of three important topics: 1) costs of whole-tree harvesting; 2) low-value wood supply in sawtimber stands; 3) landowner willingness to increase harvest intensity. In order to generate a forecast of high-price biomass supplies, the discussion is extended to include: 1) the size of the operable land base after adjusting for biophysical factors and landowner characteristics; 2) landowner response to higher wood prices and higher per-acre income levels.

Section 3.3 discusses the potential for harvesting "new" biomass supply from public lands, and covers both historical harvest levels and projections of wood harvests. Our forecasts for forest biomass supplies in Massachusetts are summarized by source for our two biomass stumpage price scenarios in Section 3.4. Section 3.5 reviews potential biomass production from other sources, including land clearing and conversion.

In Section 3.6, we present our assessment of biomass supply from nearby states by evaluating their potential relative to Massachusetts. Key topics covered include timberland area, timber inventory, timber growth, forest products industry status and associated harvesting levels, and landowner characteristics. After developing estimates of potential additional biomass production in the border region, we conclude by discussing some of the factors that determine where this wood might eventually be consumed.

Some of our work and analysis has been presented in several Appendices, which include the following topics: 1) a review of results of previous studies on forest biomass availability in Massachusetts (Appendix 3-A); 2) logging residue data and methods for estimation (Appendix 3-B); 3) firewood production and consumption in Massachusetts (Appendix 3-C); 4) an analysis of biomass potential in southern New Hampshire (Appendix 3-D).

3.2 BIOMASS SUPPLY FROM PRIVATE LANDS IN MASSACHUSETTS

Private timberlands in Massachusetts are by far the most important source of "new" or incremental forest biomass production because of their size and the ability of landowners to adjust their harvest decisions in response to changes in market conditions. The analysis in this section is organized as follows: 1) historical estimates of timber harvests; 2) review of potential supplies from logging residues; 3) projection of biomass supplies in the Low-Price Biomass scenario; and 4) projection of biomass supplies in the High-Price Biomass scenario. Our projections include a review of harvesting costs, and examine the important role of stumpage prices in influencing production volumes.

3.2.1 HISTORICAL ESTIMATES OF TIMBER HARVESTS ON PRIVATE TIMBERLAND

The economics of forest biomass production are generally most favorable when biomass harvests are integrated with sawtimber harvests. In this section, we provide a detailed analysis of historical patterns of timber harvests in Massachusetts to lay the groundwork for our projections of sawtimber and other industrial roundwood harvests. Unless income incentives increase substantially under some scenarios that are described under our High-Price Biomass scenario, the harvesting footprint with biomass is likely to be very similar to that for industrial roundwood alone. Biomass production will then come from increasing the harvest intensity on these lands, by taking tops, limbs, and low-value standing trees.

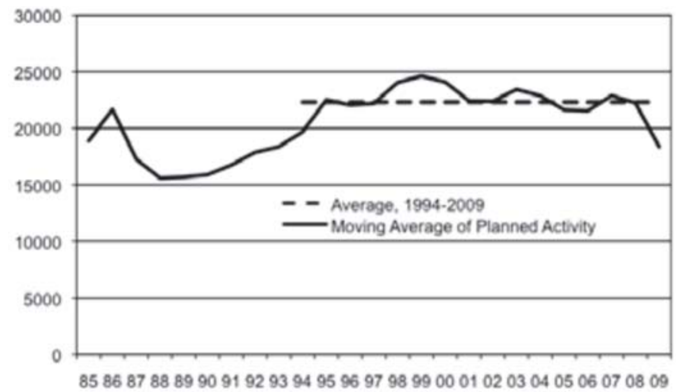
Unlike several states in the Northeast region, Massachusetts does not track and collect data on annual harvest levels. Thus, this analysis relies on forest cutting plans (FCPs) that are required by the state under the Forest Practices Act. Although FCPs have several important limitations with regard to coverage and timing[14], they are the best data source available to identify important long-term trends in harvesting activity in Massachusetts. We have obtained these data for 2001–2009 from the Massachusetts Department of Conservation and Recreation, and for 1984–2000 from research at the Harvard Forest (Kittredge et al., 2009).

The FCP data indicate that the average annual volume of wood "harvested" from private lands in 2001–2009 was 323,000 green tons.[15] Average volumes by end-use market according to these plans were 224,000 green tons of sawtimber, 84,000 green tons of "pulpwood," and 16,000 green tons of fuelwood. However, one must be cautious in interpreting these data because wood that is classified as pulpwood may actually be consumed for fuel, either in residential or industrial uses—wood classifications and conversions to green tons are discussed in more detail later in this section.

In order to analyze these data, we first consider acres harvested on all private lands, which are shown in Exhibit 3-2. Harvested acres dropped sharply in the late 1980s, but rebounded by the mid-1990s and have been relatively flat since that time. In fact, the stability of the private land area harvested over the past 15 years is remarkable given the number of factors that influence this trend, including overall demand levels for wood products, and harvest volumes supplied from public lands and land clearing activity. We should note that forest industry lands are only a small portion of the private land base in Massachusetts (harvests on industrial lands account for only about 5% of acreage as well as 5% of volume removed); thus, we have not disaggregated private lands into industrial and non-industrial components as is commonly done in timber supply analysis.

This "stable" trend is more interesting in light of the fact that the area of private timberland in Massachusetts has declined by 20% during this period, from 2.5 million acres in 1985 to 2.0 million acres in 2008 according to FIA data[16] (these data suggest that this shift was primarily due to a transfer of timberlands from private to public ownerships, with land conversion playing a much less important role[17]). While the stability in area harvested is open to various interpretations, the most probable explanation would relate to the small share of land that is harvested. Thus, in spite of the increasing fragmentation of the land base and the small average parcel size of ownership, the data suggest that much of the harvesting in Massachusetts may take place on an operable land base that may not have changed much over this period of time.

Exhibit 3-2: Acres Harvested on All Private Lands, 1985–2009



Note: Derived from Forest Cutting Plans assuming 95% of plans are completed.

As noted above, sawtimber demand is the key driver of harvesting activity on Massachusetts timberland and thus critical to the analysis of potential biomass supply. Over the historical time period, the sawtimber harvest on a per-acre basis has ranged from a low of about 1,600 board feet (International ¼" log rule) in 1991 to a high of 2,200 board feet in 2006 (Exhibit 3-3). The average in 1994–2009 was 2,000 board feet per acre.[18]

The stability in the volume of sawtimber harvested on private lands in 1994–2009 contrasts markedly with the large decline in lumber production during this period. Lumber production in Massachusetts was just over 100 million board feet in 1993 and

[14] Important limitations include: 1) they are pre-harvest plans and thus the volume to be harvested is only an estimate of what was actually cut; 2) once filed, the plans can be implemented over the following two years and there may be extensions (for two additional years); in addition, those who file may choose not to harvest at all; 3) they are only required for wood harvests greater than 50 cords or 25,000 board feet; 4) they are only required if the land remains in forest use and thus do not include land clearing. These issues are discussed in Ch. 132 of the Massachusetts Forest Cutting Practices Act and by Kittredge et al., 2009.

[15] Although these data are pre-harvest levels as stated in the Forest Cutting Plans, we refer to them as though they are "actuals," partly for convenience, but also because we have adjusted them, reducing the levels by 5% (based on information reported by Kittredge et al., 2009) and using a distributed lag function to allocate harvests over multiple years to account for the fact that those who file plans have up to two years to harvest with the possibility of extensions.

[16] Reference to FIA data is made frequently throughout this report. FIA refers to the Forest Inventory and Analysis National Program which provides detailed data on forests and forestland based on surveys by the U.S. Forest Service.

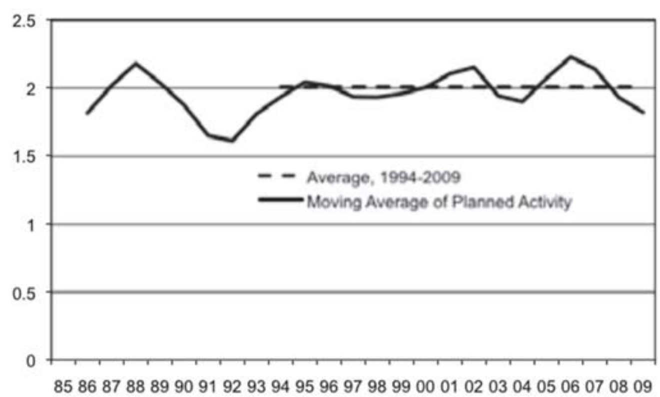
[17] It should be noted that it is difficult to quantify accurately the magnitude of these land shifts and different data sources can lead to different conclusions. For example, using the same FIA database and considering forestland in Massachusetts (forestland area is about 5% greater than timberland area) suggests larger losses in the private land base, smaller gains in the public land base, and a much higher share of land lost to conversion. Data that provide direct measurements of land conversion in Massachusetts are discussed later, but these data also have numerous problems and are not consistent with the FIA trends.

[18] It is interesting to note that Kelty et al., 2008 report that a 50% overstory thinning on average private lands in Massachusetts would yield 2 MBF (International ¼" log rule) per acre.

edged higher to 104 million board feet in 1996; however, production was estimated to have been only 69 million board feet in 2001 and 49 million board feet 2005 (Damery et al., 2006). On public lands, sawtimber harvests were also flat over the past 15 years according to FCP data. One interpretation of these trends would be that the contraction in lumber production was less a function of final demand than of the competitive position of sawmills in Massachusetts, and high-quality sawlogs continued to be cut and shipped out of state to be processed elsewhere. Another factor that needs to be considered is that it appears that land clearing dropped sharply over this time frame; thus, a potentially important source of sawlogs declined substantially and may have increased the demand for sawlogs from private lands.

Most importantly for this study, in spite of major changes in local processing capacity and demand and some significant price swings, acres harvested and sawtimber harvests have remained relatively stable. These trends provide the basis for our projections of future harvest levels in Massachusetts.

Exhibit 3-3: Average Sawtimber Harvest Intensity on All Private Lands, 1985–2009 (000 board feet, International ¼" log rule per acre)



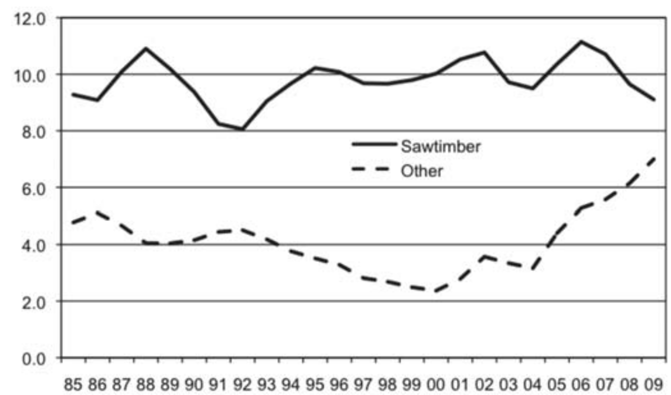
Note: Derived from Forest Cutting Plans assuming 95% of plans are completed.

In order to project forest biomass supply, it is also important to consider the volume of timber that is being harvested for other end uses. These calculations provide insight into other demands on the resource base, harvest intensities on timberland, and the potential for additional harvests of biomass. In order to compare the harvest volumes reported on the FCPs, we converted sawtimber (MBF, International ¼" log rule), pulpwood (reported as 128 cubic-foot cords), and fuelwood (reported as green tons) to common units (green tons in this case). Harvest intensity for sawtimber in green tons per acre is contrasted with the other industrial roundwood uses in Exhibit 3-4.[19] Other industrial roundwood fell from about 4 green tons per acre in the early 1990s to only about 2 green tons per acre in 2000. Since that time, other industrial roundwood harvests have climbed sharply, reaching 7 green tons per acre in 2009 (according to plan data, this consists of 5 green tons of pulpwood and 2 green tons of fuelwood).

We should also note that our analysis of historical timber harvests includes only a small percentage of the total volume of firewood that is cut and consumed in Massachusetts. FCPs are required only for harvests that exceed 50 cords and it appears that most firewood is produced in much smaller operations. This is consistent with Massachusetts landowner surveys that suggest that many owners of small parcels are interested in firewood harvests, but not harvests of industrial roundwood.

Exhibit 3-4: Average Harvest Intensity on All Private Lands, 1985–2009

Sawtimber compared with Other Industrial Roundwood (green tons per acre)



Note: Derived from Forest Cutting Plans assuming 95% of plans are completed.

For this study, we have assumed that residential fuelwood harvests do not have a significant impact on the potential for forest biomass supply since most of the biomass for industrial use is likely to come from larger harvesting operations. However, there is an interface between the two sectors as some residential fuelwood does get cut during industrial roundwood harvests, and sometimes in follow-up harvests if crews move in to remove smaller wood or standing dead wood. This area may deserve additional study because of the large volume of firewood production in Massachusetts, which we estimate may be two-to-three times the volume of industrial roundwood harvested (see Appendix 3-C).

3.2.2 LOGGING RESIDUES

Most studies of potential forest biomass availability start with logging residues because: 1) they represent a substantial volume of wood (4.5 billion cubic feet in the U.S. in 2006, which compares with 15.0 billion cubic feet of roundwood harvested for all products (Smith et al., 2009); 2) their removal has been considered integral to forest and ecological health in many situations due to potential fire hazard and insect damage; 3) they are perceived to be underutilized and have additional value as product output;

[19] We have combined pulpwood and fuelwood into "other industrial roundwood" because the two classifications are not reliable indicators of their end-use markets. Some pulpwood—perhaps more appropriately referred to as cordwood—can be cut and split for firewood, and may be chipped for biomass. Fuelwood is comprised of roundwood that is processed for residential firewood, and also wood that is chipped for industrial biomass use.

4) they are assumed to be the most easily procured—and thus the least costly—source of biomass supply from forests. Logging residues have been a central focus of many studies (for example, the "Billion-Ton-Study," Perlak et al., 2005) and are considered a key source of forest biomass fuel.

3.2.2.1 Logging Residue Generation

Here we consider the potential volume of forest biomass supplies from logging residues in Massachusetts. The primary source of logging residue data in most studies is the Timber Products Output (TPO) reports from the U.S. Forest Service. These data could not be used directly for Massachusetts due to problems in the underlying database (see Appendix 3-B for a full discussion of the logging residue data). In addition, the TPO methodology tends to overstate the volume of logging residues available for biomass fuel because the data include a significant volume attributable to breakage and residual stand damage.

For these reasons, we have devised an alternative approach in which we estimate the volume of tops and limbs associated with harvesting trees of varying diameter classes (the derivation of these estimates is provided in 3-B). When these percentages of top and limb material are applied to recent industrial roundwood harvest levels, they suggest that the total volume of "logging residues" generated on private lands in Massachusetts is on the order of 100,000 green tons per year.[20]

3.2.2.2 Logging Residue Recovery

Most studies that evaluate the availability of logging residues make the assumption (sometimes implicitly) that the bulk of logging residues are delivered to a landing as part of normal harvesting operations. In these logging operations, a tree is assumed to be delivered to the landing for the value of the sawlog and pulpwood, while the "wastewood" is assumed to be a by-product of the operation with zero costs for "delivery" to a landing. With these assumptions, the portion of the tree that could be considered biomass fuel is inexpensive and available for the cost of chipping and transport to a bioenergy facility. While this may be true in many regions, it is generally not the case in Massachusetts where logging operations commonly consist of manual felling, bucking into logs in the field at the stump, and cable skidding or forwarding; thus, most tops and limbs remain on the ground where the trees are felled.

While it may be feasible to recover scattered logging residues in some circumstances, it seems fair to conclude that biomass supply from logging residues in Massachusetts would be minimal without some modifications to existing harvesting operations. Although these logging residues do have the advantage of having been felled at no cost to the biomass producer, the high cost of collection and delivery to a central location would generally be prohibitively expensive.

In order to produce biomass competitively from tops and limbs, whole-tree harvesting operations would likely be necessary to reduce the costs of landed residue material. Rather than topping and limbing felled trees at the stump, trees could be skidded to a landing with some portion of the top and limbs remaining intact. Tops and limbs could then be removed at the landing and chipped there. If biomass is produced in this manner, the primary costs would be chipping (about \$6–\$7 per green ton for slash) and transport from the landing to a bioenergy plant (directly dependent on distance, but averaging about \$8–\$12 per green ton).[21] Thus, total delivered costs would be \$14–\$19 per green ton.[22]

3.2.2.3 Forecast of Forest Biomass Supply from Logging Residues on Private Lands

In order to project biomass supplies that can be used to meet potential demand from new bioenergy plants, we have assumed that 65% of the tops and limbs from harvested trees can be recovered on acres where silvicultural prescriptions include whole-tree biomass harvests. This percentage was selected for two reasons: 1) it leaves behind more than enough material to conform to the ecological guidelines that have been spelled out in Chapter 4; 2) it recognizes that a significant share of tops and limbs remain uneconomic due to timber breakage, small pieces, and small branches. Some issues, such as difficulties in handling large hardwood crowns, encompass both ecological and economic concerns.

Harvests of logging residues have been considered in conjunction with harvests of standing forest biomass in the following sections. We did not consider it useful to develop a separate biomass supply scenario for only logging residues. Biomass production from logging residues would be widely dispersed and given historical harvest levels, it would amount to only about 2–3 green tons on an average acre. It may be feasible to economically recover this material in some locations with small chippers and chip vans. However, in the broader context of biomass markets, the economic case for producing forest biomass makes more sense when more volume is produced on a per-acre basis. Thus, our projections of biomass supplies from logging residues are combined with harvests of other low-value standing trees and these projections are discussed below.

3.2.3 LOW-PRICE BIOMASS FROM PRIVATE TIMBERLANDS

[20] One shortcoming of this approach is that it is not possible to estimate how much of this topwood and limbwood may already be utilized for products (due to differing utilization standards), or harvested for firewood.

[21] These data are based on the combination of a literature review and informal survey of industry professionals.

[22] Although we have assumed that tops and limbs are free at the landing in this case, increased competition for this material in response to higher biomass demand would likely cause the value of the wood to be bid higher, thus raising the cost of delivered wood. There are also some additional logging costs associated with piling or "putting up' the material at the landing.

At this stage of the analysis, we remain focused on biomass supplies from acres that are already under harvest for sawtimber and other industrial roundwood products. We restrict the potential for forest biomass to this footprint because of our assumption that biomass stumpage prices remain near recent levels. As shown in Exhibit 3-5, stumpage prices for forest biomass chips averaged only $1–$2 per green ton in southern New Hampshire in 2008 and 2009. Prices were lower than this in western Massachusetts, but higher in Maine. At these price levels, there will be little incentive for landowners to bring additional acres into production. Historically (at least for the past several decades), timber harvests in Massachusetts have been driven by the demand for sawtimber[23] and in this scenario, this continues to be the case.

Exhibit 3-5: Average Cost of Fuel Grade Chips in Southern New Hampshire

Dollars per Green Ton			
	Delivered	**Stumpage**	**Difference**
2005	$18	$0.8	$17
2006	$23	$0.8	$22
2007	$22	$0.9	$21
2008	$32	$1.2	$31
2009	$30	$1.6	$28

Source: Compiled from average quarterly prices as reported by the New Hampshire Timberland Owners Association's Market Pulse and reported in the Timber Crier magazine.

If the demand for biomass fuel increases in response to an expansion in bioenergy plants, how much "new" biomass could be harvested economically from areas already under harvest for sawtimber in Massachusetts? There are three analytical tasks involved in this projection. First, we address the issue of harvesting costs in Massachusetts: if new biomass demand originates from electric power plants, it would almost certainly be accompanied by an increase in whole-tree harvesting; thus, we start with an analysis of these costs. As shown in Exhibit 3-5, delivered prices for fuel grade chips were about $30 per green ton in 2008–2009 and we are assuming that biomass producers must be close to that target for electric power plants. If new biomass demand originates from thermal and CHP plants, they can pay higher prices for wood chips and thus have the option of using alternative logging methods; in addition, they will be competing for bolewood because of their need for higher-quality chips. Second, we consider the issue of how much low-value timber (that is, timber with low stumpage prices) is available on typical stands that are being harvested for sawtimber? Once we have established how much low-value wood is available and the cost of harvesting it, we then consider whether landowners would be amenable to these higher harvest levels. Using this information, we conclude this section with a projection of how much forest biomass supply would be available at current energy prices.

23 According to Forest Cutting Plan reports for 1984–2003, 95% of harvests included sawtimber.

3.2.3.1 Costs of Whole-Tree Harvesting

In whole-tree harvesting systems, trees are felled by either mechanical or manual means and moved to a landing with most or all of their tops and branches. For our analysis, the costs of whole-tree harvesting in Massachusetts are important because low-value trees that are cut only for biomass chips have to bear the full variable costs of the harvest. If a logging operation is arranged to include biomass chip production, some portion of the cost of getting equipment to the site and setting up operations should also be covered by biomass. These fixed costs are one reason that production volume is an important economic variable in determining the profitability of biomass harvests.

In order to estimate the costs of whole-tree harvesting in Massachusetts, we have conducted a large number of simulations with the Fuel Reduction Cost Simulator.[24] Our main interest in this analysis is to understand the relationship between tree size and the chip production costs because it commonly stated that pre-commercial thinnings and small trees can make a significant contribution to forest biomass supply. This model can also be used to analyze the relationship between chip production costs and a host of other factors such as block size and skidding distance. [25]

We designed this analysis to determine the cost of producing 25 green tons of wood chips on one acre (this volume is based on our analysis of availability in the next section) using different combinations of the size and number of trees.[26, 27] The results are presented in Exhibit 3-6. Although these parameters will

24 The Fuel Reduction Cost Simulator (FRCS) was developed by the U.S. Forest Service (Fight et al., 2006) to estimate the costs associated with fuel reduction treatments in harvests of whole trees, logs, and chips with a variety of harvesting systems. Although originally developed for forests in the Northwest, the model has been subsequently expanded to other regions (including the Northeast) by Dennis Dykstra and is available on the U.S. Forest Service website at: www.fs.fed.us/pnw/data/frcs/frcs.shtml

25 Our analysis in Task 5 has also utilized this model as a key source in developing estimates of diesel consumption as a component of the life-cycle analysis.

26 Assumptions made so that conditions would be representative of average conditions of Massachusetts include: a) harvest block size of 50 acres, and thus an average skidding distance of 600 feet; b) terrain sloped 5%; c) species mix evenly distributed between softwood and hardwood.

27 We also assumed no move-in costs simply to avoid the issue of how these costs should be shared with sawtimber operations. Move-in costs depend directly on the total tons produced from a given logging operation. In our simulations, producing 25 green tons on 50 acres (1250 tons total) results in move-in costs of $1-$2 per green ton (assuming a 15-mile move) if there is no complementary sawtimber/pulpwood harvest to share the expense. If 25 green tons are produced on 25 acres, then move-in costs per green ton remain about the same because the doubling in fixed costs is approximately offset by the reduction in skidding costs due to shorter hauls.

differ by individual site, logging equipment, harvest layout and many other factors, we believe our general conclusions are robust.

Exhibit 3-6: The Influence of Tree Size on the Cost of Chips ($/GT, FOB Truck, at Landing) Using Mechanical and Manual Whole-Tree Harvesting

DBH, in	Height, ft	# Trees*	GT/Tree	Mech WT	Man WT
3.0	25	980	0.03	$92	$160
5.0	35	287	0.09	$51	$63
7.0	45	92	0.27	$26	$28
9.0	55	46	0.54	$19	$21
11.0	60	30	0.85	$16	$17
13.0	65	21	1.22	$14	$13
15.0	70	15	1.63	$13	$11

*Notes: * "# Trees" denotes the number of trees at each diameter and height that are required to yield 25 green tons of chips.*

In these calculations, mechanical harvesting uses a drive-to-tree feller-buncher and grapple skidder. Manual harvesting uses chainsaw felling in combination with chokers and cables to skid unbunched trees.

The model suggests that the minimum size threshold for whole-tree harvesting in Massachusetts is in the range of 7.0–9.0 inches DBH if the economic objective is to deliver chips to a bioenergy plant at a cost of about $30 (or less) per green ton. In addition to harvesting costs, this estimate allows for: 1) $1–$2 per green ton for biomass stumpage; 2) $8–$12 per green ton for truck transport to the bioenergy plant; 3) recognition of the potential range in model estimates due to site-specific factors and modeling errors.[28] It is important to note that these estimates include machinery and equipment costs. While lower delivered prices may not attract new investment in machinery and equipment, those who already have equipment may choose to operate if they are able to cover only their variable costs of production.

Costs rise exponentially when tree sizes decrease below this level because of the exponential relationship between tree diameter and weight. For example, it would take about 40 trees that are 3-inches DBH to produce one ton of green chips, and thus it would take almost 1000 trees to generate 25 tons of green chips. The number of trees required for 25 green tons could be reduced to about 100 at 7-inches DBH and to only 10 trees if tree DBH was 18 inches.

[28] Modeling errors can arise from many sources. For example, on the fixed cost side, key areas of concern would be the choice of equipment and the calculation of ownership costs for situations in Massachusetts. On the variable cost side, wage costs and diesel costs are important parameters that may vary significantly over time and for different operations.

We also tried to estimate the costs of such logging operations on the basis of a literature review. Available studies show wide variation in costs due to factors such as species, size, quality, terrain, and harvesting equipment: the range extends from about $20-to-$50 per green ton. However, without information that links harvesting costs to timber size, it is not possible to put these estimates in our context. It seems that pre-commercial thinnings and small trees should be excluded as part of the biomass resource in Massachusetts—as one logger in Maine told us anecdotally, "the fastest way to go broke in the biomass business is to harvest 2-to-6 inch trees."

These model results clearly demonstrate the critical importance of tree size and handling costs in the economics of whole-tree harvesting: whole-tree harvesting appears to be cost prohibitive for sapling-size trees. In addition, manual harvesting is much more expensive than mechanical in the small-diameter classes primarily due to the high costs of gathering and skidding unbunched trees. However, the cost curves for these two whole-tree systems converge (and eventually cross) as tree diameter increases. This may be important for management plans on some forests because the two systems will have different impacts on soils and harvest sites.

There are a variety of other harvesting systems that could be employed in removing forest biomass. Thermal and CHP plants often demand higher-quality chips than electric power plants and can pay more for delivered wood; thus, more harvesting options are available for procuring their wood supply. Log-length methods may be selected instead of whole-tree methods if the manager or operator wishes to leave tops and limbs scattered on the site and/or is concerned about residual stand damage (to both soils and standing trees). Two common log-length methods that could be used are cut-to-length (in which mechanized harvesters are used to fell, delimb, and buck trees at the stump) and manual systems (in which chainsaws are used to fell, delimb, and buck trees at the stump) (Fight et al., 2006). Logs can then be debarked and chipped at the landing, or transported to a plant and processed there. Using the FRCS model, we have estimated that these harvesting systems will add about $10–$15 per green ton to the cost of delivered chips.

In future decisions regarding the choice between mechanical and manual harvesting systems, labor issues also are an important consideration. As labor costs rise and the labor force ages, there will be a preference for mechanized harvesting to reduce overall labor costs (including improving safety and reducing insurance premiums for health, liability, and worker's compensation). Labor costs have been identified as having an important role in increasing mechanized harvesting—both whole-tree and cut-to-length—in some regions.

3.2.3.2 The Availability of Low-Value Wood in Massachusetts Forests

The Low-Price Biomass scenario assumes that biomass stumpage will be available for $1–$2 per green ton, which is generally the price we see throughout markets in New England. Here we provide a broad overview of the volume of wood in Massachusetts forests that might be available at such low prices.

Approximately 65% of the standing trees on Massachusetts timberland are 1"–5" DBH; however, in spite of their large numbers, these sapling-size trees represent only 5% of the timber volume on a tonnage basis (FIA Statistics for 2008). It would be cost prohibitive to harvest trees in this size class based on our analysis. In order to be competitive in current markets, biomass producers would need to harvest trees with low stumpage value that are greater than 5" DBH.

As discussed earlier, sawtimber harvests are crucial in opening timber stands to biomass production. In Massachusetts, sawtimber harvests will typically take place in stands that are 60-to-100 years old, and FIA data for 2008 indicate that these stands account for 80% of total growing stock volume. Thus, these age classes are by far the most important in identifying the availability of low-cost wood.

Exhibit 3-7 presents the total volume and volume per acre for timber stands classified in the 61–100 year age class in Massachusetts.[29] The key groups that are potential sources of biomass potential are: 1) rough cull trees, with 8% of the average stand volume; 2) grade 4 & 5 trees, with 16% of the volume; and 3) pulpwood trees,[30] with 21% of the volume. As reported in this table, the combination of these three groups totals 59 green tons per acre.

Exhibit 3-7: Timber Volume by Tree Grade, Age Classes 61–100 Years in Massachusetts (All Timberland) 000 Acres and Million Green Tons, 2008

	Quantities	Share	GT / Acre
Acres (000's)	2,120		
Total Volume (millions)	273.2	100%	129
Grades 1 & 2	76.4	28%	36
Grade 3	67.9	25%	32
Grades 4 & 5	44.7	16%	21
Pulpwood	57.8	21%	27
Rough Cull	23.0	8%	11
Rotten Cull	3.5	1%	2

Note: FIA data; include all live volume (merchantable volume, tops, limbs, and stumps) in trees ≥ 5 inches DBH.

29 These volumes represent total tree biomass, not just bole volumes. Since we are not interested in total volumes for individual ownerships, we have combined the data for private and public lands to obtain more accurate estimates of grade shares and per-acre volumes.

30 Pulpwood is defined as 5"–9" DBH for softwood trees, and 5"–11" DBH for hardwoods.

These data provide only a starting point and need several adjustments before they can serve as a useful upper bound for potential biomass supply. About 30% of grade 4 & 5 trees are greater than 25" DBH; it is not practical to harvest these trees with standard equipment. On the opposite end of the spectrum, about 20% of the pulpwood trees are less than 7" DBH and we exclude half of these (those that may be in the 5"–6" range) because of their higher harvesting costs. Finally, as discussed earlier, some poletimber-size trees are already being harvested for pulpwood/fuelwood end uses; these total about 10 green tons per acre (when adjusted to a comparable basis with the inclusion of tops and limbs).

With these adjustments, the availability of grade 4 & 5 trees is reduced from 21 to 15 green tons per acre; pulpwood is reduced from 27 to 12 tons per acre; and rough cull remains at 11 tons per acre; hence, the revised total of available biomass is 37 green tons per acre. At the risk of appearing overly precise, we should recognize that this timber will continue to grow: if we assume the volume increases by an average net annual growth rate of 2% per year for 7½ years to reflect the average availability in 2010–2025, timber availability rises to 43 green tons per acre.[31]

This review characterizes the potential availability of biomass in broad terms of value and economic accessibility, but there is still a good deal of uncertainty in defining what share of this volume would be available at very low stumpage prices. At this level of aggregation, there is no straightforward way to address this, but it would be reasonable to assume that not more than half of low-grade sawtimber and poletimber could be purchased and harvested at low stumpage prices. This would reduce available supply to the range of 20–25 green tons per acre. On the basis of the information and assumptions presented above, we think that 15 green tons per acre is a good "ballpark" estimate of incremental whole-tree biomass potential—we also consider 20 green tons per acre as a potential upper bound.

3.2.3.3 Landowner Willingness to Harvest

We have identified a significant volume of low-value wood in Massachusetts that could be harvested at low cost, at least with whole-tree harvesting systems. The question that remains is: if the demand for forest biomass from private timberlands in Massachusetts increases (from bioenergy plants established in Massachusetts, nearby states, or overseas), what is the likelihood that we would see increased biomass harvests in conjunction with sawtimber operations? Would landowners be receptive to these changes? In many cases, there could be strong economic incentives, even though they would not be the result of direct, immediate income in the Low-Price Biomass scenario.

While there is a tendency to use landowner surveys to highlight the lack of interest in timber production in Massachusetts, there is a flip side to this viewpoint. Every year, an average of 22,300 acres of private timberland in Massachusetts is harvested,

31 Increasing the available volume for growth has the same effect as the inventory variable in standard economic models of timber supply.

primarily for sawtimber. More than half of the private acreage in Massachusetts (1.2 million acres) is held in parcels that are 50 acres or larger (Butler, 2008).[32, 33] Owners of 40% of the family forest land (about 650,000 acres) reported that a commercial harvest—sawlogs, veneer logs, or pulpwood—occurred since they acquired the land.[34] The large majority of these owners stated that they harvested trees because the trees were mature and/or they wished to improve the quality of the remaining trees. Suffice to say, while timber production is certainly not the number one priority on most private forest land in Massachusetts, there is a significant component of the forest land base in Massachusetts that is used to generate timber income and would likely be available for more aggressive forest management under the right circumstances.

There are landowners who would like to pursue forest management practices that will enhance the growth of their forest for future commercial timber production. With no market for biomass, these owners need to pay loggers for the cost of harvesting and collecting low-value wood and then may have an additional cash outlay for slash disposal. This could be a substantial investment with a return not seen for many years. However, with a "new" market for biomass fuel, the prices for delivered biomass may be sufficient to cover logging costs and may go beyond break-even to generate positive stumpage values for this material. Thus, harvesting of forest biomass could open the door for alternative forest management practices that are focused on improving sawtimber growth and value.

3.2.3.4 A Forecast of Forest Biomass Supply in Massachusetts with Low-Price Biomass Stumpage

Here we combine the information above to forecast how much "new" forest biomass could be supplied if demand from bioenergy facilities increases while real biomass stumpage prices remain at recent levels. The forecast is intended as an upper limit in the sense that any volume less than this could be produced to meet the demand from bioenergy plants at similar prices.

This projection is predicated on several key assumptions:

- The total land area harvested remains at the historical average.
- One half of this area is managed as it has been in recent years. The same volume of sawtimber and other industrial roundwood will be harvested and no logging residues are harvested for biomass because such operations are not justified by the economics (due to scattered material which is costly to harvest and low volumes per acre). Due to the low level of pulpwood stumpage prices, it is possible that some of this material could be diverted to biomass fuel, but we have not included this potential shift as part of the Low-Price Biomass scenario.
- The other half of the land area harvested receives silvicultural treatments that include whole-tree biomass harvesting.[35] While many landowners will find this management option suitable for their objectives, many others will not look favorably upon heavier logging of their woodlots.
- On the acres that are harvested more intensively with whole-tree methods, 65% of tops and limbs removed for industrial roundwood production are harvested for biomass. (As noted above, pulpwood is assumed not to be diverted to biomass in this scenario.)
- For whole-tree biomass harvests, 15 green tons are cut per acre. Of this volume, 10% is left on the harvest site for ecological reasons (this is equivalent to 1/3 of tops and limbs).

Projections for this biomass harvest scenario are shown in Exhibit 3-8. Land is classified as "½ Current" (land harvested as in recent years) and "½ WT" (land harvested with whole-tree harvesting). Removals per acre average 21.8 green tons in "½ Current," compared to 36.8 green tons in "½ WT," so the removals per acre average 29.3 green tons statewide (compared to 21.8 tons with no additional biomass harvesting). Total forest biomass fuel harvested averages 16.5 green tons per acre in "½ WT," and 8.3 green tons per acre for all private lands in Massachusetts. On the acres where biomass is harvested, 13.5 green tons come from whole trees, while 3.0 green tons consist of residues from sawtimber/pulpwood harvests.

As shown in Exhibit 3-8, this scenario results in 184,000 green tons of additional biomass produced for bioenergy on private lands in Massachusetts. If we increase the biomass removal rate to 20 green tons per acre, the biomass harvest increases to 235,000 green tons. The availability of low-value stumpage (timber that will be sold for only $1–$2 per green ton) and the implications

32 Landowner survey results show that only 43% of the 1.7 million acres that are family owned are 50 acres or larger; however, 88% of the remaining 0.4 million acres held by private owners belong to this size class.

33 The National Woodland Owner Survey provides a substantial of information intended to characterize the behavior of private forest owners in the United States. The main report summarizing these data is Family Forest Owners of the United States, 2006 (Butler, 2008). An on-line version—NWOS Table Maker Ver 1.01—provides users with the ability to create their own customized tables for individual states.

34 Among survey respondents, 25–30 years seems like a reasonable approximation of the average ownership tenure for family-owned land (measured by area, not number of owners): the ownership tenure was 25–49 years for about 40% of the family-owned acreage and 10–24 years for about 30% of the acreage.

35 This assumption is consistent with an electric power demand scenario. It can be easily modified for thermal or CHP demand. We would assume that stumpage prices remain at the same level—thermal and CHP could pay more for stumpage but there is no reason to do so unless competing for higher-value timber. The main difference would be that if loggers do not use whole-tree methods, then tops and limbs would be excluded from the harvest volumes.

for removal rates is one of the key assumptions in this scenario. Further analysis of these removal rates is provided below.

The share of land assumed to be harvested using whole-tree methods is also a critical assumption in this scenario. The relationship between biomass production and this share is linear in our formulation since we are working with "average" acres. Thus, if whole-tree harvesting and increased harvesting intensity were used on only one-quarter of all private lands being harvested commercially, production of biomass for bioenergy would be reduced to 92,000 green tons; similarly, if these practices were extended to all commercial harvests on private lands, biomass production would increase to 368,000 green tons.

In the next section, we review related data from nearby states to provide some perspective on these estimates of forest biomass production for Massachusetts. The data from nearby states give us some confidence that our forecasts are in the appropriate range; however, it is difficult to say for sure without more detailed analysis of timber sales and more experience with biomass harvesting in Massachusetts.

Exhibit 3-8: Biomass Supplies Available from Massachusetts Private Lands under the Low-Price Biomass Scenario

3.2.3.5 The Experience in Nearby States

It is useful to consider this outlook for whole-tree harvesting with respect to other states in New England where whole-tree harvesting is now more extensive than in Massachusetts and has a much longer history, and thus might be considered to be in a mature phase. Maine and New Hampshire, with relatively large forest products industries and well-developed wood-fired power plant sectors, may represent the potential for whole-tree harvesting when the industry pursues more aggressive harvest yields with mechanization. State harvest reports indicate the following: in Maine (Maine Forest Service, 2009), forest biomass chips comprised 23% of the total harvest of roundwood products in 2008 (3 million green tons out of a total harvest of 13 million green tons); in New Hampshire (New Hampshire Report of Cut, 2008), the comparable share was 24% in 2000–2006 (790,000 green tons out of a total harvest of 3.2 million green tons, on average). Whole-tree harvesting is not practiced to the same extent in Vermont (Vermont Forest Resource Harvest Summary, various years), where forest biomass chips represented an average of 13% in 2000–2006 (200,000 green tons out of a total harvest of 1.5 million green tons, on average).

Annual Rates, 2010–2025 (Green Tons and Acres)				
	Current	Low Biomass Price		
	Harvest	½ Current	½ WT	Total
Area Harvested (acres)	22,300	11,150	11,150	22,300
Wood Removals	Green Tons per Acre			
Industrial Removals	21.8	21.8	21.8	21.8
Roundwood Harvest	17.1	17.1	17.1	17.1
Logging Residues Generated	4.7	4.7	4.7	4.7
Left on Site	4.7	4.7	1.6	3.2
Harvested for Biomass Fuel	0.0	0.0	3.0	1.5
Whole-Tree Biomass Removals	0.0	0.0	15.0	7.5
Whole-Tree Harvest	0.0	0.0	13.5	6.8
Logging Residues Left on Site	0.0	0.0	1.5	0.7
Total Removals	21.8	21.8	36.8	29.3
Total Biomass Harvest	0.0	0.0	16.5	8.3
Wood Removals	000's of Green Tons			
Industrial Removals	485	243	243	485
Roundwood Harvest	381	191	191	381
Logging Residues Generated	104	52	52	104
Left on Site	104	52	18	70
Harvested for Biomass Fuel	0	0	34	34
Whole-Tree Biomass Removals	0	0	167	167
Whole-Tree Harvest	0	0	151	151
Logging Residues Left on Site	0	0	17	17
Total Removals	485	243	410	652
Total Biomass Harvest	0	0	184	184

Notes: "Current Harvest" is a projection assuming that commercial harvests continue at average levels of the past several years and there is no additional harvesting for biomass. With the increased harvest in the Low-Price Biomass scenario, one half of acres are assumed to be managed in the same way as in the Current Harvest Projection ("½ Current"), and one half of acres are assumed to be managed more intensively using whole-tree harvesting techniques ("½ WT").

For Massachusetts, our Low-Price Biomass scenario (assuming removal of 15 green tons in silvicultural treatments with biomass) yields a harvest share for forest biomass chips of about 33% (this figure includes whole-tree chips from tops and limbs produced in harvesting industrial roundwood). Thus, relative to the northern New England experience, it appears that our scenario would represent a reasonable upper bound for expected outcomes. With assumed biomass removal rates of 20 green tons per acre, the forest biomass harvest share in Massachusetts would increase to 38%, which would seem high, particularly when considered in the context of differences in parcel size, attitudes, and social factors among the states. However, this share will depend on other factors that could favor a higher share in Massachusetts including: the availability of low-value timber on forest stands that are being harvested; and, the extent of alternative outlets for pulpwood along with the relative strength of demand and prices for pulpwood and biomass fuel. Given these uncertainties, we have reported the likely biomass harvest as a range from 150,000 to 250,000 green tons per year, thus spanning the estimates (184,000 and 235,000 tons) provided above.

3.2.4 HIGH-PRICE BIOMASS FROM PRIVATE TIMBERLANDS

How much would forest biomass supplies increase if bioenergy plants could pay higher prices for stumpage? As demand and prices increase, more wood can be supplied from private lands by increasing the volume of wood removed from sites that are already under harvest for industrial roundwood, diverting wood from other end-use markets (such as pulpwood) to biomass, and increasing the number of acres being harvested. This scenario is intended to provide perspective on the upper bound for forest biomass production if bioenergy demand and prices increase beyond the level established in the Low-Price Biomass scenario. It is not reasonable to specify an absolute maximum for biomass supply since supply is an economic concept that depends on timber prices (and a host of other factors). Thus, we need to specify a "high" biomass stumpage price, and then consider how private landowner harvests might respond to this price level. Forest biomass volumes could still increase beyond this level, but it would be increasingly difficult to due to biophysical, economic, and social constraints and increasingly unlikely due to macroeconomic and energy constraints.

The amount that bioenergy plants can afford to pay for wood is a function of the prices they receive for their output. In order to determine a biomass stumpage price in this limiting case, we have assumed that the increase in demand for biomass comes from an expansion in electric power capacity (this assumption does not, however, restrict the usefulness of these results for other types of bioenergy). We have considered several electric price scenarios and selected $20 per green ton as the real biomass stumpage price that would reflect the high end of projections for electricity prices.

A biomass stumpage price of $20 per green ton would be consistent with a significant increase in the price of electricity. Although we have not modeled the dynamics of the harvesting and transport sector, it would be reasonable to assume that these costs would also increase in the near term due to the limited supply of loggers, foresters, machinery, and equipment; thus, delivered wood prices would likely rise well above $50 per green ton. However, we would anticipate that harvesting and transport costs would subsequently retreat with increasing competition and new investment in harvesting machinery and equipment. If these increases in wood costs were fully incorporated into the price of electricity, the impact would be as follows: a $20 per green ton increase in delivered wood prices (from $30 currently to $50) would equate to an increase of 3.2 cents per Kwh; delivered wood prices of $60 per green ton would translate to an increase of 4.8 cents per Kwh; and $70 per green ton would equate to an extra 6.4 cents per Kwh.

There are a variety of other scenarios that could lead to the production of much higher volumes of forest biomass fuel supplies. A key factor distinguishing these scenarios are those in which exogenous factors affect biomass demand directly (examples would be increasing energy production or high export demand for biomass fuel) and those that stimulate other commercial timber production (examples would be housing policy or local product promotion) and increase biomass production as by-product. Generally, biomass prices will rise in cases where there is direct demand stimulus; however, if biomass production rises as a by-product of expanded sawtimber production, biomass prices will remain low. We have assumed that higher biomass demand drives this scenario for two reasons: 1) we are primarily interested in energy policy, and whether forest biomass supplies would be adequate to support an expansion of bioenergy capacity; and 2) the probability of a substantial increase in sawtimber production seems fairly remote.[36]

There are several issues that need to be considered in gaining an appreciation for how much biomass could be harvested from private lands in Massachusetts if biomass stumpage prices were to rise substantially. These include:

- How large is the operable land base, or in other words, how much land should be excluded from potential harvesting due to biophysical constraints or lack of landowner interest in timber production?

36 Although lumber production is likely to recover from the recent downturn, we are aware of no studies that project the lumber industry in this region (or in the U.S. North in general) to move above the trend levels of the past decade. Although the sawtimber inventory is rising in Massachusetts, there appear to be few other competitive advantages that would promote an expansion of the sawmilling industry: 1) maturing timber has not resulted in increasing sawtimber harvests in the past two decades; 2) sawmills are closing in Massachusetts, not expanding, and lumber capacity has contracted sharply over the past decade; 3) there are questions about sawtimber quality due to age and years of partial cutting for sawtimber production; 4) there is plenty of "cheap" timber in competing areas of North America and the world and this is especially true over the coming decade due to delays in timber harvesting that have occurred as the result of the housing debacle of 2007–2010.

- What is an appropriate harvest schedule for these lands, or over what period might we expect initial harvests to begin and for these lands to be brought under management?
- What share of this land is likely to be drawn into production at different price levels? Harvesting these lands is not an all or nothing proposition, so here we consider how landowners may respond to higher biomass prices and the higher income they may receive from such harvests.

After discussing each of these factors, we provide a forecast of biomass supplies at much higher demand and price levels. We then review some key areas of uncertainty and provide some sensitivity analysis for important assumptions.

3.2.4.1 Estimation of the Size of the Operable Private Forest Land Base in Massachusetts

As shown earlier, the area of private land harvested in Massachusetts has been very stable over the past 15 years, and has not exceeded 25,000 acres during the 25 years for which we have data. This sort of stability would be consistent with a regulated forest where each age class has the same number of acres. However, this is far from the case in Massachusetts, which would be better described as an even-aged forest due to the high concentration of timber in a few age classes: Exhibit 3-9 indicates that about 50% of the acreage on private lands in Massachusetts is in the 61–80 year stand-age grouping (according to Kelty et al., 2008, this is about the age that the first partial thinning is done by most owners interested in harvesting timber). Much of the standing timber inventory in Massachusetts can be considered already mature or approaching maturity; in fact, natural mortality exceeds removals according to the FIA data for 2008.[37] These age-class data suggest that with higher demand and higher prices, harvesting activity could increase and break out of the stable pattern seen historically.

Exhibit 3-9: Number of Timberland Acres by Age Class, Private Land Owners, 000's (2004–2008)

Age Class	Acres	Percent
0–20	24	1%
21–40	69	3%
41–50	142	7%
51–60	202	10%
61–70	529	26%
71–80	507	25%
81–90	373	18%
91–100	101	5%
100–120	60	3%
120+	18	1%
TOTAL	2,026	100%

Source: FIA data.

[37] Although these differences are not statistically significant given the large sampling errors associated with both removals and mortality.

In order to estimate the size of the operable land base on private lands, we rely on a variety of studies and a growing body of research on landowner behavior and factors that affect willingness to harvest. Our general approach, which has become fairly standard, is to reduce the total land area to account for: 1) physical land attributes that limit logging access; 2) small parcels that have a low probability of being harvested due to economic and social factors; and 3) lack of landowner interest in producing timber due to the higher value of nontimber benefits.[38]

Physical factors appear to be relatively unimportant in limiting harvesting activity in Massachusetts. A study by Butler et al. (2010) indicated that 6% of the land in family-forest ownership should be considered unavailable due to biophysical restrictions (primarily slope and hydric physiographic class). Kelty et al. (2008) assumed 7% of forest land was off limits to logging based on a review of forest plans for the Quabbin state forest. For our scenarios, we have reduced the private land area by 5% to account for these factors, and have done so assuming that the restrictions are distributed equally across all groups and size classes.

Our next step is to eliminate parcels of small size. The rationale for their removal is twofold: 1) the attitudes of owners holding small parcels, who tend to be focused on forest benefits other than timber income; and 2) the relatively high costs of wood production on small parcels, which becomes much more important when whole-tree harvesting of biomass fuel is considered. The distribution of acres across ownership size classes is presented in Exhibit 3-10.

Exhibit 3-10: Number of Acres Held by Size of Holdings, Private Land Owners, 000's (2002–2006)

Acre Class	Family	Other	Total	Percent	#Owners
1–9	562	0	562	26%	261
10–19	208	0	208	10%	17
20–49	187	61	248	11%	8
50–99	250	62	312	14%	4
100+	479	370	849	39%	3
TOTAL	1,686	493	2,179	100%	293

Notes: Data are from Family Forest Owners of the United States, 2006 (Butler, 2008). Family owners are defined as "families, individuals, trusts, estates, family partnerships, and other unincorporated groups of individuals that own forest land." Other private owners are industry, corporations, clubs, and associations.

[38] We should note that we have not adjusted the total land area for land clearing and conversion. If forest land clearing continues at recent historical rates (which we discuss in more detail in Section 3.5.1), this would mean a reduction of about 70,000 acres of private forest land (only 3% of the total) over the next 15 years. However, as noted earlier, this number could be much larger historically (and going forward), but it is difficult to measure the magnitude of the shift accurately and to document the exact causes of land use changes. However, this shift clearly becomes of greater consequence over a longer time horizon. In addition, land clearing is linked to trends in land fragmentation which has important implications for wood supply.

Analysis of landowner attitudes leads to the conclusion that interest in timber production is highly correlated with size of forest holdings, and most owners of small parcels choose to own forest land for reasons other than wood harvesting (although they are often interested in obtaining fuelwood for their own use). For example, for the land held in parcels less than 10 acres, a large majority of the land would not be logged or there would be "minimal activity to maintain forest land" during the next five years, while all respondents said they would not harvest sawlogs or pulpwood.[39]

Butler et al. (2010) suggest that the minimum operable size for timber harvesting may now be about 15 acres, and might be increasing into the range of 30 acres, based on studies that have evaluated the economies of scale associated with modern harvesting equipment. Surveys of minimum economical scale for whole-tree harvesting in Vermont among different stakeholder groups provided responses that were concentrated around 800 green tons per logging operation (Sherman, 2007). Average responses by group were: foresters, 27 acres at 12 cords per acres (810 green tons); logging contractors, 23 acres at 14 cords per acre (805 green tons); chipping contractors, 15 acres at 21 cords per acre (788 green tons). These data suggest that removing an average of 25 green tons of the wood on an acre would require a logging site of at least 30 acres.

Using the information on both landowner attitudes and economies of scale, we have excluded parcels less than 20 acres from the operable land base. While there seems to be evidence that the harvest threshold may now be above this level, we have tried to be conservative in an effort to establish an upper bound to the operable harvest base. In addition, this lower level allows for the use of current equipment and harvesting methods that may be suitable for smaller-scale production for thermal and CHP plants.

Another reason that this threshold is likely to be "conservative" and tend to overstate the amount of land available for harvesting and biomass production is that we have not attempted to project changes in the distribution of land ownership by parcel size in the future. There have been significant reductions in average parcel size historically (Kittredge, 2009). Perhaps more importantly for our analysis, projections suggest that there are likely to be significant increases in private forest land development in central and southeastern Massachusetts from 2000 to 2030 (Harvard Forest, 2010). However, as noted with land clearing, it is difficult to quantify these developments and they are more critical for long-term projections than over the next 15 years.

The final adjustment to the land base relates to landowner attitudes of those who hold parcels that are greater than our threshold of 20 acres. Surveys of family forest owners indicate that those who hold parcels greater than 50 acres also place high value on benefits other than commercial timber production. For example, when asked about their management intentions for the next five years, owners of 56% of the land said they would do nothing or engage in minimal activity as compared to 43% who planned to harvest sawlogs. In response to their reason for owning their land, 71% (again, based on acreage) said for beauty and scenery, 51% said for privacy, and only 34% said to produce sawlogs or pulpwood. At the same time, although timber income is not a primary motivation for owning land, it is still important as owners of 66% of the land reported having a commercial harvest on some portion of their land during their tenure. (All data are from the National Woodland Ownership Survey, on-line data, Butler et al., 2008.)

Based on these survey data, we have reduced the available area of family-owned forest parcels that are greater (or equal to) 20 acres by 20%, which believe is conservative. We have assumed the same adjustment is appropriate for landowners in the "other private" category.

A summary of the results from our process of netting down the private land area to obtain the operable land base is shown in Exhibit 3-11. Our methodology and assumptions reduce the total private land base by 51%, thus leaving 1,071,000 acres of private land available for harvesting in Massachusetts. It is interesting to compare these results with two other studies for Massachusetts that use similar methods, but different assumptions. Kelty et al. (2008) provides two scenarios of private land availability: the higher has 1,072,000 operable acres when 10 acres is used as a parcel size threshold (and other constraints are introduced) [40]; a second scenario with a 100-acre threshold shows only 379,000 acres available (which seems somewhat extreme compared to our calculations). Butler et al. (2010) estimate that biophysical and social constraints on private lands might reduce the wood available from family-owned forests by 68% (we show a 59% reduction for the family-forest category). That study also uses a 20-acre threshold, but assumes a much larger reduction due to social constraints.

Exhibit 3-11: Private Land Area Available for Timber Harvesting in Massachusetts, After Deductions for Biophysical and Social Constraints 000 Acres

	Family Owners	Other Private	Total
Total Timberland Area	1,686	493	2,179
Reduce for Physical Constraints (5%)	1,602	468	2,070
Reduce for Small Parcels (< 20 Acres)	870	468	1,339
Reduce for Other Social Factors (20%)	696	375	1,071
Percentage Available	41%	76%	49%

3.2.4.2 Harvest Schedule for the Operable Land Base

The above analysis provides an estimate the *total* size of the operable land base. The 22,300 acres that are already being harvested

[39] The rationale for eliminating these parcels from biomass harvesting becomes more obvious when one considers that the average parcel size in the 1–9 acre size class is only 2 acres.

[40] It is tempting to consider the nearly identical results as confirmation of the validity of one or both approaches. The two approaches are different, and the fact that the results are almost identical is coincidental.

each year in Massachusetts (and in our Low-Price Biomass scenario) are assumed to be part of this land area. In this new scenario, higher biomass stumpage prices encourage more of the landowners in the operable land base to harvest timber in any given year. How many more acres would be harvested annually? Or, put another way, what would be a reasonable time frame over which to enter these stands and initiate forest management?

We have assumed that 25 years would be a reasonable period over which bring these stands into production. The most important factor is the age structure of these stands. As shown earlier (Exhibit 3-9), the majority of the timber on private lands in Massachusetts has reached the age where it is appropriate to begin thinning based on silvicultural and economic considerations. Another important factor is that the harvest is "scheduled" to accommodate the life expectancy of electric power and other bioenergy plants—the facilities will need some assurance that wood supplies will be adequate on an ongoing basis in order to attract capital for large-scale investments.

If we assume that 1,071,000 acres are available among the private land base in Massachusetts, and that partial harvests will occur on these lands over a 25-year period, then 42,800 acres would be potentially available for harvest each year.

3.2.4.3 The Supply Curve for Landowner's Who Harvest Timber

Our analysis so far has attempted to determine the maximum operable land base, which we have defined as the land that would be harvested at much higher prices. In order to provide more perspective on how much of this land might be accessed, we need to incorporate the assumptions of our High-Price Biomass scenario (biomass stumpage prices averaging $20 per green ton). How do these owners value their nontimber amenities and at what prices would they be willing to become active players in the timber market? Would these price levels be sufficiently compelling to bring all of these lands into production?

The prices required to increase harvests significantly on private lands in Massachusetts are outside the range of recent historical experience. This is obvious from the remarkable stability in harvest levels that we have seen in Massachusetts over the past two decades. In order to assess whether this harvest stability is simply the result of limited price variation or the fact that landowners are insensitive to price swings, we have examined the relationship between timber prices (a weighted index of real red oak and white pine sawtimber stumpage prices) and harvest volumes (sawtimber harvests according to FCPs).

From 1994 to 2005, observations on prices and volumes are tightly clustered and somewhat random: the average absolute deviation from the mean is only 5% for prices and 6% for volumes. However, a much different story emerges over the last few years. From the average of 2003–2005 to 2009, planned sawtimber "harvests" fell about 30%, while real prices dropped 60%. This would suggest a price elasticity of timber supply of about 0.5, a result that is consistent with the conventional wisdom that short-run timber supply is inelastic. Of course, this calculation is merely suggestive of ownership behavior because of the quality of the data and the limited sample size.[41] Furthermore, there is no possibility to consider asymmetric behavior and to evaluate whether landowners would respond in a similar fashion if prices rose sharply.

While this result is interesting, one must also be cautious in extrapolating the conclusions much beyond the historical range: in this scenario, we are considering prices and potential landowner income that is far above historical levels. Over the 2000–2006 period, an average harvest on private lands generated about $400 per acre.[42] If we assume that 20 tons of biomass are harvested on an acre with stumpage prices of $1 per green ton, then per-acre income would rise by $20, or by only about 5%. However, if biomass prices jump to $20 per green ton, landowners could now earn an additional $400 per acre, thus doubling their income on a per-acre basis.

As biomass stumpage prices increase, we would expect that many of the owners in the operable land base would move to take advantage of the opportunity to earn more income. However, landowners possess a complex set of objectives and it is difficult to say how high prices would need to rise to induce all landowners in the operable land base to harvest biomass. It seems likely that the response would be mixed at $20 per green ton: the financial incentives would likely be too compelling for many to ignore; on the other hand, they are probably not adequate to attract many landowners who place high value on the nontimber benefits of owning forests and are not focused on timber revenue.

A final consideration in making a realistic assessment of the response in biomass harvests to higher prices, particularly in the near term, is the limitations of the labor and logging infrastructure. These would need to expand dramatically to achieve much higher harvest levels and this is another development that would be at odds with recent trends. In assessing the ramifications of this from the perspective of biomass supply, the concern is that harvesting costs may need to rise sharply to attract investment in this sector: this could mean reduced stumpage prices that would mitigate the supply response, or an increase in delivered wood prices that would choke off demand. We would anticipate that harvesting and transport costs would subsequently retreat with increasing competition and new investment in harvesting machinery and equipment.

3.2.4.4 A Forecast of Forest Biomass Supply with Higher Biomass Stumpage Prices

This outlook assumes that biomass stumpage prices rise to $20 per green ton as a result of higher demand from bioenergy plants. A

41 We should underscore this point by recalling that the FCP data report only planned harvests, not actual harvest volumes.

42 We calculated this value by assuming a harvest of 2 MBF and using a weighted average of median red oak and white pine stumpage prices for western Massachusetts from 2000–2006 (University of Massachusetts Amherst, 2008).

substantial increase in landowner income brings more land into production. Forest biomass fuel becomes a primary timber product, much as pulpwood is today, and we assume that bioenergy plants can outbid their competitors for pulpwood and low-grade sawlogs and that this material is harvested more intensively as well. It is worth noting that $20 per green ton is equivalent to prices of about $50 per cord and $100 per MBF (International ¼" log rule).

While is a good deal of uncertainty associated with many of the assumptions in this analysis, we believe that developing this forecast provides useful guidance while demonstrating many of the important factors at work. Following the presentation of the results, we provide some sensitivity analysis to key assumptions along with some discussion of the conclusions.

This projection is predicated on the following key assumptions:

- One half of the original harvest footprint of 22,300 acres continues to be managed as it has been in recent years. The same volume of sawtimber and other industrial roundwood will be harvested and no logging residues are harvested for biomass because the economics do not justify such low-volume operations. (As in the previous scenario, the pulpwood produced in this "original" share of the harvest is still assumed to be consumed in this end-use market, although it could easily be diverted to biomass fuel at the assumed price levels.)
- One half of the "original" 22,300 acres receive silvicultural treatments that include whole-tree biomass harvesting.[43] With the introduction of whole-tree harvesting on these acres, trees formerly harvested for other industrial markets are now chipped for biomass. Sixty-five percent of sawtimber tops and limbs are harvested for biomass.
- Of the remaining acreage available annually (20,500 acres, or 42,800 minus 22,300), one half is assumed to be drawn into production for whole-tree biomass harvests. The same amount of sawtimber is removed as on other lands, but all other roundwood harvested is used for biomass.
- For whole-tree biomass harvests, 25 green tons are cut per acre as higher prices increase the harvest intensity of "lower-value" wood. Of this volume, 10% of all material is left on the site for ecological reasons (equivalent to 1/3 of tops and limbs).

Projections for this High-Price Biomass scenario are shown in Exhibit 3-12, with the land classified as "½ Current" (land harvested as in recent years) and "Bal WT" (the balance of land harvested with whole-tree harvesting). Removals per acre average 21.8 green tons in ½ Current, compared to 46.8 green tons in Bal WT; removals per acre average 38.2 green tons statewide, as more acres are brought into production and harvested more intensively than in the Low-Price Biomass scenario. Total forest biomass fuel harvested averages 32.4 green tons per acre in Bal WT, resulting in an average of 21.3 green tons per acre for all private lands in Massachusetts. On the acres where biomass is harvested, 31.0 green tons come from whole trees, while only 1.4 green tons consist of residues from sawtimber harvests.

As shown in Exhibit 3-12, this scenario results in 694,000 green tons of additional biomass produced for bioenergy from private lands in Massachusetts. This represents an increase of about 510,000 green tons from our Low-Price Biomass scenario: approximately 1/3 of the additional material comes from increased harvesting of "low-value" timber and the diversion of wood formerly harvested for non-sawtimber industrial uses to biomass; the remaining 2/3's comes from new land that is brought into production. This estimate is intended to represent an upper limit for biomass fuel production in Massachusetts, given the biophysical availability of wood and our assessment of how landowners might respond in a situation with much higher biomass prices. We think this scenario provides a reasonable representation of biomass supply over the medium term with biomass stumpage prices near $20 per green ton (as noted earlier, this analysis does not account for logging and infrastructure constraints that may restrict harvesting in the near term).

There are, of course, many uncertainties in this scenario and thus some sensitivity analysis to key assumptions is important. One crucial assumption is the harvest intensity with higher stumpage prices. Our scenario shows total timber removals averaging 47 green tons an acre for harvested acres that include biomass production. This is more than twice the current average harvest of about 22 green tons per acre. Nevertheless, with biomass stumpage prices of $20 per green ton, bioenergy plants could compete for most timber on a typical stand and could probably consistently outbid lumber producers for Grade 3 sawtimber. If we raise per-acre biomass removals from 35 green tons to 50 green tons (total removals increase to 62 green tons per acre), then the biomass harvest would increase from 0.7 million tons to 1.0 million tons. A further biomass increase to 60 green tons per acre would increase the forest biomass harvest to 1.2 million tons.

Another important assumption is the percentage of operable area that is harvested at higher prices. If we increase the additional area that is brought into production from one-half to two-thirds (from 10,250 acres to 13,667 acres), then the total biomass harvest would increase to about 800,000 green tons. On the other hand, if all acres were brought into production (20,500 additional acres), then the total biomass harvest from private lands would increase to 1.0 million green tons.

Relaxing some of our assumptions increases harvest estimates to 800,000 tons and above. In order to acknowledge these key uncertainties, we have summarized our results as a range from 650,000 to 850,000 green tons. Estimation of the upper end of this range is not scientific, but simply reflects our judgment of the uncertainty in these estimates and the likelihood that harvests could be higher. Importantly, it is a reminder to use caution in using these harvest levels as point estimates.

43 As noted in our previous scenario, this assumption is consistent with an electric power demand scenario and can be easily modified for thermal or CHP demand. The main difference would be that if loggers do not use whole-tree methods, then tops and limbs would be excluded from the harvest volumes.

Exhibit 3-12: Biomass Supplies Available from Massachusetts Private Lands under the High-Price Biomass Scenario

Annual Rates, 2010–2025 (Green Tons and Acres)				
	Current	High Biomass Prices		
	Harvest	½ Current	Bal WT	Total
Area Harvested (acres)	22,300	11,150	21,400	32,550
Wood Removals	Green Tons per Acre			
Industrial Removals	21.8	21.8	12.3	15.5
Roundwood Harvest	17.1	17.1	10.1	12.5
Logging Residues Generated	4.7	4.7	2.2	3.1
Left on Site	4.7	4.7	0.8	2.1
Harvested for Biomass Fuel	0.0	0.0	1.4	0.9
Whole-Tree Biomass Removals	0.0	0.0	34.5	22.6
Whole-Tree Harvest	0.0	0.0	31.0	20.4
Logging Residues Left on Site	0.0	0.0	3.4	2.3
Total Removals	21.8	21.8	46.8	38.2
Total Biomass Harvest	0.0	0.0	32.4	21.3
Wood Removals	000's of Green Tons			
Industrial Removals	485	243	263	506
Roundwood Harvest	381	191	216	406
Logging Residues Generated	104	52	48	100
Left on Site	104	52	17	69
Harvested for Biomass Fuel	0	0	31	31
Whole-Tree Biomass Removals	0	0	737	737
Whole-Tree Harvest	0	0	664	664
Logging Residues Left on Site	0	0	74	74
Total Removals	485	243	1,001	1,243
Total Biomass Harvest	0	0	694	694

Notes: "Current Harvest"" is a projection assuming that commercial harvests continue at average levels of the past several years and there is no additional harvesting for biomass. With the High-Price Biomass scenario, one half of acres of the "original" footprint are assumed to be managed in the same way as in the Current Harvest Projection ("½ Current"), and balance of the acres are assumed to be managed more intensively using whole-tree harvesting techniques ("Bal WT").

To put these results in perspective, we have looked to the literature for estimates that may provide useful comparisons of the timber supply response. The response of harvest levels to prices is commonly measured as the timber supply elasticity. For statistical reasons, harvest response to income is not comparable to harvest response to prices. Nevertheless, a few comments on timber supply elasticities are useful. Most econometric studies have found timber supply to be very inelastic for non-industrial private ownerships. In fact, a meta-analysis indicated that of the 19 relevant studies that were reviewed, seven did not find a significant relationship between harvests and prices, that is, prices do not affect harvest decisions (Beach et al., 2003). The study also concluded that there often was not enough information in this research to compute supply elasticities (some were binary choice models). In spite of all the work and research that has been done over the past two decades on this topic, the default value for the supply elasticity that frequently appears for non-industrial private landowners is 0.3, which seems to date from Adams and Haynes (1996).

In our scenario, we have assumed that biomass stumpage prices increase to $20 per green ton. With our price and harvest assumptions, per-acre incomes about double. The High-Price Biomass scenario also shows a 50% increase in acres harvested. If we consider the landowner decision variable to be how many acres to harvest, then our results suggest that a 1% increase in income results in a 0.5% increase in harvest activity. As we have said, this "elasticity" cannot be directly compared with the timber supply elasticity; however, in terms of first-order approximations, both are inelastic suggesting that the behavior assumed for Massachusetts landowners is not inconsistent with previous research.

3.2.5 POTENTIAL BIOMASS SUPPLY BASED ON FOREST GROWTH

Previous studies of potential biomass supply in Massachusetts (reviewed in Appendix 3-A) have considered supply to be the maximum volume of low-value wood that could be harvested without reducing timber inventories below current levels. It is useful to compute this estimate to see how it compares with our estimate of biomass supply in the High-Price Biomass scenario. This also provides information as to whether our estimate is "sustainable" when using the criteria that harvests do not exceed net growth and that biomass harvests can be maintained at the same level for the foreseeable future.

The calculation of the total "sustainable" volume of biomass that can be harvested in Massachusetts depends critically on how the land area is defined and how net growth is estimated. While there are a variety of ways to make these calculations, here we follow the methodology used by Kelty et al. (2008). We define the land area as the size of the operable land base on private lands, which we have derived to be 1,071,000 acres in the previous section. For the growth rate, we use data from Chapter 5 on the average annual growth of unmanaged "mature" stands in all cover types. The average annual increase in the volume of above-ground live trees over the next 50 years is 1.3 green tons per acre. Thus, the long-term average annual growth (net of mortality) in Massachusetts would be 1.4 million green tons per year. Finally, if we reduce this estimate by 36% to account for timber that would be expected to be consumed as sawtimber (again following Kelty et al., 2008), average annual biomass availability would be 900,000 green tons per year.[44]

The upper end of our estimate of biomass supply of 850,000 green tons per year in the High-Price Biomass scenario is within the range of what would be considered "sustainable" based on the rule of harvest not exceeding growth, and thus would not result in a reduction of timber inventories across the operable land base. However, our sensitivity analysis of biomass supplies showed some projections as high as 1.2 million green tons per year which would exceed "sustainable" annual volumes as we have defined them here.

The discussion of sustainability in this context raises two important theoretical issues. One issue concerns the approach of calculating "sustainable" growth rates using initial inventory levels and fixing the time horizon in the future.[45] The majority of the timber inventory in Massachusetts is over 60 years old, and given the shape of the timber yield curves, average timber growth rates are decelerating over time. As a result, the longer the future time span that is selected, the lower the average "sustainable" growth rate. We have selected 50 years in parallel with the analysis by Kelty et al. (2008). However, the simple fact that our starting year is 2010—compared to the base year 2000 used by Kelty et al. (2008)—changes the growth trajectory enough to reduce our "sustainable" growth levels compared to their results.

The second theoretical issue concerns scale: there is no simple answer to the question of how to define the appropriate land base. If all forest land in Massachusetts were included, the total land area would jump to about 3.0 million acres and average timber growth would be about 4.0 million green tons per year. Using this theoretical approach, it would be feasible to harvest wood much more aggressively on operable private lands due to the ongoing increase in timber inventories on public lands and private lands that are not being harvested.

3.3 BIOMASS SUPPLY FROM PUBLIC LANDS IN MASSACHUSETTS

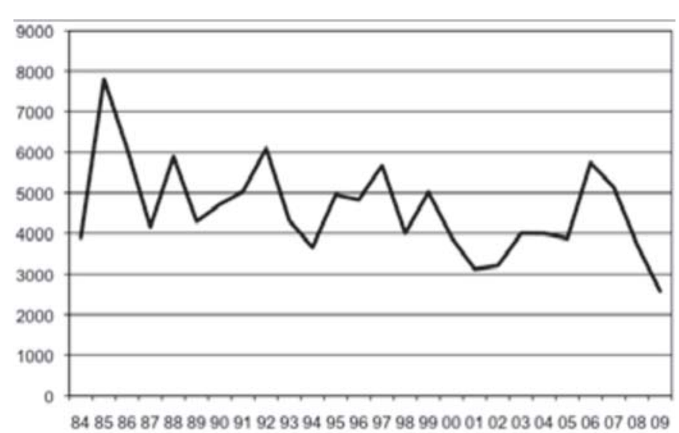
This section considers the availability of forest biomass supply from harvesting on public lands in Massachusetts. We first review estimates of historical harvest levels on all public lands and then explore these in more detail by major agency. These trends are then used to develop projections of commercial timber harvests for public lands for 2010–2025.

Using this background and perspective, we provide two forecasts of biomass supply from public lands that are consistent with our Low-Price Biomass and High-Price Biomass scenarios. As discussed previously, these are projections of incremental biomass production and do not include biomass chips that may already be counted in historical wood production totals.

3.3.1 HISTORICAL HARVEST ESTIMATES

As noted earlier, we have obtained data on Forest Cutting Plans (FCPs) for public sector lands for the period from 1984 to 2009. Exhibit 3-13 shows the number of acres targeted for harvest on public lands according to these plans. There is a general downward trend in these data: the annual average for 2005–2009 was 4,300 acres, significantly less than the average of 5,600 acres in 1984–1988.

Exhibit 3-13: Acres Planned for Harvest on All Public Lands, 1984–2009



[44] Note that this approach provides a "ballpark" estimate and does not attempt to adjust for logging residues and similar details. Estimates of biomass availability from previous studies using the "forest-growth" approach are discussed in Appendix 3-A.

[45] Another approach that is commonly used but beyond the scope of this study is to evaluate the volume of wood that could be produced if the forests of Massachusetts were brought into fully regulated management under optimal rotation ages. Such an approach would likely lead to a higher estimate of long-term timber and biomass supply.

We have assembled planned harvest data by public agency for 2001–2009 in several tables that follow. Exhibit 3-14 provides annual averages of the number of acres to be harvested, along with timber harvests of sawtimber (MBF, International ¼" rule), pulpwood (cords), and fuelwood (tons).[46] During this nine-year period, state lands accounted for an annual average of 3,092 acres, or 79% of the public area to be harvested. City and town lands accounted for 811 acres per year, or 21% of the total. The "Other" category was less than 1% of the total and consists of occasional harvests by the University of Massachusetts and the Army Corps of Engineers.

Exhibit 3-14: Summary of Forest Cutting Plans for Public Lands in Massachusetts

Area and Volumes, Annual Averages, 2001–2009				
	Acres	MBF	Cords	Tons
DCR, State Parks & Recreation	1,490	4,884	4,030	2,470
DCR, Water Supply Protection	1,454	4,873	5,069	6,766
Fisheries & Wildlife	148	465	502	450
Cities & Towns	811	2,789	2,033	1,804
Other	30	137	75	388
Total Public Lands	3,933	13,148	11,709	11,877

Harvest rates on a per-acre basis are presented in Exhibit 3-15. Among the major groups, the harvest intensity for sawtimber was very consistent, ranging from 3.2-to-3.4 MBF per acre; these compare with harvest rates of 2.0 MBF per acre on private lands. "Pulpwood" harvests averaged 3.0 cords per acre and "fuelwood" harvests averaged 2.9 green tons per acre.

Exhibit 3-15: Summary of Forest Cutting Plans for Public Lands in Massachusetts

Harvest per Acre, Annual Averages, 2001–2009			
	MBF	Cords	Tons
DCR, State Parks & Recreation	3.3	2.7	1.7
DCR, Water Supply Protection	3.4	3.5	4.7
Fisheries & Wildlife	3.2	3.4	3.0
Cities & Towns	3.4	2.5	2.2
Other	4.5	2.5	12.8
Average, All Public Lands	3.3	3.0	3.0

[46] As noted earlier, "pulpwood" is sometimes referred to as "cordwood" and likely contains a combination of wood that will be shipped to pulp mills and processed for fuelwood. Fuelwood includes both residential fuelwood that will be cut and split and wood that will be processed into biomass chips.

Per-acre harvest rates have all been converted to a green ton basis in Exhibit 3-16. Excluding the "Other" group, sawtimber harvests average 17 green tons per acre, while the total harvest per acre ranges from 25-to-30 green tons. Thus, sawtimber has accounted for 56% to 67% of the wood harvested from public lands.

Exhibit 3-16: Summary of Forest Cutting Plans for Public Lands in Massachusetts

Harvest in Green Tons per Acre, Annual Averages, 2001–2009				
	Sawtimber	Pulpwood	Fuelwood	Total
DCR, State Parks & Recreation	16	7	2	25
DCR, Water Supply Protection	17	9	5	30
Fisheries & Wildlife	16	9	3	27
Cities & Towns	17	6	2	26
Other	23	6	13	42
Average, All Public Lands	17	7	3	27

3.3.2 TIMBER HARVEST PROJECTIONS FOR 2010—2025

As with timber harvest projections for private lands, historical trends provide the starting point for this assessment. Our next step was to review the 15-year Forest Resource Management Plans for state forests, several of which have already been approved. Finally, we contacted representatives from each of the three main state divisions—State Parks & Recreation, Water Supply Protection, and Fisheries and Wildlife—to review historical cutting levels and discuss their expectations for harvests in the future.

On the basis of our review and discussions, it appears that historical averages for 2001–2009 probably provide the best estimate of acres to be treated and timber harvest volumes over the next 15 years. Information from some of the individual Forest Plans suggest that acres and harvests could be higher than we have observed historically, but it seems more likely that there will be some downward adjustments to reflect the recommendations of the Forest Futures Visioning Process (2010). There will, no doubt, be other adjustments to harvest areas and to harvest intensity and silvicultural treatments, but we do not anticipate that these will be significant enough to alter our assessment of future biomass potential.

With regard to the issue of biomass harvesting, there are at least two key factors that distinguish our analysis of potential supplies from private versus public lands. First, private landowners have the flexibility to be much more responsive to market forces and can adjust the acreages they choose to harvest as well as their silvicultural treatments. In contrast, public lands are subject to a wider array of objectives and planning issues and it is more difficult for these plans to be modified in response to changes in market demand and prices. Second, the harvest of tops and limbs will not be permitted from public lands if new management guidelines suggested by the Forest Futures Visioning Process are adopted.

Thus, once management plans have been established on public lands, undergone public scrutiny, and been officially approved by the responsible agency, it is more difficult to increase harvests in response to potential new demand from bioenergy plants. However, while the volume of wood to be harvested may be pre-determined, the ultimate disposition of the wood is not—planned harvests of pulpwood and residential fuelwood might be diverted to biomass fuel depending on demand conditions and relative prices.

3.3.3 LOW-PRICE BIOMASS SCENARIO

The economics of biomass production on private lands in Massachusetts suggest that in order to obtain sufficient volumes to furnish bioenergy plants and make logging operations profitable, it is necessary to harvest some combination of cull material, small trees, and low-grade sawtimber: the harvest of whole trees generates the volume that makes it economic to enter the stand for biomass production. Once that process is underway, then tops and limbs from industrial roundwood harvests can also be harvested for biomass.

Given the various constraints associated with harvests on public lands, we find that there is not likely to be any increase in biomass production above the levels that are already being produced for the market. (There are no estimates of the volume of biomass chips produced from public lands historically, but it is known that whole-tree biomass chips account for much of the "fuelwood" volume that is reported in tons on the FCPs.) There are several key reasons for our assessment: 1) we are not anticipating an increase in the total volume of wood harvested on public lands; on average, future annual harvest levels are projected to be about the same as during 2001–2009; 2) we are not anticipating any diversion from previous end-use markets (pulpwood, for example) because of the assumed low-price levels for biomass stumpage; 3) restrictions on the removal of tops and limbs mean that logging residues from industrial roundwood harvesting will not be available.

Thus, while there is already some production of chips on public lands, we do not project any significant increase in biomass supplies beyond recent levels.

3.3.4 HIGH-PRICE BIOMASS SCENARIO

It is likely that biomass supplies from public lands would become significant in response to a large increase in biomass stumpage prices. In this scenario, biomass stumpage prices are assumed to increase to $20 per green ton in response to higher demand from bioenergy plants. As we have noted, if the higher demand originates from electric power plants, higher electricity prices will be needed for wood-fired utilities to remain in operation. For thermal and CHP plants, it is likely they could afford wood at these prices and remain profitable.

The main vehicle for achieving the increased biomass production on public lands will be the diversion of wood from other end uses: at the projected price levels for biomass stumpage, bioenergy plants will be able to outbid their competitors for low-grade sawtimber, pulpwood, and residential fuelwood. We do not expect that forest management plans on public lands would be modified to increase the total volume of material that could be harvested on designated logging sites.

In this scenario, incremental biomass production from public lands is estimated as follows: 1) about 4,000 acres will be harvested each year; 2) all of the pulpwood harvested—7 green tons per acre—will now be chipped for biomass; 3) half of the fuelwood harvested—1.5 green tons per acre—will also be chipped for biomass (it is known that much of the reported fuelwood volume is already consumed for biomass fuel so we have assumed half simply to recognize this phenomenon). Thus, "new" biomass supplies from public lands would total 34,000 green tons per year (4,000 acres x 8.5 tons/acre).

We have assumed that the removal of tops and limbs will not be acceptable under new silvicultural guidelines for state lands. We should note that if the removal of logging residues were permissible, this would further increase biomass supplies by about 17,000 green tons, thus bringing the total from public lands to approximately 50,000 green tons per year.

We should point out that our scenarios reflect relatively light harvests on state lands relative to the volume of timber grown each year. In these scenarios, timber inventories on state lands continue to rise, resulting in rising levels of carbon storage. If the political winds on harvesting shift, these policies could be modified so that much more biomass is harvested from state lands. However, we think that such a scenario would have low probability because of the state's mandate to balance a wide array of timber and nontimber objectives.

3.4 SUMMARY OF FOREST BIOMASS SUPPLIES IN MASSACHUSETTS

The volumes of biomass available from private lands and public lands for our two scenarios are summarized in Exhibit 3-17. Importantly, we should re-emphasize that these data represent the incremental volumes of biomass that we project could be supplied in response to expanded demand from new bioenergy plants, and thus would be available to furnish these facilities.

Our Low-Price Biomass scenario was designed to evaluate the potential supplies of forest biomass that might be produced if there was an expansion in demand from bioenergy plants. This analysis was motivated by the assumption that if the increase for demand originates from wood-fired electric power plants, they will not likely be able to pay much more than the current price of $30 per green ton without significant increases in real electricity prices; thus, given the harvesting and transport costs, there is little value left for stumpage. This same volume of wood could be utilized by thermal and CHP plants—they could pay more for stumpage than the $1–$2 per green ton that we have assumed, but would not need to until demand increases to higher levels.[47] On private lands, income from biomass production is not adequate to justify bringing more land into production and biomass volumes will be limited to increasing the harvest intensity on sites already being logged for sawtimber. On public lands, we do not anticipate an increase in the incremental volume of biomass production: planned harvest volumes are not likely to be modified in response to increased biomass demand, and low biomass stumpage prices will not provide the economic incentives to divert timber from current uses to biomass chips.

Exhibit 3-17: Summary of Forest Biomass Fuel Supplies for 2010–2025

Low- and High-Price Biomass Scenarios 000 Green Tons per Year		
	Low-Price	High-Price
Private Lands	150–250	650–850
Public Lands	0	35
TOTAL	150–250	685–885

Note: Some estimates are rounded for this table.

In our High-Price Biomass scenario, total "new" forest biomass supply increases from 150,000–250,000 green tons per year to about 650,000–850,000 green tons per year. We have postulated that increases in demand from bioenergy plants drive biomass stumpage prices up to $20 per green ton, and prices in energy markets are high enough so that electric power, thermal, and CHP plants can compete for this wood. The large volume increase from private lands occurs primarily because much higher income levels provide incentives to bring more timberland into production. Public lands are also assumed to yield more biomass as relative prices cause timber to be diverted from pulpwood markets to biomass markets.

3.5 BIOMASS SUPPLY FROM NON-FOREST SOURCES IN MASSACHUSETTS

Our study has focused on biomass supplies from forest biomass sources, which include the harvesting of whole trees (including thinnings, cull, pulpwood, and low-grade sawtimber) and logging residues. These are sometimes classified as primary sources (see, for example, the *Billion-Ton Study*, Perlak et al., 2005). Wood from land clearing from development is also considered to be a primary source of wood biomass fuel in the taxonomy of the *Billion-Ton Study*. The potential volume from this source is evaluated below.

There are two other important general sources of non-forest biomass material that should be mentioned. Secondary sources ("mill residues") include any wood residues generated in the processing of logs (mill residues from sawmills, veneer mills, etc.) or lumber (manufacturing residues, from furniture, pallets, etc.). It appears that most secondary-source material is already being fully utilized in Massachusetts, and this is consistent with recent trends that show significant inflation in their prices. Tertiary sources (often referred to as "urban wood") include all other wood material and consists mainly of municipal solid waste, construction and demolition debris, and wood from landscaping and tree care. Tertiary material may potentially be a source of substantial volumes of biomass that could provide feedstock for new bioenergy plants and this source is briefly discussed below.

[47] There are several reasons (including administrative, logistical, and transport costs) that may lead some facilities to pay higher prices for biomass stumpage in their own timbershed, rather than purchase biomass from other locations where stumpage may be available at lower cost.

3.5.1 LAND CLEARING AND CONVERSION

According to a report by Mass Audubon (2009), forest land clearing and conversion averaged 4,700 acres per year from 1999 to 2005. Forest land clearing and conversion was reported at much higher levels in the previous three decades, but there are numerous inconsistencies between these data and independent data on building and construction. In addition, the new techniques and methods used in the 2005 survey (involving computer imaging and digitization) provide much finer resolution and greater accuracy in measuring land areas cleared. Given that average building permits in 1999–2005 were similar to the average levels of the past 20 years, we have assumed that recent levels of land clearing and conversion represent a reasonable estimate of land clearing for 2010–2025.

We have not been able to identify any information that would allow us to track the volume and disposition of the wood removed from these lands. It is probably safe to assume that higher-value sawtimber material is cut and sold, whereas the fate of the low-value material is much harder to predict.

Given the lack of information on these land clearing and conversion operations, it is not feasible to provide a rigorous quantitative projection of biomass supply from these sources. However, we can provide a framework for understanding the important parameters in evaluating this supply—this framework can then be used to demonstrate the biomass potential from land clearing. The potential increase in biomass supply from this source over the next 15 years will depend on: 1) the relative size of the land area cleared (future versus history); and 2) the relative rates of biomass recovery between the two periods. As noted above, we have assumed that land clearing will remain at the recent historical level of 4,700 acres per year. Thus, any increase in biomass production will require an increase in biomass recovery rates.

In order to demonstrate the potential biomass supply from land clearing, two important assumptions are necessary. The first concerns removals of sawtimber and other high-value timber for industrial products: we assume that the economics always justify harvesting this material first and for this example we assume that it accounts for an average of 36% of standing timber volume. The second assumption is the initial stocking levels of lands to be cleared and we assume that an average acre has 100 green tons of wood (this is less than the average shown in Exhibit 3-7 which applies only to stands of mature timber). Thus, the maximum volume of wood that could have been harvested for biomass in each year of the historical period—as well as in the forecast period—would be about 300,000 green tons (4,700 acres x 64 tons/acre).

At this stage, it is easy to see the importance of the recovery rate. If biomass demand increases due to the expansion of bioenergy plants, then we would expect that there would be an increase in the percentage of material from land clearing that would be chipped and used for biomass fuel. Although it is not possible to quantify historical recovery rates, we can demonstrate the potential magnitude of this biomass source by considering the impact of different recovery rates. A recovery rate of 30% would imply that 90,000 green tons of material was collected and utilized. Each increase of 10% in the recovery rate would add an additional 30,000 green tons to the supply base, so at 70%, the total volume of supply available would be 210,000 green tons.

While the disposition of wood from land clearing sources is not known in 2000–2005[48], it is highly probable that if demand increases significantly for bioenergy uses, a greater share of this wood would be recovered and shipped to these markets. Logistics and economics will govern how much biomass can be recovered from land clearing. The kinds of machinery used, the harvesting methods, and the end-use markets for this wood will vary depending on the size of the parcel being cleared and other site-specific factors. The price of biomass delivered to a bioenergy plant will also be a critical factor in determining how much biomass is actually recovered, as will transport costs and tipping fees when the option is sending the material to a landfill.

The potential volume of wood that could be generated from land clearing in 2010–2025 will depend critically on the current disposition of this wood. If current recovery and utilization are low, the incremental volumes available in the future could be substantial. At the extreme, one might consider the increase in volume to be as much as 120,000 green tons if recovery rates were to increase from 30% to 70%. Conversely, if current recovery rates are higher due to tipping fees and competing uses, "new" biomass from these sources in the future would be reduced accordingly. A final consideration is the possibility that this material in being "underutilized" in current markets. That is, if wood is chipped and used in landscaping primarily because it is a good economic option compared to disposal, it is possible that some of this wood could be diverted to bioenergy in situations where that might become a higher value use.

3.5.2 TREE CARE AND LANDSCAPING SOURCES

Among the tertiary sources mentioned above, the most significant is wood from tree care and landscaping sources. This wood is often referred to as "urban wood" which is somewhat of a misnomer because it includes wood not only from tree care in urban areas, but also wood from tree care from sources such as county parks and recreation areas and maintenance of electric power lines. The term can also be confusing because it is not always clear whether it includes "urban waste" such as construction debris.

A literature review conducted in 2002 indicated that tree care/landscaping sources accounted for 1.0 million tons (42%) out the total available supply of 2.5 million tons of non-forest wood biomass in Massachusetts (Fallon and Breger, 2002). However, given the difficulties in estimating this volume (noted in the report), this estimate is perhaps best used to suggest that the potential from these sources may be substantial and worthy of further investigation (importantly, the carbon profile of this material is generally similar to logging residues and thus very favorable compared to that of harvesting standing trees). Problems in measuring supplies from these sources may be attributed to: 1) the actual generation of this material is difficult to estimate; 2) it appears that wood from land clearing may be included in this estimate; 3) little is known about the current disposition of these materials, although some broad generalizations are possible such as more than half of the material in the Northeast is "managed on-site"; and 4) the economics of recovering this material are quite variable due to the wide variety of sources from which it is generated.

3.6 BIOMASS SUPPLY FROM NEARBY STATES

The outlook for how much wood is available to furnish an expansion of bioenergy capacity in Massachusetts is certainly not complete without considering potential wood supply and demand from the surrounding region. State boundaries mean little in the wood biomass market, as demand, supply, and prices are determined on a regional basis. New bioenergy facilities in Massachusetts would have access to wood from nearby states, while, at the same time, new bioenergy facilities in nearby states would have access to wood supplies in Massachusetts.

There are a number of ways to gain some insights into this issue. Our strategy is as follows. Given the objectives of this study, we have focused most of our effort on a detailed analysis of forest biomass fuel supplies within Massachusetts. It is not possible to use the same approach for the Massachusetts timbershed, so we assess the potential of this region by putting it in perspective relative to Massachusetts. Among the key features that we compare are: timberland areas, timberland inventory, timber growth rates, landowner characteristics, and forest products output. We have defined the timbershed as the counties which border Massachusetts: the distance across these counties is similar to the maximum that biomass could be economically transported to bioenergy plants located in Massachusetts.

Once estimates of "new" biomass supply potential are developed for the border counties, the question remains as to where this wood will be consumed. This will depend on many factors including local demand, permitting requirements for new energy facilities, who builds first, transportation costs and infrastructure. In the last section, we discuss the implications of these factors for future wood flows to—and from—Massachusetts.

48 The startup of the Schiller plant in Portsmouth, New Hampshire in 2006 makes the comparisons going forward more problematic. The plant consumes about 500,000 green tons of wood per year and has ready access to wood from land clearing in eastern Massachusetts (where most land clearing in the state occurs).

This section thus addresses two central questions:

- How much incremental biomass supply is available in the border counties?
- How much of this supply is likely to be shipped to new bioenergy plants in Massachusetts?

3.6.1 TIMBERLAND AREA AND TIMBER INVENTORY

Timber inventory is an obvious place to start in considering the border counties' potential contribution in meeting future demand from Massachusetts bioenergy plants. In Exhibit 3-18, we show the timberland areas and timber growing stock inventories in Massachusetts and in the major counties that border Massachusetts.[49] These FIA data indicate that timberland areas in the border counties are nearly 30% greater than those of Massachusetts. The conclusion is the same using the growing stock data.

Also noteworthy is that Massachusetts has a much higher share of public land (30%) than the border counties (an average of 19%, ranging from 28% in the Vermont and Connecticut sub-regions to only 5% in New York's three counties). Thus, when private lands only are considered, timberland areas and timber volumes in the border counties are about 50% greater than those in Massachusetts. This distinction is important because harvesting regulations for biomass fuel are generally more restrictive on public lands than on private; for example, in New Hampshire, whole-tree harvesting is prohibited on National Forest lands.

Exhibit 3-18: Timberland Area and Growing Stock Inventory in Massachusetts Timbershed, 000 Acres and Million Green Tons; 2008

	Area			Inventory		
	Total	Private	Public	Total	Private	Public
Massachusetts	2,895	2,026	869	207	146	62
Border County Total	3,712	3,018	694	262	212	50
New Hampshire (3 counties)	1,075	938	137	81	70	11
Vermont (2 counties)	755	543	212	57	43	15
New York (3 counties)	747	708	38	46	43	3
Connecticut (4 counties)	983	709	274	69	49	19
Rhode Island (1 county)	152	120	33	10	8	2
Combined Total	6,607	5,044	1,563	470	358	112
Border Counties ÷ Mass.	1.28	1.49	0.80	1.27	1.46	0.81

Source: FIA On-line; volumes converted from original units assuming 30 green tons per 1000 cubic feet. Note that 2008 is the nominal date for the survey data, but the data were compiled from annualized surveys and thus reflect an average of data collected over the period 2004–2008. County List: New Hampshire: Cheshire, Hillsborough, Rockingham; Vermont: Bennington, Windham; New York: Rensselaer, Columbia, Dutchess; Connecticut: Litchfield, Hartford, Tolland, Windham; Rhode Island: Providence

3.6.2 TIMBER GROWTH

When interpreted strictly from a biophysical standpoint, there is a large volume of "excess" wood available in both Massachusetts and the border region in the sense that forests are growing more wood than is being removed through harvesting and mortality. Here we compare the potential of the border counties to Massachusetts on the basis of relative rates of timber growth. We should emphasize that relationship between net growth and removals is not a measure of supply; it only speaks to how much timber could be harvested without reducing inventory levels.[50]

There are a number of ways of measuring and evaluating timber growth. Ultimately, the key variable of interest is how much additional wood will become available in different regions. As noted above, we are primarily interested in private inventories because biomass harvesting is subject to fewer restrictions and owners tend to be more responsive to market forces.

Most often, this growth has been evaluated by comparing net growth (gross growth less mortality) and removals. This relationship would be an excellent metric (it essentially defines inventory accumulation at any point in time) were it not for the poor quality of the data on removals. Furthermore, issues of data accuracy have become more of a concern in recent years due to the new annualized survey procedures that have been adopted by the Forest Service. For example, the sampling error for removals in 2008 is 45% in Massachusetts and 31% in New Hampshire. At the county level, the sampling error for removals is so large as to make these data effectively meaningless.[51]

Although any approach will encounter problems with accuracy due to sample size and sample frequency issues, we believe that comparing inventory levels over time is a better method for

[49] Data on growing stock volumes significantly understate the volume of biomass available because of the availability of wood from non-growing stock sources, notably cull trees, tops and limbs. However, our analysis is focused on relative levels—not absolute volumes—and this omission has little effect on our conclusions.

[50] Even if a forest is not adding new wood each year, it still has the potential to contribute to biomass production; biomass supplies can come out of existing stocks, not growth. From a carbon standpoint, a forest that has matured to the point that the yield curve has leveled off (net growth = mortality) may be a preferred source of material.

[51] Data for 2008 for timber removals in 12 Massachusetts counties show: no removals recorded in 7 counties, sampling errors of 100% or greater for 3 counties. For the 13 selected counties that are adjacent to Massachusetts, there were no removals recorded in 2 counties, sampling errors of 100% or greater for 4 counties, and the minimum sampling error for the remaining 7 counties was 53%. The reason for the poor accuracy is that removals are a rare event given the sampling methodology; for example, in Massachusetts, about 120 plots were re-measured in 2008 (20% of the 600 plots in the sample) and with about one percent of timberlands harvested in Massachusetts each year, that means that one would expect to find, on average, only about six plots with harvest activity every five years.

evaluating growth trends. The primary reason is statistical in that standing inventory can be measured on each plot that is surveyed each year. Likewise, with regard to components of change in the FIA data, net growth is much more reliable than data on removals. Since we are interested in small areas, we have also combined private and public inventories for this comparison because sampling errors for areas and inventories increase significantly for separate ownerships.

3.6.2.1 Growth per Acre

When all lands (private and public) are considered together, timber growth rates in Massachusetts are similar to the border region on per-acre basis. In Exhibit 3-19, average stocking levels are shown along with two sets of growth rates. The data on net growth per acre (gross growth less mortality) are derived by dividing net growth (as reported directly by FIA data) by the area in each region. The data indicate that growing stock timber inventories in Massachusetts are increasing at an average rate of 1.6 green tons per acre. The average growth rate in the border counties is essentially the same (1.5 green tons per acre), spanning a range of 1.2–1.8 green tons per acre.

The second set of growth data is derived by calculating the annual rate of change in per-acre stocking levels using FIA data between the 2004–2008 inventory/area surveys and the surveys from 10-to-15 years ago. This is a more inclusive measure of timber accumulation on an average acre by accounting for not only net growth and mortality, but also removals. These data also show very little difference between Massachusetts and the border counties—timber inventory volume is increasing at an average of about 0.8–0.9 green tons per acre, and with the exception of Rhode Island, the border counties are clustered around this number.

According to the above data, timber volume per acre is increasing at very similar rates throughout the area we have defined as the Massachusetts timbershed. These similarities reinforce the idea of using relative land areas as a measure of potential supply. Thus, if timberland use and ownership were to remain the same over the next 15 years, the potential contribution of the border counties areas—from a growth perspective—would be about 50% greater than Massachusetts (based on the private timberland area).

Exhibit 3-19 Stocking Levels and Inventory Growth for Growing Stock

All Timberlands (Private + Public), Green Tons per Acre			
	Stocking	Net G	Inv Δ
Massachusetts	71.7	1.6	0.8
Border County Total	70.7	1.5	0.9
New Hampshire (3 counties)	74.9	1.3	0.7
Vermont (2 counties)	76.1	1.2	0.7
New York (3 counties)	61.1	1.8	1.0
Connecticut (4 counties)	70.0	1.8	1.0
Rhode Island (1 county)	65.9	1.2	2.4

Notes: See Exhibit 3-18 for county definitions. Net G is net growth per acre: the net growth volumes are taken directly from FIA data for 2008 and divided by area for 2004–2008 (Exhibit 3-18). Inv Δ is a more inclusive measure of volume change on an average acre and accounts for net growth, removals and mortality; it is calculated as the change in stocking levels over the last 10-to-15 years (depending on the date of the previous inventory).

3.6.2.2 Total Volume Growth

Does the conclusion change when we adjust overall inventory growth for historical land use changes? There are two aspects of land-use change to consider: 1) shifts in total timberland area over time; 2) shifts from private to public ownership. For the border counties as a whole, the change in total timberland area has been negligible (a decrease of less than 1% from the earlier inventory years). However, over this same time frame, there has been a large shift from public to private ownership: approximately 20,000-to-25,000 acres per year have shifted into public ownership according to FIA data (as noted earlier, there are inconsistencies in these data due to measurement errors and sampling errors and their accuracy has been disputed). Thus, while the total increase in timber inventory was about 2.6 million green tons per year in the border zone, the increase in *private* timber inventories was only 0.9 million green tons per year, while inventories on *public* lands increased by 1.7 million green tons per year.

When measured on a comparable basis, private timber inventory volume in Massachusetts has increased at a rate of about 1.1 million green tons per year. Thus, in the important area of private timber inventory growth, the data suggest that inventories in Massachusetts are increasing at rates similar to those in the surrounding counties. From this perspective, the border countries lose the 50% advantage that we observed when considering growth rates on a per-acre basis.

Of course, there is no *a priori* reason to assume that land use changes will continue at the same rates as in the recent past. Good arguments can be made that future shifts from private to public lands could accelerate or proceed more slowly. In any case, it does seem clear that a serious assessment of biomass fuel availability in the border counties should consider an in-depth analysis of land-use changes in the region. To the extent that significant reductions in private timberland will continue, this would likely have an important influence on potential supplies from the surrounding region.

3.6.3 THE FOREST PRODUCTS INDUSTRY AND REGIONAL HARVESTING

Another possibility for assessing the relative importance of the border counties is to consider harvesting levels given that the greatest potential for biomass (at least in the near term) comes from integrated harvesting with higher-value industrial roundwood. Logging residues—generally considered to be a prime source of biomass fuel—will be directly proportional to the amount of industrial roundwood harvested. Perhaps more importantly, areas that already have a significant forest industry may be good candidates for biomass fuel harvests through additional cutting of low-value timber, or possibly because forest industry intensity is a good indicator of timber availability and underlying landowner attitudes.

For this overview, we have used TPO data because they have the appropriate concepts at the county level (Exhibit 3-20). These data indicate that production in the border counties is about three times that in Massachusetts; thus, from the vantage point of current harvesting activity, the border counties show a lot more promise as a source of biomass than Massachusetts. The table also shows an index which compares the intensity of harvests in the different areas—this is calculated as roundwood harvests divided by total timberland acres, and is indexed to Massachusetts = 1.0.

Exhibit 3-20: Industrial Roundwood Harvests in Massachusetts Timbershed, 000 Green Tons and Index; 2006

	Sawlogs	Pulpwood	All Ind.	Cut/Acre
Massachusetts	217	33	254	1.0
Border County Total	605	174	819	2.5
New Hampshire (3 counties)	252	111	387	4.1
Vermont (2 counties)	142	28	170	2.6
New York (3 counties)	92	30	137	2.1
Connecticut (4 counties)	101	6	107	1.2
Rhode Island (1 county)	17	0	17	1.3

Source: Harvest data from TPO. All Ind. is "All Industrial" and, in addition to sawlogs and pulpwood, includes veneer logs, composite products, posts, poles, piling, and miscellaneous. Cut/Acre is an index (Massachusetts = 1.0), measured as All Ind./Timberland Acres. See Exhibit 3-18 for county definitions.

3.6.4 LANDOWNER CHARACTERISTICS IN THE REGION

Ownership characteristics provide another perspective on future wood biomass fuel availability in the border counties for at least three reasons: 1) the size of forest holdings is generally considered to be highly correlated with the landowner's propensity to harvest timber; 2) the size of forest holdings is of particular importance for biomass fuel because of economies of scale in whole-tree harvesting; and 3) landowner attitudes are important in the decision of whether or not to use their land for commercial timber production.

In Exhibit 3-21, data that address the above issues are presented at the state level.[52] In Massachusetts, the average parcel size for family-owned forest land is 6 acres, while Rhode Island is also 6 acres and Connecticut averages 9 acres per owner. Forest holdings are much larger in New Hampshire and Vermont, where the average owner has 19 acres and 36 acres, respectively (although it is likely to be the case that parcel sizes in the border counties are more similar to those in Massachusetts than the state averages would imply). Notably, a significant area of New Hampshire's private forest land (1.3 million acres) is held by non-family owners (average forest holdings of owners in this group are substantially larger). According to these survey data, only 43% of the family forest land area in Massachusetts is held in parcels that are 50 acres or larger. New Hampshire and Vermont are much higher at 64% and 75%, while Connecticut is 48% and Rhode Island is 33%. Importantly, New Hampshire has twice as much family-owned land as Massachusetts in 50+ acre parcels, while Vermont has three times as much land; however, we do not have data on the relative areas for the border county region.

Exhibit 3-21: Attributes of Family Forest Landowners

	MA	NH	VT	CT	RI
Private Lands (000 acres)	2,179	3,646	3,864	1,383	303
Family Forest Owners (000 acres)	1,686	2,358	3,109	898	204
Family Forests, 50 acres or more	729	1,514	2,343	434	68
% of Family Forests, 50 acres or more	43%	64%	75%	48%	33%
Average Size, Family (acres per parcel)	5.8	19.0	35.7	8.9	5.5
Timber production is important*	20%	21%	29%	12%	11%
Commercial harvest in past 5 years	40%	59%	68%	39%	26%
Commercial harvest in next 5 years	20%	29%	39%	9%	11%
% of family forests available given constraints*	32%	43%	57%	20%	21%

Source: National Woodland Ownership Survey, Butler et al., 2008; on-line data.

Notes: 1) Data are state level, not for county sub-regions.

2) The survey asks landowners to rank the importance of producing commercial timber on a 7-point scale from "very important" to "not important." These data show the percentage that ranked production as '1' or '2' on this scale.

3) "% of family forest available given constraints" is taken from Butler et al. (2010) and reflects reductions for biophysical and social constraints, including parcel size and landowner attitudes and preferences.

With respect to timber production, probably the three most important questions asked in the National Woodland Ownership Survey are: 1) how important is timber production?; 2) did you conduct a commercial harvest in the past five years?; and, 3) do you plan to conduct a commercial harvest in the next five years? The results shown in Exhibit 3-21 are much as one might expect: Vermont and New Hampshire owners gave answers that

[52] We evaluated these data at the survey unit level in New Hampshire and Vermont to focus more directly on the sub-regions of concern. However, there were no obvious differences within the states, particularly given the large sampling errors associated with this survey. We did not consider the data for New York because the three-county area accounts for such a small share of the state's total forest land.

most favored timber production, Massachusetts was ranked in the middle of this group, and Connecticut and Rhode Island owners were least oriented toward timber production.

There appears to be a fairly high degree of correlation between parcel size and landowner interest and willingness to pursue commercial timber harvests. A recent study by Butler et al. (2010) developed a methodology to combine these factors in a manner to eliminate double counting in the presence of multiple constraints. Harvest "participation rates" from this study are shown on the last line of Exhibit 3-21: Vermont had 57% of family forest land available for harvest (ranking the highest of all 20 northern states); New Hampshire was second of this group with 43% available; Massachusetts had only 32% of land available; Connecticut and Rhode Island were the lowest with only about 20% of land available (and ranked among the lowest of the 20 northern states).

Some question the validity and usefulness of landowner surveys, so it is useful to have additional information from other sources. Participation rates in current use programs provide further insights into the level of interest in forest management and related income incentives. The Chapter 61-61A-61B program in Massachusetts has had limited success relative to its counterparts in New Hampshire and Vermont. In Massachusetts, about 15% of private forest lands were enrolled in this program in 2009 (Massachusetts Department of Conservation, 2009). This is in stark contrast to New Hampshire where about 27,000 landowners participate in the current use program, covering nearly 3 million acres (New Hampshire Timberland Owners Association, 2010). In Vermont, more than 1.6 million acres of forest land were enrolled in their current use program in 2009 (Vermont Department of Taxes, 2010).

Ownership attributes clearly reinforce the patterns shown earlier on the basis of area, inventory and harvesting. The potential for forest biomass fuel from border counties in Connecticut and Rhode Island appears limited. On the other hand, the border counties of New Hampshire, Vermont, and New York are similar in size to Massachusetts (on the basis of timberland area, inventory, and growth) and their forest products industry and industrial roundwood harvests are significantly higher. Furthermore, landowner surveys for New Hampshire and Vermont show family owners in these states to be more supportive of timber harvesting.

3.6.5 SUMMARY OF FOREST BIOMASS SUPPLY POTENTIAL IN BORDER COUNTIES

In order to assess potential forest biomass supplies from the counties surrounding Massachusetts, we have looked at several key measures relative to Massachusetts. The general conclusion from our analysis of timberland area, timber inventory, and timber growth is that private lands in the border counties have the ability to supply about 50% more biomass than Massachusetts.

When the analysis is expanded to account for landowner characteristics and the development of the forest products industry, the potential biomass contribution of border counties becomes more difficult to evaluate. It is certainly the case that New Hampshire, Vermont, and New York would be much more conducive to increased harvesting than Massachusetts based on landowner attitudes and the distribution of ownership by parcel size. This already manifests itself in a much larger forest industry and much higher roundwood production. Thus we are faced we this analytical dilemma: these regions may be more attractive for timber harvesting, but given that more harvesting is now taking place, how much further expansion is likely? Has investment to date put the production in these regions in equilibrium relative to Massachusetts? Are there still more promising opportunities in the border counties? Or are they already approaching production levels that make it more difficult to expand further? Whole-tree harvesting already has a long history in southern New Hampshire for example, suggesting that future increases might be more difficult to achieve and come only at higher cost.

While this issue will not be settled in this analysis, we have made an effort to better understand the situation in southern New Hampshire: it has been suggested that New Hampshire has the most potential for increasing supplies of forest biomass because of its inventory, harvest rates, and favorable stance toward timber production. Our evaluation of recent harvest relationships and price trends is provided in Appendix 3-D. We did not find any obvious pockets of opportunity or expansion possibilities in the southern counties, nor any evidence to support claims that southern New Hampshire may be in an advantageous position to produce more biomass compared to neighboring areas.

Since we have considered the availability of biomass from border counties in relation to supplies from Massachusetts, it is important that we consider these supplies in the context of our two scenarios for Massachusetts. In our Low-Price Biomass scenario, we expect that biomass supplies in Massachusetts will increase as a result of more intensive harvesting using whole-tree harvesting. Given the development that has already taken place in some of the border areas, we would not expect that increased biomass demand at current biomass prices would spur additional harvesting to the same extent that we might see in Massachusetts. However, in our High-Price Biomass scenario, more land is harvested and more timber is harvested from that land. We would expect that this will cause a substantial response in the border counties, just as we expect in Massachusetts. Given landowner characteristics in the region, one might argue that the response in border counties might be greater than in Massachusetts.

Mindful of the numerous uncertainties involved in projecting the potential supply of biomass in the counties bordering Massachusetts, we consider a reasonable "guesstimate" to be 50% more than can be produced within this state. In our Low-Price Biomass scenario, this would suggest the border counties could produce an additional 225,000–375,000 green tons of forest biomass annually. If the High-Price Biomass scenario unfolds, border county supply would jump to an annual average of 1.0–1.3 million green tons.

3.6.6 INTER-REGIONAL TRADE AND IMPLICATIONS FOR BIOMASS SUPPLIES FOR FUTURE BIOENERGY PLANTS IN MASSACHUSETTS

Understanding potential wood biomass supplies in the counties that surround Massachusetts is critically important in estimating biomass availability for bioenergy plants that may get built in Massachusetts. But where will this wood be consumed? It is crucial to consider future demand outside of Massachusetts and possibilities for biomass trade. Biomass produced in the border counties could stay within its home zone for local use, it could flow between sub-regions (from New Hampshire to Vermont, for example), it could flow to the northern areas, or it could flow to Massachusetts. Likewise, wood in Massachusetts is not limited to home use; in fact, with few outlets for wood biomass in Massachusetts currently, biomass chips are now being shipped to bioenergy facilities in New Hampshire.

3.6.6.1 Historical Wood Products Trade

Recent patterns in wood products trade in this region provide some perspective on trade possibilities. Data available on wood trade for New Hampshire, Vermont, Maine, and New York show that the four-state region is a net importer of wood, purchasing 195,000 green tons in 2005. (We caution that the data are for only one year and they do not indicate specifically what is happening with Massachusetts.)

Data for Vermont (Northeast State Foresters Association, 2007b) indicate that Vermont consumed about 400,000 green tons of biomass chips in 2005. Of this total, about 300,000 green tons were imported from other states, while at the same time, Vermont exported 75,000 green tons; thus, net imports were just over half of wood chip consumption.

Based on the limited data that we have on Massachusetts wood trade, it appears that trade between Massachusetts and Vermont has been one-directional, with Massachusetts exporting a small volume of sawlogs to mills in Vermont.

Exhibit 3-22: Wood Trade Among Northeast States, 2005 (000 green tons; does not include international trade)

	Import	Export	Net Imports
New Hampshire	353	820	-468
Vermont	508	630	-123
Maine	1,115	363	753
New York	838	805	33
TOTAL	2,813	2,618	195

Source: Northeast State Foresters Association, 2007a. Original data in cords; converted to green tons assuming 2.5 green tons per cord.

3.6.6.2 Potential Future Trade in Forest Biomass Fuel

One of the advantages of Massachusetts size and shape is that it has access to a large horseshoe of wood as part of its timbershed. However, it is important to recognize that an even larger horseshoe envelops this timbershed, which means that wood available from that area may provide incentives to build bioenergy facilities in the border region, or that wood could flow from Massachusetts to feed plants in that area. Exhibit 3-23 provides a list of facilities that—if built—might potentially compete for the same wood that could provide feedstock to proposed plants in Massachusetts. Plans and proposals change frequently and this list is intended only to be suggestive of some of the facilities—and their size—that are now under consideration in this region. This list does not include facilities that are located overseas, but there is always the possibility that biomass produced in this region could be directed to export markets.

Exhibit 3-23: Proposed Bioenergy Plants that Could Influence Biomass Availability for Massachusetts (Wood Use in Green Tons per Year)

State	Company	Location	Size	Wood Use
MA	Russell Biomass	Russell	50 MW	550,000
	Greenfield Biomass	Greenfield	50 MW	550,000
	Tamarack Energy	Pittsfield	30 MW	350,000
	Palmer Renewable	Springfield	30 MW	*235,000
NH	Clean Power Development	Berlin	29 MW, CHP	340,000
	Clean Power Development	Winchester	15 MW	150,000
	Alexandria Power	Alexandria	16 MW (re-start)	200,000
	Greenova Wood Pellets	Berlin	pellets	400,000
	Laidlaw Energy	Berlin	40 MW	400,000
VT	Vermont Biomass Energy	Island Pond	pellets	200,000
	Brattleboro District Heat	Brattleboro		
CT	Decker International	Plainfield	30 MW	400,000
	Tamarack Energy	Watertown	30 MW	400,000

*Notes: *plan calls for construction and demolition debris as feedstock.*

Two important strategic issues in siting large-scale bioenergy facilities are relevant to this discussion. One is that transportation costs are a significant component of delivered biomass costs and so the location of new facilities should be optimized so that they have access to the most wood within short distances. Thus, plants should be built where there are ample supplies of wood in the "home" area. This could be analyzed with mathematical optimization models, but the results would probably be of little use due to the large number of other factors that affect plant location, many of which are specific to individual locations and facilities.

A second strategic issue is what has been termed "first-mover advantage," which suggests that the facility that starts up first will have a competitive advantage in establishing its network and logistics for wood supply. In addition, the first mover may discourage future investments that would need to access the same timbershed. However, being first does not rule out the possibility that other new facilities that may start later: they may be willing to compete for the same wood due to proximity or the belief that they will be more efficient and thus able to pay more for their fiber.

3.6.6.3 Wood Supplies Available for Massachusetts

How much in the border counties would be available for new bioenergy facilities in Massachusetts? This will depend on how the bioenergy industry in the region evolves and depends on the following:

- How many new facilities will be built and how large will they be?
- Where will they be built?
- When will they be built?

In order to provide some general guidelines, such an analysis might proceed as follows. For economic reasons, it would seem most likely that the majority of wood produced would remain in its home market: it might be reasonable to assign that a 50% probability. The remaining 50% could be shipped to Massachusetts or shipped "outside" to the facilities in the next ring of border counties. Thus, in this example, the supply of biomass being shipped to Massachusetts from the border region would be 25% of the total available. If the amount of wood available in Massachusetts is X, and the amount available from outside is 1.5X, then Massachusetts could plan on increasing its supplies by 0.375X (or 0.25 * 1.5X).

These numbers can be adjusted to develop some insights into what might represent a reasonable upper bound. Suppose we make the assumption that the amount of "new" biomass available in the border counties is actually twice that available in Massachusetts (call this 2X). Furthermore, suppose that Massachusetts is able to purchase half of that wood by virtue of location or the timing of establishing new plants and their supply infrastructure. In this case, Massachusetts could increase its supply by X (or 0.5 * 2X), thus doubling the amount available only within the state.

In order to provide some general guidance and indication of the volumes of biomass that could be available from the border counties to supply new bioenergy facilities in Massachusetts, we have assumed that Massachusetts could successfully purchase 50% of the potential incremental production. In our Low-Price Biomass scenario, this would suggest that 110,000–190,000 green tons of forest biomass from border counties could augment the supplies available within Massachusetts. Supplies available from border counties increase to 515,000–665,000 green tons in the High-Price Biomass scenario.

Suffice to say, there is no simple answer to the question of how much biomass might be available from the border counties to furnish new bioenergy facilities in Massachusetts. However, it would seem prudent that each new facility (particularly those with large annual wood consumption) conduct its own feasibility study and carefully establish that the supplies it needs are available and not destined for other bioenergy plants.

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CHAPTER 4
FOREST SUSTAINABILITY AND BIOMASS HARVESTING IN MASSACHUSETTS

4.1 INTRODUCTION

The objective of this task of the Biomass Sustainability and Carbon Policy study is to evaluate the potential impacts posed by increased biomass harvesting in the forests of Massachusetts and offer recommendations for mitigating any negative outcomes that are identified. Although biomass harvesting offers opportunities to enhance silvicultural treatments and produce greater quantities and quality of traditional forest products such as sawlogs these economic impacts are not the focus of this chapter. This chapter reviews indicators of forest sustainability for Massachusetts forests and gauges the impact of increased biomass harvesting on forest ecosystem sustainability. It also suggests options for policies, guidelines, or regulations that might be needed to protect ecological values while producing a forest based energy supply and realizing the economic benefits from increased silvicultural productivity.

The concept of forest sustainability requires consideration of what is being sustained, over what time period, and at what landscape scale. Section 2 addresses these issues at the stand-level, focusing on the localized ecological impacts of biomass harvesting. These stand-level considerations are most readily observed and quantifiable. The stand-level analysis discusses the potential impacts to ecological systems and processes and then reviews the biomass harvesting guidelines used by other states and political entities to minimize any impacts at the stand level. Then the adequacy of Massachusetts' current forest management regulations and guidelines are evaluated. Section 3 considers a broader set of sustainability factors at the landscape rather than the stand level. This discussion includes socio-economic indicators that go beyond stand-level ecological effects and have the potential to alter the provision of forest ecosystem services at a regional scale. The chapter concludes with a discussion of policy options that the state may want to consider for addressing these potential stand- and landscape-scale impacts.

To help answer questions about the potential impact of increased biomass harvests on forest sustainability at both stand and landscape scales, this report draws heavily on information from three separate but related reports that were developed or updated for this study by the Forest Guild. These documents are included as appendices to this report. *Ecology of Dead Wood in the Northeast* consists of a literature review of important topics relevant to biomass harvesting in forest types common to Massachusetts. Excerpts from this report and implications for Massachusetts policies are included in Section 2. *An Assessment of Biomass Harvesting Guidelines* (2009) was revised for this study, and the unpublished revised version is included. Finally, *Forest Biomass Retention and Harvesting Guidelines for the Northeast* is a complete set of recommendations to protect Massachusetts forest types that was developed in a parallel process by Forest Guild members and staff.[1] These guidelines provide a useful starting point for the development of state-specific guidelines for Massachusetts.

These reports provide more detailed background information and a richer exploration of the underlying science and issues. Overviews of each of these reports and their implications for policies addressing increased biomass harvests in Massachusetts are included in Section 2 with the stand-level discussion.

4.2 STAND-LEVEL IMPACTS TO FOREST HEALTH RESULTING FROM INCREASED BIOMASS DEMAND

As we learned from the analysis in Chapter 3, woody biomass generated solely from logging debris (tops and branches) will contribute minimally to commercial-scale biomass facilities. This implies that the only way to meet higher demand would be to increase the annual forest harvest, i.e., cut more trees per acre or harvest additional acreage. Increasing harvest levels does not automatically mean an unsustainable forest ecosystem. As noted in Chapter 3, timber inventories have been increasing in Massachusetts for many decades and harvests can potentially be increased without reducing future wood supplies. The challenge with increased harvests is to provide assurances that forest ecosystem health would be preserved. There are three main areas where forest ecosystem sustainability might be affected. These issues are relevant to any harvesting operation, but become of greater concern if additional wood is removed for biomass:

- Impact on hydrology and water quality
- Impact on soils and site productivity
- Impact on habitat and biodiversity

4.2.1. INTRODUCTION

Hydrology and water quality are already covered with existing Best Management Practices (BMPs) in Massachusetts (reference to BMPs). Increasing the harvest levels to meet biomass demands should therefore not compromise water resources because of the protections already in place. It is not clear that protections are in place for soils and productivity, or biodiversity, and therefore we focus on these issues in this Task.

Many of the possible impacts related to biomass harvesting relate to the removal and retention of woody material. This is true for soil protection as well as wildlife and biodiversity. Although dead wood and declining trees have traditionally had little commercial value, they do have substantial ecological value. For this reason, we focus our analysis on the ecology and benchmarks for retention of this material.

[1] The three Forest Guild reports mentioned here are included in the Appendices.

Ecology of Dead Wood in the Northeast was prepared to provide background information for this study as well as to policymakers and foresters involved in biomass harvest issues elsewhere.

The paper reviews the scientific literature to provide information about the amount of dead wood retention necessary for forest health in the forest types of the northeastern U.S. Establishing the ecological requirements for dead wood and other previously low-value material is important because expanded biomass markets may cause more of this material to be removed, potentially reducing the forest's ability to support wildlife, provide clean water, and regenerate a diverse suite of vegetation. The paper covers the topics of dead wood, soil compaction, nutrient conservation, and wildlife habitat in temperate forests generally as well as in specific forest types of the Northeast. The sections that follow include excerpts from the report that cover the relevant major research findings and then summarize the implications for policies in Massachusetts.

4.2.2. IMPACTS ON SOILS AND PRODUCTIVITY

Biomass harvesting can affect chemical, physical, and biological attributes of soils. The silvicultural choices of what to harvest, the amount of material harvested, and the way the material is harvested are all factors that need to be considered, and sometimes mitigated, to protect soils. This section covers issues related to soil nutrients and productivity.

4.2.2.1 Definition of Downed Woody Material

Woody material is sometimes divided into coarse woody material (CWM), fine woody material (FWM), and large woody material. The U.S. Forest Service defines CWM as down dead wood with a small-end diameter of at least 3 inches and a length of at least 3 feet and FWM as having a diameter of less than 3 inches (Woodall and Monleon 2008). FWM tends to have a higher concentration of nutrients than CWM. Large downed woody material, such as logs greater than 12 inches in diameter, are particularly important for wildlife. Fine woody material is critical to nutrient cycles. In this report, we use the term **downed woody material (DWM)** to encompass all three of these size classes, but in some circumstances we discuss a specific size of material where the piece size is particularly important.

Implications for Massachusetts Policies: In order to avoid confusion, it will be important for Massachusetts to settle on definitions and terminology that are most helpful to discussions of native forest types and associated concerns.

4.2.2.2 DWM: Stand Development and Harvesting

The process of dead wood accumulation in a forest stand consists of the shift from live tree to snag to DWM, unless a disturbance has felled live trees, shifting them directly to DWM. During stand development following a clear cut, there is a large amount of DWM. The DWM remaining from the initial harvest decomposes rapidly in the first 25 years and continues to decline to age 40. The young stand produces large numbers of trees, and the intense competition produces an increasing number of snags. As the trees grow larger, more snags of larger sizes begin to appear. From age 40 to 100 years, DWM increases as small snags fall. Then larger snags begin to contribute to DWM. Very few large pieces of DWM are produced. Large DWM often results from wind or other disturbances that topple large trees in the old-growth stage. Thus, large dead wood tends to accumulate periodically from these disturbance pulses, whereas small pieces of DWM accumulate in a more predictable pattern throughout all stages of stand development.

Implications for Massachusetts Policies: The patterns of DWM development indicate the importance of retaining large live trees and large snags at the time of harvest. As the stand moves through the younger stages of development, it creates minor amounts of DWM of larger sizes. Retaining larger-diameter trees in all stages can provide larger size classes of DWM.

The concern at the stand level is that increased biomass harvests in Massachusetts might alter natural patterns of DWM accumulation and cause ecological damage. This can occur in stands that have not previously been harvested or by adding the additional removal of biomass to any kind of previous harvest. With new biomass markets becoming available, all sizes of woody material might be removed. Harvests that include taking material for biomass energy could lead to the removal of most or all of the dead or dying standing material, as well as low-quality trees that would eventually enter this class. Regeneration harvests, cuttings that are intended to establish new seedlings, might be helped by the ability to remove cull material that hinders new regeneration, but if the biomass removals are too heavy and too consistent, the amount of DWM could be reduced to insufficient levels. In some cases, increased prices for biomass, coupled with under-utilized equipment and logging contractors looking for work, might persuade a landowner to do a more intensive harvest than under a pre-biomass market scenario. Without guidelines for DWM retention, these heavier harvests might, in some cases, pose a greater risk for soils by depleting the structures—FWM, and to a lesser extent CWM and large woody material—that store and release nutrients back into the mineral soil.

4.2.2.3 DWM: Soil Productivity

DWM plays an important physical role in forests and riparian systems. DWM adds to erosion protection by reducing overland flow (McIver and Starr 2001, Jia-bing et al. 2005). DWM also has substantial water-holding capacity (Fraver et al. 2002).

In many ecosystems, DWM decomposes much more slowly than foliage and fine twigs, making it a long-term source of nutrients (Harmon et al. 1986, Greenberg 2002) (Johnson and Curtis 2001, Mahendrappa et al. 2006). While there is great variation across ecosystems and individual pieces of DWM, log fragmentation generally appears to occur over 25 to 85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007).

In some ecosystems, CWM represents a large pool of nutrients and is an important contributor to soil organic material (Graham and Cromack Jr. 1982, Harvey et al. 1987). However, a review of CWM in Northern coniferous forests suggested that it may play a small role in nutrient cycling in those forests (Laiho and Prescott 2004).

A review of scientific data suggests that nutrient capital can be protected when both sensitive sites (including low-nutrient) and clearcutting with whole-tree removal are avoided (see also Hacker 2005). However, there is no scientific consensus on this point because of the range of treatments and experimental sites (Grigal 2000). A study of an aspen/mixed-hardwood forest showed that even with a clear-cut system, calcium (Ca) stocks would be replenished in 54 years (Boyle et al. 1973). Minnesota's biomass guidelines present data that showed soil nutrient capital to be replenished in less than 50 years even under a whole-tree harvesting scenario (Grigal 2004, MFRC 2007). Whole-tree clearcutting and whole-tree thinning (Nord-Larsen 2002) did not greatly reduce amounts of soil carbon or nitrogen (N) in some studies (Hendrickson 1988, Huntington and Ryan 1990, Olsson et al. 1996, Johnson and Todd 1998). Lack of significant reduction in carbon and N may be due to soil mixing by harvesting equipment (Huntington and Ryan 1990). However, intensive cutting, such as clear-cutting with whole-tree removal, can result in significant nutrient losses (Hendrickson 1988, Federer et al. 1989, Hornbeck et al. 1990, Martin et al. 2000, Watmough and Dillon 2003)—in one case, an initial 13% loss of Ca site capital (Tritton et al. 1987).

Overall, the impact of biomass harvesting on soil nutrients is site dependent. Low-nutrient sites are much more likely to be damaged by intensive biomass removal than sites with greater nutrient capital or more rapid nutrient inputs, which is one reason scientific studies on the nutrient effects of whole-tree harvesting may yield different results.

Low-impact logging techniques that reduce soil disturbance can help protect nutrient capital (Hallett and Hornbeck 2000). Harvesting during the winter after leaf fall can reduce nutrient loss from 10 to 20% (Boyle et al. 1973, Hallett and Hornbeck 2000). Alternatively, if logging occurs during spring or summer, leaving tree tops on site would aid in nutrient conservation. Nordic countries have demonstrated that leaving cut trees on the ground in the harvest area until their needles have dropped (one growing season) can also reduce nutrient loss (Nord-Larsen 2002, Richardson et al. 2002).

Implications for Massachusetts Policies: The scientific literature makes clear that DWM plays a critical role in ensuring continued soil health and productivity. Modeling indicates that biomass harvests have the potential to reduce soil nutrient capital and cause long-term productivity declines (Janowiak 2010) at some sites; but other studies identify cases where soil nutrient capital is replaced in reasonable time periods even under whole-tree harvesting scenarios.

A recent report, *Silvicultural and Ecological Considerations of Forest Biomass Harvesting In Massachusetts*, suggested that with partial removals (i.e., a combination of crown thinning and low thinning that removes all small trees for biomass and generates from 9 to 25 dry t/ac or 20 to 56 Mg/ha) stocks of Ca, the nutrient of greatest concern, could be replenished in 71 years (Kelty et al. 2008). The Massachusetts study was based on previous research with similar results from Connecticut (Tritton et al. 1987, Hornbeck et al. 1990).

During the Forest Guild's working group discussions of soil productivity, the Kelty study was investigated thoroughly as it raised serious questions of long-term sustainability. As general cautionary context for soil productivity, it should be noted that leaching, particularly of Ca due to acidic precipitation, can reduce the nutrients available to forests even without harvests (Pierce et al. 1993). In the case of Ca and the Connecticut research there are important questions as to whether the input rates from natural weathering were accurate. Other researchers believe the weathering rates are much higher and the Ca-phosphorus mineral apatite may provide more sustainable supplies of Ca to forests growing in young soils formed in granitoid parent materials (Yanai et al. 2005). For example, a recent study using long-term data from Hubbard Brook Ecosystem Study indicated that "the whole-tree harvest had little effect on the total pool of exchangeable calcium" after 15 years (Campbell et al. 2007).

Consequently, the analysis provided in the Kelty study does not provide sufficient scientific justification to generalize about Ca depletion. The bottom line is that even while some available studies suggest that soil capital should be protected by avoiding sensitive sites and prohibiting clearcutting with whole-tree removals, there is no scientific basis for concluding that avoiding clearcutting or whole-tree harvesting are necessary at all sites to maintain productivity. Sensitive soil types should be determined and appropriate guidelines applied. We recommend a conservative approach that includes the retention of some DWM in all harvests. The Forest Guild Biomass Retention and Harvesting Guidelines deal directly with these issues and are summarized in this report.

4.2.2.4 Quantities of Dead Wood

Site productivity and the rate of decomposition help determine the amount of dead wood in a given stand (Campbell and Laroque 2007, Brin et al. 2008). As mentioned above, DWM decomposition varies greatly but generally occurs over 25–85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007). All mortality agents including wind, ice, fire, drought, disease, insects, competition, and senescence create dead wood (Jia-bing et al. 2005). These mortality agents often act synergistically.

A review of 21 reports of quantitative measures of DWM in Eastern forest types shows great variability across forest types and stand-development stages (Roskoski 1980, Gore and Patterson 1986, Mattson et al. 1987, McCarthy and Bailey 1994, Duvall and Grigal 1999, Idol et al. 2001, Currie and Nadelhoffer 2002). The reports ranged from 3 to 61 t/ac (7 to 137 Mg/ha) with a median of 11 t/ac (24 Mg/ha) and a mean of 15 t/ac (33 Mg/ha). Measurements of old forests (>80 years old), had a median of 11 t/ac (24 Mg/ha) and a mean of 13 t/ac (29 Mg/ha) in DWM.

In contrast, a study of U.S. Forest Service inventory plots found a mean of 3.7 t/ac (8.3 Mg/ha) and a median of 2.9 t/ac (6.5 Mg/ha) of DWM across 229 plots in the Northeast (Chojnacky et al. 2004 see Figure 2). This low level of DWM across the landscape may be due to widespread clearcutting in the 1880-1930 periods.

Implications for Massachusetts Policies: The amount of dead wood varies across forest types and stand ages. In order to determine appropriate benchmarks that correlate with forest health, more data by stand and age is required than current research provides. However, we find there is sufficient data to construct some initial, but likely conservative, guidelines. These are detailed in the Forest Guild's Biomass Retention and Harvesting Guidelines and summarized in Section 4.5.2 of this report.

4.2.2.5 Soils and Productivity Issues by Forest Type

Northern Hardwood Forests: In general, the amount of DWM in Northern hardwood forests follows the 'U' pattern mentioned above. Young stands have large quantities of DWM (usually due to a harvest); mature stands have less; older or uncut stands have more. For example, a study in New Hampshire measured 38 t/ac (86 Mg/ha) of DWM in a young stand, 14 t/ac (32 Mg/ha) in mature stands, 20 t/ac (54 Mg/ha) in old stand, and 19 t/ac (42 Mg/ha) in an uncut stand (Gore and Patterson 1986). Gore and Patterson (1986) also note that stands under a selection system had lower quantities of DWM, i.e., 16 t/ac (35 Mg/ha). A review of other studies identified similar temporal patterns and quantities of DWM (Roskoski 1977, Gore and Patterson 1986, McCarthy and Bailey 1994, McGee et al. 1999, Bradford et al. 2009).

Estimates of the volume of down dead wood in Maine's northern hardwood forests are 598 ft^3/ac (42 m^3/ha) or 9 t/ac (20.5 Mg/ha) (Heath and Chojnacky 2001). Keeton (2006) estimates a volume of 600 ft^3/ac (42 m^3/ha) of DWM in a multi-aged northern hardwood forest.

Transitional Hardwoods: As with the other forest types discussed, DWM density tends to follow a 'U' shape in oak-hickory forests. For example, Idol and colleagues (2001) found 61 t/ac (137 Mg/ha) in a one-year post-harvest stand, 18 t/ac (40 Mg/ha) in a 31–year-old stand, and 26 t/ac (59 Mg/ha) in a 100-year-old stand. Tritton and colleagues (1987) measured 5.8 t/ac (13 Mg/ha) in an 80-year-old stand in Connecticut.

Estimates of the volume of down dead wood in Maine's oak-hickory forests are 244 ft^3/ac (17 m^3/ha) or 0.7 (1.5 Mg/ha) (Heath and Chojnacky 2001). Wilson and McComb (2005) estimated the volume of downed logs in a western Massachusetts forest at 143 ft3/ac (10 m^3/ha).

A study in Appalachian oak-hickory forests showed that the decomposing residues left after a saw log harvest increased concentration of Ca, potassium (K), and magnesium in foliage and soils after 15 years in comparison to a whole-tree harvest (Johnson and Todd 1998). However, the study found no impacts on soil carbon, vegetation biomass, species composition, vegetation N or P concentration, soil bulk density, or soil N because of the whole-tree harvest (Johnson and Todd 1998).'

White Pine and Red Pine Forests: Estimates of the volume of down dead wood in Maine's pine forests are 255 ft3/ac (18 m^3/ha) or 1.6 t/ac (3.5 Mg/ha) (Heath and Chojnacky 2001). A review of research on DWM in the red pine forests of the Great Lakes area showed that there were 50 t/ac (113 Mg/ha) of DWM in an unmanaged forest at stand initiation and 4.5 t/ac (10 Mg/ha) in a 90-year-old stand (Duvall and Grigal 1999). In comparison, the managed stand Duvall and Grigal (1999) studied had less DWM at both initiation 8.9 t/ac (20 Mg/ha) and at 90 years 2.9 t/ac (6.6 Mg/ha). The same review showed the unmanaged stand had 30 snags per ac (74 per ha) while the managed forest had 6.9 per ac (17 per ha) (Duvall and Grigal 1999). Red and white pine that fall to the ground at time of death will become substantially decayed (decay class IV of V) within 60 years (Vanderwel et al. 2006).

While not a recognized forest type, stands with a mix of oak, other hardwoods, white pine, and hemlock are common. Many of the red oak and white pine stands on sandy outwash sites are susceptible to nutrient losses because of a combination of low-nutrient capital and past nutrient depletion (Hallett and Hornbeck 2000).

Implications for Massachusetts Policies: The amount of DWM and natural patterns of decay and soil replenishment vary by forest type in unmanaged stands. Ideally, DWM retention targets would also vary by forest type; but presently there are not enough data across forest types and ages to set specific targets. The Forest Guild Retention and Harvesting Guidelines for the Northeast include examples of DWM ranges by forest types.

Exhibit 4.1: DWM Ranges by Forest Type

	Northern HW	Spruce-Fir	Oak-Hickory	White and Red Pine
Tons of DWM per acre*	8–16	5–20	6–18	2–50

**Includes existing DWM and additional material left during harvesting to meet this target measured in dry tons per acre.*

The Forest Guild's guidelines also include general targets for retaining logging residues to protect soil nutrient capital. Over time, Massachusetts and other state guidelines may be able to hone in on specific targets by forest type.

4.2.2.6 Impacts from Changing Harvesting Technology Caused by Increased Biomass Harvesting

All harvesting practices disturb forest sites, but the overall impact on soil structure and nutrients depends on the site, operator skill, and conditions of operation. A comprehensive study of site impacts in Maine (Benjamin 2010) reviewed the literature regarding soil compaction and erosion from logging. A comparison of nine related studies (Martin, 1988) concluded "the percentage of disturbance per area has increased over time with changes in equipment (tracked to wheeled machines, chain saws to harvesters) and harvest methods (partial cuts to clearcuts to whole-tree clearcuts)." However, the research also suggests that biomass harvesting will not contribute to or create additional physical impacts on the soil productivity as compared to conventional harvesting as long as BMPs are followed (Shepard 2006)

The supply scenarios developed in the Chapter 3 Forest Biomass Supply analysis indicate that "if biomass demand increases due to the expansion of electric power plants, it will almost certainly be accompanied by increases in whole-tree harvesting due to the limited supply of other forest biomass and the cost advantages of whole-tree methods." The concerns for physical soil structure and erosion revolve around the equipment that will likely be introduced on harvesting operations. Whole-tree harvesting systems come in a variety of designs that rely on different pieces of equipment. In Massachusetts, the most common whole-tree logging systems employ a feller/buncher, one or more grapple skidders, and some kind of loader at the landing. This equipment can be larger and heavier than traditional harvesting equipment and has the potential to magnify adverse effects on soil. Also, many biomass harvests use a two-pass system in which one piece of equipment cuts trees and stacks them and another piece eventually picks them up for transportation to the landing. Repeated equipment passes can cause greater degrees of soil compaction, resulting in increased soil strength, which can (1) slow root penetration and reduce the regeneration and tree growth (Greacen and Sands, 1980; Miller et al., 1996); and (2) reduce soil infiltration rates, thereby increasing the potential for erosion through changes in landscape hydrology (Harr et al.1979).

The extent of impacts on soil properties and site productivity will depend on the degree current best management practices (BMPs) and new guidelines are followed. Current BMPs include fundamental approaches that apply to biomass harvests as well as traditional harvests. They include anticipating site conditions, controlling water flow and minimizing and stabilizing exposed mineral soil. These guidelines should be re-emphasized and implemented in biomass harvests. Additional guidelines related to the retention and use of woody biomass will be helpful especially on skid trails and stream approaches. For example, research shows that spreading tops and limbs along skid trails and other operating areas and driving the equipment on this buffer can reduce soil impacts. In order to have this material available for these purposes it must be retained in place or brought back to the operating area. There are competing values of biomass that pit the desire to remove the material as a renewable fuel and to mitigate the global effects of climate change on forest ecology versus its onsite ecological benefits.

4.2.3 IMPACTS ON HABITAT AND BIODIVERSITY

Increasing harvests to include greater biomass removal will have two primary effects on habitat and biodiversity. First, a greater volume of wood will be removed from many harvest operations to meet the biomass demand. This will initially result in a more open residual stand than would have occurred otherwise and can range from stands with slightly lower residual stocking all the way to clearcuts. Habitat will change on individual parcels providing opportunities for new species and eliminating them for others. The other potential impact is on dead wood. Both standing snags and fallen logs (DWM) are important habitat features for many forest species. Dead wood is a part of a healthy forest. Forests that are intensively managed for forest products may eliminate important dead and dying structural components which could result in a lack of habitat and species on those managed landscapes. To ensure forest health for biodiversity, safeguards will be needed to ensure that dead wood remains a component of the forest ecosystem.

4.2.3.1 DWM: Wildlife and Biodiversity

Dead wood is a central element of wildlife habitat in forests (Freedman et al. 1996). Many forest floor vertebrates have benefited or depended on DWM (Butts and McComb 2000). In New England, De Graaf and colleagues (1992) catalogued at least 40 species that rely on DWM.

Some examples from the Northeast of relationships between animals and DWM include a study showing that low densities of highly decayed logs (less than one highly decayed log/ha) had a negative impact on red-back voles (*Clethrionomys gapperi*) in a northern hardwoods forest in New Brunswick, Canada (Bowman et al. 2000). DWM retention increased spotted salamander (*Ambystoma maculatum*) populations in a Maine study (Patrick et al. 2006).

In aquatic environments, DWM provides a crucial refuge from predation (Angermeier and Karr 1984, Everett and Ruiz 1993). Logs that fall in the water formed a critical component of aquatic habitat by ponding water, aerating streams, and storing sediments (Gurnell et al. 1995, Sass 2009). In fact, removal of large woody material from streams and rivers had an overwhelming and detrimental effect on salmonids (Mellina and Hinch 2009).

DWM is a key element in maintaining habitat for saproxylic (live and feed on dead wood) insects (Grove 2002). For example, some specialist litter-dwelling fauna that depend on DWM appear to have been extirpated from some managed forests (Kappes et al. 2009). Extensive removal of DWM could reduce species richness of ground-active beetles at a local scale (Gunnarsson et al. 2004). More generally, a minimum of 286 ft^3/ac (20 m^3/ha) of DWM has been suggested to protect litter-dwelling fauna in Europe (Kappes et al. 2009).

Dead logs serve as a seedbed for tree and plant species (McGee 2001, Weaver et al. 2009). Slash could be beneficial to seedling regeneration after harvest (Grisez, McInnis and Roberts 1994). Fungi, mosses, and liverworts depend on dead wood for nutrients and moisture, and in turn, many trees are reliant on mutualistic relationships with ectomycorrhizal fungi (Hagan and Grove 1999, Åström et al. 2005). In general, small trees and branches host more species of fungus-per-volume unit than larger trees and logs; however, larger dead logs may be necessary to ensure the survival of specialized fungus species such as heart-rot agents (Kruys and Jonsson 1999, Bate et al. 2004).

Implications for Massachusetts Policies: It is clear that dead wood is a central contributor to biodiversity in our forests and that many species are dependent on sufficient quantities and sizes. This requires retention of DWM, standing cull trees and live trees that will eventually create these structures.

4.2.3.2 Habitat and Biodiversity Issues by Forest Type

Northern Hardwood Forests: The number of dead trees in five hemlock-yellow birch forests range from 16 to 45 per ac (40 to 112 per ha) or from 3 to14% of the basal area (Tritton and Siccama 1990). The 14 sugar maple-beech-yellow birch stands survey ranged from 14 to 99 dead trees per ac (35 to 245 per ha) or 5 to 34% of basal area (Tritton and Siccama 1990). Other estimates of snag densities in northern hardwood forests include 5 per ac (11 per ha) (Kenefic and Nyland 2007), 15 per ac (38 per ha) (Goodburn and Lorimer 1998), and 17 per ac (43 per ha) (McGee et al. 1999).

The number of cavity trees is another important habitat element in northern hardwood forests that is reduced by harvest. For example, studies in northern hardwood forests have shown a reduction from 25 cavity tree per acre (62 per ha) before harvest and to 11 (27 per ha) afterward (Kenefic and Nyland 2007). Another study measured 7 cavity trees per ac (18 per ha) in old growth, 4 per ac (11 per ha) in even-aged stand, and 5 per ac (13 per ha) in a stand in selection system (Goodburn and Lorimer 1998).

Transitional Hardwoods: Out of seven oak stands in Connecticut, the number of dead trees ranged from 19 to 44 per ac (46 to 109 per ha) or 5 to 15% of basal area (Tritton and Siccama 1990). The decadal fall rates of snags in a Massachusetts study varied from 52 to 82% (Wilson and McComb 2005). Snags, particularly large-diameter snags, provide important nesting and foraging sites for birds (Brawn et al. 1982, Gunn and Hagan 2000). In general, wildlife habitat requirements for dead wood are poorly documented, but it is clear that some wildlife species rely on dead wood in oak-hickory forests (Kluyver 1961, DeGraaf et al. 1992).

Implications for Massachusetts Policies: The number of standing dead trees varies by forest type in unmanaged stands. Ideally, biomass retention targets would also vary by forest type; but presently there are not enough data across forest types and ages to set specific targets for standing dead trees by forest type. The Forest Guild Retention and Harvesting Guidelines for the Northeast include guidelines with targets for retaining standing live and dead trees that are general for all forest types in Massachusetts. Over time Massachusetts and other state guidelines may be able to hone in on specific targets by forest type.

4.3 Lessons from Other Initiatives: Protecting Stand Level Ecological Values through Biomass Harvest Guidelines

States from Maine to Missouri, Canada, and some European countries have addressed or are addressing stand-level ecological concerns by developing guidelines for harvesting woody biomass from forests. To inform the Massachusetts process, we have expanded on the Forest Guild's report *An Assessment of Biomass Harvesting Guidelines* to provide updates, include additional states in New England, and give a thorough assessment of northern European initiatives. This section begins with an overview of the Guild report highlighting key points relevant to Massachusetts. It concludes with a brief review of the harvesting regulations and BMPs in Massachusetts and the gaps in those directives that indicate that a new set of guidelines is needed.

4.3.1 Overview of Regulatory Frameworks

In the U.S., forestry on private and state lands is regulated primarily at the state level. At least 276 state agencies across the country have some oversight of forestry activities, including agencies focused on forestry and others concerned with wildlife or environment protection policies (Ellefson et al. 2006). All 50 states have BMPs. In general, BMPs originally focused on water quality and did not anticipate the increased removal of biomass. Consequently, BMPs historically have offered little or no specific guidance on the amount of removal that is healthy for ecosystems or how much biomass should be retained. However, this situation is changing. Pennsylvania's old BMPs encouraged operators "to use as much of the harvested wood as possible to minimize debris," while more recent guidelines recommend leaving "15 to 30% of harvestable biomass as coarse woody debris."

Woody biomass is usually considered to be logging slash, small-diameter trees, tops, limbs, or trees that cannot be sold as higher-value products. Depending upon prevailing market conditions, however, material meeting pulp or pallet specifications may also be used in biomass energy facilities. Reasons for biomass harvesting guidelines are likely to mirror the reasons forestry is regulated in general, which include (Ellefson and Cheng 1994):

- general public anxiety over environmental protection,
- the obligation to correct misapplied forestry practices,
- the need for greater accountability,
- growth of local ordinances,
- landscape-level concerns, and
- following the lead of others.

Biomass harvesting guidelines are designed to fill the gaps where existing BMPs may not be sufficient to protect forest resources under new biomass harvesting regimes. In other words, BMPs were developed to address forest management issues at a particular point in time; as new issues emerge, new guidelines may be necessary. State BMP manuals usually include sections on timber harvesting, site preparation, reforestation, stream crossings, riparian management zones, prescribed burning and fire lines, road construction and maintenance, pesticides and fertilizers, and wetlands. These programs are routinely monitored, and literature suggests that when these BMPs are properly implemented they do protect water quality (Shepard, 2006).

U.S. federal law requires states to address non-point source pollution of waterways. State programs vary with some states prescribing mandatory practices while others rely on voluntary BMPs and education and outreach programs. These programs can be categorized in three ways: non-regulatory with enforcement, regulatory, and combination of regulatory and non-regulatory. In the Northeast, Massachusetts and Connecticut are considered regulated; Vermont and New Hampshire are non-regulated with enforcement; and Rhode Island, New York, and Maine use a combination of approaches.

Over time BMPs for water quality have expanded to include aesthetics, wildlife, and other resources. A survey in 2000 noted that nine states had extended their BMPs in such fashion, three of those from the Northeast (NASF Edwards and Stuart). This indicates a precedent for expanding BMPs to include issues such as increased biomass harvesting. In fact, some of the BMPs developed for water quality and conventional forestry already contain guidelines that would serve to protect water quality during increased biomass harvests. When these guidelines were developed, however, they were designed to specifically and solely address the issue of water pollution. Full implementation of these guidelines is necessary for protection of water quality. As harvests become more intense, other ecological issues, such as soil nutrient protection and wildlife habitat, come into play; previous BMPs likely do not account for them.

Although in many cases BMPs are voluntary, water pollution control requirements are not, and therefore landowners are compelled by law to adopt water quality BMPs to avoid legal penalties. This may explain the relatively high rates reported for national compliance (86%) and in the Northeast (82%) (Edwards 2002). Biomass harvesting standards must address several management criteria such as protection and maintenance of forest structure for wildlife habitat, soil nutrient protection, and forest-stand productivity. These criteria, unlike those for water quality, typically have no legal foundation to compel compliance.

The recently updated Forest Guild report, *An Assessment of Biomass Harvesting Guidelines*, reviews the biomass harvesting or retention guidelines from New York and New England, other states with specific biomass guidance, parts of Canada, northern European counties, and other organizations including the U.S. federal government and certification groups. We have grouped New York and the New England states together to offer a snapshot of the current situation in states geographically near Massachusetts. Maryland, Minnesota, Missouri, Michigan, Pennsylvania, Wisconsin, and California are also covered because of their forest practices guidance on biomass harvest and retention.

Entities interested in addressing concerns about biomass removal have taken at least three different approaches. One is to verify that existing forest practice regulations cover the issues raised by biomass harvests, obviating the need for new guidelines. Second, in instances where existing rules or recommendations are found to be insufficient, some entities—including Minnesota, Missouri, Pennsylvania, Wisconsin, and Maine—have taken a different approach and chosen to craft separate biomass guidelines that augment existing forest practice guidance. In the third case, standards-setting entities, such as the Forest Stewardship Council (FSC), have chosen to address concerns particular to biomass harvests in a revision of existing rules or recommendations. The examples in this report detail the status of rules and recommendations for removing biomass from forests.

The existing guidelines cover topics such as dead wood, wildlife and biodiversity, water quality and riparian zones, soil productivity, silviculture, and disturbance. *An Assessment of Biomass Harvesting Guidelines* lists the commonly used subtopics for each and identifies which are covered in a given set of guidelines. In some cases, a subtopic is noted as covered because it appears in another set of forestry practice rules or recommendations instead of that state's biomass guidelines. The list of subtopics was developed from section headings of the existing guidelines and is similar to other criteria for sustainable production and harvest of forest biomass for energy (Lattimore et al. 2009).

4.3.2 KEY FINDINGS FROM *AN ASSESSMENT OF BIOMASS HARVESTING GUIDELINES (REVISED)*

An Assessment of Biomass Harvesting Guidelines reveals a number of approaches to the development of biomass guidelines that provide useful insights for Massachusetts. While not necessarily directly applicable to the ecological conditions in Massachusetts, these approaches illustrate the general types of measures that have been adopted by other states and government entities. Three important questions are addressed:

Do other guidelines offer specific targets backed by scientific research, or are they more general and open to further interpretation?

The ability to assure the public that sustainable forestry is being practiced is often confounded by vagueness and generalities in forestry BMPs or guidelines. Foresters are leery of prescribing targets that are expected to be carried out on every acre of forestland. Each forest stand is subject to different ecological factors, historical trends, disturbance patterns, landscape context, and management intent and should be treated as unique. Despite these difficulties, it is important for the profession to define targets and a system of monitoring to win public confidence and retain what has been called a "social contract" to practice forestry. The struggle between the need to set specific measurable targets and the realities of on-the-ground forestry is now being played out as states and others entities attempt to set biomass harvesting guidelines.

In Maine, the earlier drafts of voluntary guidelines provided specific numeric targets, but the final version is more general (Benjamin 2010). Although background materials refer to specific targets recommended in an important multi-stakeholder report on biodiversity in Maine, targets were not incorporated in the final draft. The final guidelines call for leaving "some wildlife trees" without incorporating the numbers of trees per acre suggested in the report. Also, these guidelines call for leaving "as much fine woody material as possible" without specific requirements for top retention found in other states. Similarly, the Forest Stewardship Council's standards for the U.S. require the maintenance of habitat structure and well-distributed DWM, but are not specific about the amount that should be left on site.

How do other guidelines address the concern over the depletion of soil nutrients?

As noted above, some biomass harvest guidelines call for sufficient material to be retained to protect ecological functions such as soil nutrient cycles but offer no targets. A number of guideline documents, however, do offer targets in this category. The following is a sampling of the various ways retention of DWM has been approached.

- **Alabama**: Enough logging slash should be left and scattered across the area to maintain site productivity.
- **Maine**: Where possible and practical retain and scatter tops and branches across the harvest area.
- **Michigan**: retention of 17% to 33% of the residue less than four inches in diameter.
- **Minnesota**: tops and limbs from 20% of trees harvested.
- **Missouri**: 33% of harvest residue.
- **New Hampshire**: "Use bole-only harvesting (leaving branches and limbs in the woods) on low-fertility soils, or where fertility is unknown."
- **Pennsylvania**: 15 to 30% of "harvestable biomass."
- **Wisconsin**: tops and limbs from 10% of the trees in the general harvest area with a goal of at least 5 tons of FWM per acre.
- **Sweden**: 20% of all slash must be left on site.
- **Finland**: 30% of residues should remain and be distributed evenly over the site.

How do other guidelines address the concern over retention of forest structure and wildlife habitat?

The literature confirms that forest structure is important for wildlife habitat. Existing BMPs and new biomass harvesting guidelines use both general and specific approaches to address this issue. The following samples provide a snapshot of the range of approaches.

- **Maine**: leave some wildlife trees; retain live cavity trees on site; vary the amount of snags, down logs and wildlife trees; and leave as much FWM as possible.
- **New Hampshire**: Under uneven-aged management, retain a minimum of 6 secure cavity and/or cavity trees per acre with one exceeding 18 inches diameter at breast height (DBH) and 3 exceeding 12 inches DBH.
- **California**: retain all snags except where specific safety, fire hazard, or disease conditions require they be felled.
- **Minnesota**: on non-clear cut sites, leave a minimum of 6 cavity trees, potential cavity trees, and/or snags per acre. Create at least 2-5 bark-on down logs greater than 12 inches in diameter per acre.

4.3.3 ADEQUACY OF MASSACHUSETTS BMPS FOR INCREASED BIOMASS HARVESTS

The situation in Massachusetts is very similar to that in other states: current regulations and guidelines were developed for protection of water quality and did not anticipate the intensification of biomass harvesting. In Massachusetts, current regulations require a cutting plan that describes the harvest and the approaches to mitigate water-quality problems such as erosion and sedimentation.

Current regulations and BMPs, however, do not direct silvicultural or harvesting activities to sustain all the ecological values that might be negatively affected by increased biomass harvesting. There are no retention rules or guidelines that would prevent the harvest of every cull tree or den tree on a property, a situation that could take place with or without an expanded biomass market. Similarly, there are no harvesting guidelines that would prevent the scouring of DWM. Our literature review reveals these activities have the potential to degrade wildlife habitat, biodiversity, and soil nutrient levels. In addition, the current cutting plan process does not require sound silvicultural practice and the ecological safeguards that these proven practices offer in comparison to undisciplined harvesting. Finally, the introduction of larger, heavier whole-tree harvesting equipment presents new challenges and opportunities. Larger equipment can damage forest soils through soil compaction and increase residual stem damage because of their size. However, in some cases, new forest equipment can reduce soil impacts because they can provide less pressure per inch and reduce stand damage because of their longer harvesting reach. In practice, some of these impacts are and will be mitigated through good decisions by landowners, foresters and loggers, and the influence of supervising foresters through the cutting plan process. In most situations, however, there are no regulatory or voluntary guidelines in place that compel compliance.

The assessment of guidelines in other states and countries reveals a number of additional approaches that can be tailored to state forest types and conditions to prevent ecological damage from biomass harvesting. We recommend that a similar set of guidelines be developed in Massachusetts and integrated into the cutting plan process. The newly developed *Forest Guild Biomass Retention and Harvesting Guidelines* for the Northeast utilize the best thinking and approaches from other states to develop a set of guidelines for northeastern forest types. These should be directly applicable to Massachusetts and provide a starting point for developing guidelines tailored to the regional ecology and forest types of the Commonwealth.

4.4 FOREST SUSTAINABILITY INDICATORS AND LANDSCAPE LEVEL EFFECTS OF BIOMASS HARVESTING

4.4.1 INTRODUCTION

Beyond stand-level impacts, biomass harvesting has the potential to affect the provision of a broad suite of ecosystem services at larger regional or statewide scales. In this context, we are adopting the ecosystem services definitions used in the recent Forest Futures Visioning Process conducted by the Massachusetts Department of Conservation and Recreation (DCR). These include ecological, socio-economic and cultural values provided by forests—essentially the term ecosystem services refers to all the public and private values provided by our forests. The sustainability of this broad suite of ecosystem services across the landscape is not primarily a scientific problem; instead it involves balancing a complex set of public values that go far beyond simply ensuring that biomass harvests leave a well-functioning ecosystem in place on harvested sites.

Landscape ecological processes operate at varying spatial scales (e.g., across multiple stands, within a watershed, or an entire ecoregion). In the case of forests, the spatial arrangement and relative amounts of cover types and age classes become the ecological drivers on the landscape. The two most relevant ecological processes of interest in Massachusetts' forests include facilitating or blocking movement of organisms and loss of "interior" habitat because of smaller patch sizes. Pure habitat loss is not necessarily a landscape ecological issue until it reaches a threshold where it influences the spatial pattern of habitats. At that time, which will vary by species, the spatial pattern can drive impacts beyond the effects of pure habitat loss. For most species (including plants, invertebrates, and vertebrates), we do not know where this threshold exists (Andren 1994, Fahrig 2003, Lindenmeyer & Fischer 2006). In the discussion below, effects at the "landscape scale" generally refer to loss of habitat at different scales (e.g., watershed, statewide) and we do not attempt to address ecological processes that are influenced by the spatial arrangement of habitats.

The wood supply analysis in Chapter 3 suggests that absent very significant changes in energy prices, we do not expect dramatic increases over the next 15 years in harvest acreage across the state. But that analysis is really focused on overall supplies, and has not attempted to define more localized spatial impacts of these harvests. Moreover, although we do not foresee major changes in electricity pricing that would provide incentives for much heavier harvests, we cannot rule out such an occurrence in the event of a major energy price shock or a change in energy policies that significantly raises long-term prices. Consequently, for any specific bioenergy facility, we cannot rule out that forest impacts are potentially more dramatic within the "wood basket" of the facility than would occur on average across forests in the state.

Such localized, wood basket effects could take the form of rapid reduction or change in the quality of forest cover if many landowners respond to the demand from a new biomass facility by cutting more heavily on acres they would have harvested for timber anyway or by increasing the acreage they decide to harvest. From the ecosystem services perspective, such an increase in cutting could have a variety of effects. First, if enough landowners decide to conduct relatively heavy biomass harvests, we might see a reduction in older forest habitat and a shift to plant and animal species that prefer younger forests. Second, heavier or geographically concentrated cutting by private landowners could have broad aesthetic impacts that might be unacceptable to the public, potentially having negative impacts on other ecosystem services like forest-based recreation or tourism. Third, at a regional scale, increased harvest area or intensity may have long-term implications for the local timber and wood products economy if stands are harvested in a manner that results in a reduction in long-term supplies of high-quality timber. These various effects are discussed below in greater detail.

4.4.2 POTENTIAL ECOLOGICAL IMPACTS OF BIOMASS HARVESTS

The ecological impacts from differing harvest scenarios can be considered at different scales. At the broadest scale—the forested land base of Massachusetts—a total harvest of 32,500 acres per year is approximately 1% of the total land base. This rate of harvest is unlikely to cause statewide ecological changes. The state's forestland is on a trajectory to be comprised of older age classes, and harvests on 32,500 acres will not alter that trajectory significantly other than to provide the opportunity to make small shifts toward younger successional forests. The harvest intensities predicted at the stand level are close to historical ranges, and the total volume of removal is far below growth rates. Other factors such as climate change, rapid land conversion, large-scale disturbance from insect, disease, or hurricanes could all play a cumulative role to cause landscape-wide ecological disturbance, but the harvest scenarios are not widespread enough to have this broad effect alone.

However, landowner response to increased demand from bioenergy facilities could create more significant changes at smaller landscape scales. It is possible that several adjacent landowners or a significant number of landowners in a watershed or viewshed independent of each other could all respond to biomass markets with regeneration cuts over a short time period. Although this cannot be ruled out, the historical trends and landowner attitudes predict otherwise. Historically, rising prices at local sawmills do not appear to have stimulated widespread harvests of sawtimber for parcels nearby. Varying landowner attitudes and goals for their properties apparently work at even the smaller scale to mitigate a mass movement in any one direction of harvest or management, and we expect this to hold for biomass markets as well.

The public's major landscape ecological concern focuses on wildlife habitat and the potential risks to individual or groups of species. The fact is, the abundance of any given species will wax and wane as forest age classes change and as those age classes shift across the landscape. The challenge, whether biomass harvesting becomes prevalent or not, is to make sure that no species declines to a level where it is at risk of being extirpated from the landscape as a result of forest harvesting. Once again, the number of different private landowners and varying nature of private landowner attitudes and behaviors serves to insulate forest landscapes from trends in harvesting strong enough to cause anything other than slight landscape scale changes in habitat or species composition.

Wildlife habitat could potentially be affected at smaller landscape scales (such as a watershed) if many landowners in the wood basket of a power plant suddenly change their historical cutting patterns. If clearcutting or acceleration of regeneration harvests in even-aged stands are used, this could create a loss of mature, interior habitat (depending on the spatial level of harvesting) and species associated with that habitat. Although these species would likely shift elsewhere and still maintain viable populations across broader landscape scales, they might not exist in certain sub-regions for periods of time. Our scenarios do not predict broad-scale clear cutting, and it is more likely that habitat could be affected by practices that are more acceptable to landowners such as more intensive thinnings. One possible scenario for landowners would be to use the new markets for biomass to combine a partial thinning of the dominant trees with a low thinning to remove understory vegetation. If poorly managed, these practices could eliminate certain structural layers from the forest or deplete the forest of the dead and dying material necessary for certain species. The importance of dead wood has been covered elsewhere in the report. The lower forest structure provides important habitat as well. For example birds, particularly long-distance migrants prefer stands with an understory component (Nemi and Hanowski 1984, DeGraaf et al 1998).

In order to gauge the effect that increased biomass harvesting could have on the amount of habitat at the landscape scale, it is instructive to consider neighboring regions. Maine and New Hampshire have a longer history with markets for low-grade material and the introduction of whole tree harvesting and clearcutting for pulp and biomass. How well these landscapes have fared in an ecological sense depends on perspective. If one compares these landscapes to an old growth ideal, they fall resoundingly short. However, a recent review of the ecological literature (Jenkins 2008) for the Northern Forest region indicates the difficulty in quantifying landscape-wide ecological damage.

Jerry Jenkins, a scientist with the Wildlife Conservation Society, reviewed the scientific literature on ecological factors in the intensively managed Northern Forest region for the Open Space Institute. The subsequent report, *Conservation Easements and Biodiversity in the Northern Forest Region*, includes sections on Northern Forest biodiversity and the effects of logging on biodiversity. Although the conclusions of this review are debated in the Northern Forest region, his introduction is helpful in understanding the different perspectives in evaluating landscape ecology. The "pragmatic" approach is to maintain the biodiversity that exists at present. The "idealistic" approach is to restore the forest to a more natural state. Jenkins notes that the pragmatists point to the literature which suggests "there have been almost no losses of vertebrates or higher plants from the working forests and that overall levels of biodiversity in clearcuts and managed forests often exceed those of old, undisturbed forests." The idealists "see the working forest as a conservation failure, and while they grudgingly accept it has considerable biodiversity, they argue that it is the wrong kind." They draw on the general literature of biodiversity and landscape ecology to suggest that our current forests are fragile and impoverished or will become so when the "extinction debt" induced by dissection and fragmentation is finally "paid." These proponents however, have not able to come up with good lists of the species that have actually been lost from managed forests.

The history of the intensively managed industrial landscape of northern New England and New York is far different than Massachusetts. The low harvest rates of the last century have allowed the Massachusetts forests to mature. The current forest landscape of the state offers management possibilities for the pragmatist and the creation of old growth for the idealists. The lessons from the Northern Forest indicate that even in regions with much heavier harvesting the debate over the impacts of changing habitat patterns across the landscape continues unresolved. We can certainly expect this debate to continue in Massachusetts as we try to understand a dynamic and shifting land cover that is resilient but faces a number of pressures. While the number of landowners and their attitudes and behaviors seem to ameliorate the possibility of widespread harvests, there still remains the possibility of localized habitat loss within a watershed as well as stand-level effects. For this reason, in a concluding section we suggest a number of policy options that Massachusetts officials could consider if they wish to assure a greater degree of protection for these ecological values.

4.4.3 POTENTIAL IMPACTS OF BIOMASS HARVESTS ON LANDSCAPE AESTHETICS

The forests of Massachusetts play a number of supporting roles in the socio-economic framework. They are the predominant natural land type and form the backdrop for most communities and many economic enterprises, including tourism and recreation. The forest landscape is integral to the way of life of Massachusetts residents and shapes the image of Massachusetts for visitors and employers locating businesses there. Although historically these forests have been heavily cut, and at one time reduced to 20% of the landscape, the current perception is one of dense unmanaged forests covering most of the landscape. At the more localized or regional scale, biomass harvesting could potentially alter this forest landscape. The heavily harvested forest landscape of northern Maine is one extreme example of what a forested landscape can look like when subject to available markets for low-grade material and landowners willing to harvest using clearcutting and short rotations. From the level of public reaction and media attention paid to clearcutting on public lands in the past, it is expected that broad scale clearcutting on private lands would likely have severe socio-economic impacts for Massachusetts.

While the harvest scenarios do not anticipate broad scale clearcutting, reactions to aesthetic landscape changes are difficult to quantify. The view-shed of most forested areas of Massachusetts now consists of rolling acres of consistent overstory. Even a small amount of clearcutting, consistently repeated across the landscape would dramatically alter these views and probably create a different and negative reaction from tourists or residents. Therefore, any significant increase in clearcutting methods as a form of forest management could have potentially dramatic impacts on recreation and tourism and face significant challenges from residents accustomed to a maturing forest. The quantification of these effects is beyond the scope of this study.

Fortunately, alternative forms of forest management are available including uneven-aged management that maintains a continuous overstory, and forms of even-aged management that delay final harvests until sizable regeneration has occurred. These alternative methods would mitigate the landscape-scale aesthetic effects on tourism and recreation and likely be more acceptable to residents.

4.4.4 POTENTIAL IMPACTS OF BIOMASS HARVESTING ON ECONOMIC PRODUCTIVITY OF FORESTS

Massachusetts forests have historically supported a vibrant forest products industry that has declined dramatically in the last two decades. Although harvest rates of sawtimber remain steady, the number of Massachusetts sawmills and wood product businesses has declined. More of the current harvest leaves the state for processing. The future of this industry is directly connected to a continuing availability of high-quality forest products. The growth and harvest of these higher-quality forest products could be either enhanced or diminished by increased biomass harvesting.

As demand and price for biomass rises, the number and choice of trees removed in harvests change. Trees that previously had no value and were left behind can now be removed profitably or at no cost. We expect that increased demand for biomass will lead to the introduction of whole-tree harvesting equipment on a wider scale, which will enable smaller trees to be harvested more economically. One positive effect of these new markets is to make it possible for foresters to remove portions of the stand that have little future economic value and thus provide growing space for trees with better potential. Without a biomass market, such improvement operations cost money and are typically not possible to perform.

However, new biomass markets may cause the harvest of trees that would eventually develop into valuable crop trees if left to grow. A straight, healthy 10" oak tree that would someday grow to be an 18" high-value veneer log might be removed too early in order to capture its much lower biomass value today. The misuse of low thinnings to remove biomass could also remove the future sawtimber crop as well as the forest structure referred to earlier. Whole tree harvesting equipment may make such removals more profitable, but these trees can also be added to the harvest in conventional operations that use skidders and chain saws. Whether these negative scenarios play out depends on whether the stand is managed with a silvicultural prescription, and that in turn depends on landowner intentions and state regulations for forest management.

4.4.5 EXISTING APPROACHES TO MANAGING LANDSCAPE LEVEL IMPACTS IN MASSACHUSETTS

Historically, Massachusetts has not had programs to manage silviculture and forest harvesting at the landscape (i.e., multi-owner) level. This may be a function of the historical fact that over the last century Massachusetts forests have been recovering from heavy harvesting and deforestation from a prior period when much of the landscape was in agricultural use. In addition, the statewide harvest has been limited in number of acres and intensity. The advent of increased biomass harvesting, the continued loss of forestland to development and the effects of climate change may change the perception of an expanding healthy forest and need for greater oversight of harvesting at the landscape level. While the state does limit the size of individual clearcuts and requires adequate regeneration from harvests and in some cases regulates harvesting in concern for endangered species, nothing in current regulations or guidance limits the ability of private landowners to independently decide to harvest their forests, even if this results in very heavy and rapid cutting in a relatively small area. Furthermore, under the existing regulations, it is theoretically possible for an individual landowner to legally harvest an entire standing forest within a relatively short timeframe (5–10 years) by using a combination of clearcutting and shelterwood harvests.[2]

There are many historical reasons why forest regulatory policy has been implemented at the stand level rather than the landscape level. The focus of existing regulations has generally been aimed at protecting *public* rather than *private* ecosystem services values. For example, BMPs came into existence to protect water quality, which is clearly an ecosystem service that affects the public good—either through off-site contamination of drinking water supplies or damage to public recreational resources. Proposed policies that assert control over ecosystem services that are viewed as purely private in nature have been much more controversial. The recent proposed changes to introduce better silviculture into the Forest Cutting Practices regulations are a case in point where the State Forestry Committee wrestled with these issues and ultimately agreed on an approach that would require sound silviculture practices across all harvests. The practice of silviculture was determined to be a public value and worthy of addressing in the cutting plans. But again, the only controls on forest harvesting now are at the stand level and focused on protecting values that are traditionally considered in the greater public's interest, such as clean water, rare species, adequate forest regeneration, and fire protection. Landscape aesthetics, for example, are not captured by any existing regulation. Voluntary programs, such as land

[2] Shelterwood harvest are heavier cuttings that are intended to regenerate the forest with seedlings but leave a sheltering mix of larger trees that are removed shortly after the regeneration is established.

purchases for conservation through land trusts and the state, have been the mechanism to achieve landscape objectives.

A second hypothesis for the lack of landscape-level forest management policies is a purely practical one. How such controls might be implemented is a difficult question. For example, what type of system would be put in place to decide who can harvest their land and when? Suppose a landowner needs short-term income for a medical emergency or college tuition. It will be difficult for the state to assume too much control over an individual's rights when a widely held public value is not being obviously compromised.

Finally, in the past 50 to 75 years, we generally have not had a forest landscape "problem" caused by over-cutting that the public believed needed to be addressed. Forests have been increasing in both area and wood volume for many years as abandoned farmland has returned to woodland. However, that trend may be changing as urbanization and other land-use changes begin to reduce the amount of forestland in the Eastern U.S. (Drummond and Loveland 2010).

From this discussion, it should be clear that the sustainability of ecosystem services at the landscape level raises a wide array of complex issues involving public values. Forest ecology and science can help inform decisions about the need for an approach to ensuring biomass harvests do not compromise ecosystem services at a landscape scale. But ultimately, public policy on this issue will be a value-based exercise. As a result, our recommendations on this issue, included in the final section of this chapter, focus on options that could be considered as part of a broader process of assessing public perceptions about what would be unacceptable impacts at the landscape level.

4.5 RECOMMENDATIONS FOR ADDRESSING STAND AND LANDSCAPE LEVEL IMPACTS OF INCREASED BIOMASS HARVESTING

4.5.1 STAND LEVEL RECOMMENDATIONS

The science underlying our understanding of the potential impacts posed by increased biomass harvests and the efficacy of policies to minimize these impacts is currently far from providing definitive guidance. While it is clear that DWM is fundamental to nutrient cycling and soil properties, there appears to be little or no consensus on the amount of woody debris that should be maintained. In fact, the literature generally suggests that minimum retention levels will differ based both on underlying site productivity as well as with the volume of material harvested and the anticipated amount of time the stand will have to recover before the next harvest. DWM is also essential for maintaining habitat and biodiversity; but again the scientific studies do not provide a definitive answer to the question of how much DWM should be left after a harvest. The impacts of logging equipment on soils are also likely to depend on site-specific conditions.

Fundamentally, in the face of imperfect scientific information, the choice of policies for protecting ecosystem functions at the stand level must factor in public values regarding how conservative biomass retention policies should be. In addition, it may be important to understand the public's views on the extent to which biomass standards should rely on voluntary or mandatory standards. This likely will depend on the extent to which the public believes the proposed harvest practices are needed to protect public versus private values.

In light of these considerations, Massachusetts may find it useful to utilize the State Forestry Committee to convene an appropriate public process to establish biomass harvesting retention and harvesting guidelines for Massachusetts. The scientific data we reviewed in Section 3 provide a starting point for these public discussions. One approach other states have used is to create a panel of experts from across the spectrum of forestry interests to come up with recommendations which are then reviewed and commented on by stakeholders. The revision of Chapter 132 regulations could easily fit this format by using the State Forestry Committee as the expert panel.

Embedded within our process recommendation is a second broad recommendation that the State Forestry Committee use the *Forest Guild's Forest Biomass Retention and Harvesting Guidelines* for the Northeast as a starting point for the substantive discussion of the options for ensuring biomass harvesting does not result in diminished ecosystem function at the stand level. The Forest Guild's proposed guidelines are readily adaptable to the Commonwealth and cover the major Massachusetts forest types. The Forest Guild Biomass working group consisted of 23 Forest Guild members representing field foresters, academic researchers, and members of the region's and country's major environmental organizations. The process was led by Forest Guild staff and was supported by the previously referenced reports *Ecology of Dead Wood* in the Northeast (Evans and Kelty 2010) and *An Assessment of Biomass Harvesting Guidelines* (Evans and Perschel 2009a).

Wherever possible the Forest Guild based its recommendations on peer-reviewed science. As noted above, however, in many cases available research was inadequate to connect practices, stand level outcomes, and ecological goals. Where this was the case, the Forest Guild relied on field observation and professional experience. The guidelines are meant to provide general guidance and where possible offer specific targets that are indicators of forest health and can be measured and monitored. They are not intended to be applied on every acre. Forests vary across the landscape due to site differences, natural disturbances, forest management, and landowner's goals. All of these elements need to be taken into consideration when applying the guidelines. These guidelines should be revisited frequently, perhaps on a three-year cycle, and altered as new scientific information and results of field implementation of the guidelines becomes available.

In the following section, the Forest Guild's stand-level recommendations for ensuring biomass harvests do not damage ecosystems are examined in. six major categories.

4.5.1.1 Forest Guild Biomass Harvest Guidelines

Site Considerations to Protect Rare Forests and Species

Biomass harvests should be avoided in critically imperiled or imperiled forest types that can be determined through the State National Heritage Program. Biomass harvesting on sensitive sites may be appropriate to control invasive species, but they should only be done for restorative purposes and not to provide a long-term wood supply. Old-growth forest should be protected from harvesting. In Massachusetts, old growth exists exclusively on public lands.

Retention of Coarse Woody Material

A review of scientific literature reveals a limited number of studies that address the biomass and nutrient retention issue. Some studies suggest that biomass harvesting is unlikely to cause nutrient problems when both sensitive sites (including low-nutrient sites) and clearcutting and whole-tree harvesting are avoided. However, there is no scientific consensus on this point because of the wide array of treatments and types of sites that have not yet been studied. Given this lack of consensus, the Guild's recommendations adopt a conservative approach on this issue. They direct harvesting away from nutrient-limited sites. On sites with operable soils, we recommend that between 25% and 33% of tops and limbs be retained in harvests where 1/3 of the basal area is being removed on 15 to 20 year cycles. When harvests remove more trees or harvests are more frequent, greater retention of tops and limbs may be necessary. Similarly, where the nutrient capital is less rich or the nutrient status is unknown, greater retention of tops, branches, needles, and leaves is recommended. Conversely, if the harvest removes a lower percentage of basal area, if entries are less frequent, or if the site is known to have high nutrient levels, then fewer tops and limbs need to be left on site.

In Massachusetts it will be important to identify the soils where there are concerns regarding current nutrient status as well as those soils that could be degraded with repeated biomass harvests. Much of the current harvesting activity falls into the low-frequency and low-removal categories and will require lower levels of retention. It is difficult in most operations to remove all the tops and limbs even if the operator is attempting to do so. In these cases, the retention guidelines may not call for a significant change in operations. If whole-tree harvesting becomes more commonplace, the guidelines would become more important and the balance of acceptable retention and the frequency of harvests and removal intensities a greater issue. Whole-tree operations in some jurisdictions have dealt with retention targets for tops and limbs by cutting and leaving some whole trees that would otherwise have been designated for removal or transporting and scattering a certain percentage of the material back to the woodlot from the landing during return trips to remove additional material.

Retention of Forest Structures for Wildlife and Biodiversity

The Forest Guild recommends a number of approaches for retaining forest structure. All live decaying trees and dead standing trees greater than 10 inches should be left. In areas under even-aged management, we suggest leaving an uncut patch for every 10 acres of regeneration harvest, with patches totaling 5% to 15% of the area. These guidelines also call for maintaining vegetation layers (from the over-story canopy to the mid-story), shrub, and ground vegetation layers to benefit wildlife and plant species diversity. There are targets for retention of downed woody material by weight and forest type. In addition, there are specific targets by forest types for snags, cavity trees, and large downed logs.

In Massachusetts, there has been an awareness of the importance of forest structure for wildlife but no specific guidelines that broadly influence the retention of this material. The targets recommended here can be readily integrated into forest inventories, tree selection, and forest cutting plans.

Water Quality and Riparian Zones

In general, water quality and riparian concerns do not change with the addition of biomass removals. Massachusetts State BMPs currently cover these issues, and habitat management guidelines are available for additional protections for streams, vernal, pools, and other water bodies. These can be integrated into a set of guidelines tailored to Massachusetts.

Silviculture and Harvesting Operations

Most concerns about the operational aspects of biomass harvesting are very similar to all forestry operations. However, some key points are worth mentioning for Massachusetts forestlands:

- Integrate biomass harvesting with other forest operations to avoid re-entering a site and increasing site impacts such as soil compaction.
- Use low-impact logging techniques such as piling slash to protect soil from rutting and compaction.
- Use appropriate equipment matched to the silvicultural intention and the site.

Forest Types

Different forest types naturally develop different densities of snags, DWM, and large downed logs. Currently, available science leaves uncertainty around the exact retention targets for specific forest type and does not provide enough data to provide detailed guidance on each structure for every forest type. The Forest Guild guidelines, however, do discuss the relevant science that is available by forest type. Massachusetts can take that information and augment it with more localized research or prompt new research on specific topics. This information can be used to establish minimum retention targets for Massachusetts forest types. Wherever possible, targets should be exceeded as a buffer against the limitations of current research.

4.5.1.2 Improved Silvicultural Requirements for Forest Ecosystem Management

Finally, we would like to note that Massachusetts has for a number of years been considering changes to the forest cutting plan regulations. In our view, putting these improved silvicultural guidelines in place, while not directly aimed at biomass harvests, will provide greater assurance that Massachusetts forests are managed to maintain ecosystem functions at the stand level. The remainder of this section discusses the current regulatory context and the changes that have been proposed.

Existing Regulatory Framework

Regulations for harvesting forest growth in Massachusetts are guided by intent to promote sound forestry practices and the maintenance of the health and productivity of the forest base. The licensing of foresters in Massachusetts is a recognition of their unique professional education, skills, and experience to practice forestry. One of the keystones of forestry is the practice of silviculture, the art and science of controlling the establishment, growth, composition, health, and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis. Therefore, the argument has been made that all harvesting in the state should adhere to an acceptable form of silviculture and be performed by a licensed professional forester.

The state requires an approved harvesting plan for any harvest over 25,000 board feet. Any harvest is subject to oversight by Natural Heritage and Endangered Speices Program which imposes "life zones" around vernal pools and limits harvesting to certain months of the year in turtle habitat. But most harvested acres are ultimately subject only to requirements indicated in the state approved cutting plan for the property. Unfortunately, the current harvesting plan does not need to be filled out by a licensed forester, nor does it need to follow any accepted form of silvicultural practice.

On the cutting plan, landowners are offered a choice of long-term management and short-term management. A long-term management choice "employs the science and art of forestry." However, the short-term option does not and is characterized as follows:

> *Harvest of trees with the main intention of producing short-term income with minimum consideration given to improving the future forest condition ... [and] the selection of trees for cutting based on the economic value of individual trees which commonly results in a residual forest stand dominated by poor-quality trees and low-value species. While this strategy produces immediate income and meets the minimum standards of the act, it does little to improve the future condition of the forest.*

DCR takes the position that long-term management is the preferred option and warns that the short-term harvests retain slower-growing and poor-quality trees which can limit management options. Still, the short-term option is acceptable and used by 20% of current harvests. This means that aside from restrictions on some harvest areas through the Natural Heritage and Endangered Species Program the door remains open for virtually any kind of harvest as long as it protects water quality and assures adequate regeneration of some kind of tree species- a near certainty in Massachusetts forest conditions. The current system is not designed to assure protection and oversight of a number of ecological and socio-economic sustainability indicators that could be affected by increased biomass harvesting.

Proposed Changes to the Cutting Plan Process

In 2006 the Massachusetts Forestry Committee ended a three-year process where regular public committee meetings were held to completely revise the Chapter 132 Forest Cutting Regulations. By statute, the Committee involves representatives from the key stakeholder interests, and each meeting included a number of public members from various stakeholder groups. The process also involved work in several sub-committees and data analysis from the DCR. The process ended in the spring of 2006 with the Committee completing its voting on a complete package of revisions to the Regulations. The result, supported by the majority of members, was forwarded to DCR in anticipation of public hearings on the Regulations.

Two of the proposed changes are directly related to ensuring that biomass harvesting protects ecological and socio economic values.

- A requirement that all forest cutting be based in silviculture, regardless of the owner's intent, and allowing state foresters to require that trees of high-timber quality be left distributed across the stand after thinning or intermediate cuttings.

- A requirement for marking all trees either to be cut or to be left, regardless of value or cost.

The committee was considering using the silvicultural requirement as a way of getting around opposition to a third suggestion that would mandate that only licensed foresters could fill out a harvesting plan. We recommend that when the Chapter 132 review process begins again, these proposed changes be resurrected in light of the interest in increasing the biomass harvest.

The requirement that all cutting plans be based on silviculture would help assure that biomass harvesting would be ecologically sound and aligned with the long-term economic productivity of the stand. In our view, the requirement for marking trees will also promote good silviculture and ecological practices. However, it may not be necessary in every case, and some flexibility should be considered. These changes would ensure the engagement of professional foresters, require that the harvest be silviculturally sound, and refine the decision making process for selecting trees for harvest by requiring the marking of trees in most cases.

4.5.2 LANDSCAPE LEVEL RECOMMENDATIONS

To determine the need for and nature of approaches to minimizing ecosystem service losses at the landscape-scale as a result of forest biomass harvests, we recommend a public process-based approach. A broad-based and legitimate public process is necessary for addressing landscape-scale impacts of biomass harvesting, particularly because the scientific literature has much less to offer at the landscape scale than it does at the stand level. A key driver of public concerns about diminished ecosystem services at the landscape level is uncertainty about the local and regional impacts of specific bioenergy facilities. Resolving these uncertainties requires gaining a better understanding of the spatial dimensions of harvests for specific proposed facilities. These uncertainties depend on facility size, wood demand, and the extent to which the facility relies on forest versus other biomass. Another uncertainty relates to future energy prices. While landowner reaction to price trends is difficult to predict with accuracy, the likelihood of increased harvests and the concern over landscape-scale impacts increases if policies result in greater use of bioenergy technologies that can afford to pay more for wood (e.g., thermal, CHP, cellulosic ethanol).

Uncertainty, however, will not be the only driver of public preferences. Equally important is how the public perceives and values possible impacts to competing ecosystem services (e.g., renewable energy production versus biodiversity across the landscape), and how risk averse the public is to potential negative impacts of biomass harvesting. Only through a legitimate public process will it be possible to gauge the public's desire for some landscape-level controls on biomass.

With these issues in mind, we have developed some options that could form the basis for a public dialogue on the need for and desirability of policies addressing landscape-scale impacts of biomass harvesting. These range from non-regulatory, information-based approaches to more stringent and enforceable regulatory processes. In general, it may be easier for an individual bioenergy facility to implement voluntary sustainable guidelines for the procurement of their biomass than for a state to implement the same sort of policies. Four possible options are discussed briefly below.

Option 1: Establish a transparent self-monitoring, self-reporting process for bioenergy facilities that includes a commitment towards continual improvement.

Bioenergy facilities could report their procurement status on a year-to-year basis. The report could include a report card that indicated where the supply came from according to a number of assurance criteria. Examples of these criteria can be found in the Forest Guild's A*ssurance of Sustainability in Working Forest Conservation Easements* and the Biomass Energy Resource Center's *Wood Fuel Procurement Strategies for the Harwood Union High School* report. Using a licensed forester or a management plan would be at one end of the assurance of sustainability spectrum. Compliance with the Forest Guild's biomass harvesting and retention guidelines might be in the middle of the spectrum and receiving supply from forest certified by FSC could be one of the highest assurances. Each year the facility would be expected to show improvement.

Option 2: Require bioenergy facilities to purchase wood from forests with approved management plans

If bioenergy facilities were allowed to purchase wood only from landowners with approved forest management plans approved by licensed foresters, there would exist a base level of assurance that biomass energy supplies would be harvested in a manner that would not result in damage, at least at the stand level. Vermont and New York require their biomass power producers to obtain their supply from forests with approved forest management plans. Such a requirement would be a start for Massachusetts facilities, but the harvests should also be certified as having been conducted under an acceptable set of biomass harvesting and retention guidelines. The Forest Guild guidelines or other state guidelines could be used where deemed sufficient, or enrollment in one of the existing forest certification programs that incorporate biomass retention guidelines could work as well.

One wood pellet manufacturer in New York State is supplied by 100% FSC-certified lands. Historically, certification has not been a practical option for a diverse, small forest-ownership land base such as Massachusetts. To the extent that aggregation of land ownerships into certification systems becomes more common, this may become feasible. In addition, the state has recently developed a new program that will allow small owners who seek Chapter 61 property tax exemption for their forest land to prepare "stewardship plans" that will automatically confer third-party certification status on their lands. The biomass facility would periodically report and be evaluated on the ecological and socioeconomic sustainability of the supply. This kind of transparent reporting has proven effective in the toxic waste sector and is applicable to biomass supply.

Another level of assurance is to require the biomass facilities that receive subsidies or incentives to monitor, verify, and report on the sustainability of their supply, including an annual geographic analysis of the facility's geographic wood basket. Some of the supply may come from other states; so the biomass facilities will need to account for supply not produced under the various safeguards that may be instituted in Massachusetts.

Overall, while these approaches improve the likelihood that bioenergy facilities are supporting good forestry practices, they may not be sufficient to fully protect against over harvesting at the local or regional scales.

Option 3: Require bioenergy facilities to submit wood supply impact assessments

This option would require that a facility submit information on its anticipated wood supply impacts as part of the facility siting and permitting process. The facility would identify the area from which it anticipates sourcing most of its forest biomass and would present information on the level of the cut across this region over the life of the facility. As conceived here, this is purely an informational requirement and would not be used as the basis for a positive or negative determination on a permit. But requiring information from a developer on the long-term impacts of their operation on wood supplies within the wood basket of the facility, may result in greater public accountability for the facility and a better understanding of the likely impact on forests. Similar informational programs, such as requiring manufacturing companies to submit information on toxic chemical use, have created positive incentives for improved environmental outcomes.

Option 4: Establish formal criteria for approval of wood supply impact assessments

This option differs from Option 3 in that the state would establish criteria that would have to be met in order for a facility to receive approval for its wood supply impact assessment. For example, possible approval criteria might be based on limits on the amount of harvests relative to anticipated forest growth in the wood basket zone. These could take a variety of forms. For example, the state could require a demonstration that biomass harvests could be conducted without reducing future harvest levels in the wood basket zone (i.e., a non-declining even flow) or other types of limits on how much forest inventories in the wood basket could be reduced over the life of the facility. Once approved, the facility might also be required to submit annual comparisons of actual wood supplies with those included in the approved wood supply impact assessment. Measures could also be put in place requiring corrective actions to be taken by a facility if impacts exceed those anticipated in the impact assessment. Such an approach is more regulatory in nature and likely will be more expensive for facilities but it would give added assurance to the public that local and regional harvests would not diminish broader forest-based ecosystem services.

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CHAPTER 5

FOREST CARBON MODELING: STAND-LEVEL CARBON DYNAMICS AND IMPLICATIONS OF HARVESTING FOR CARBON ACCUMULATION

We evaluated the carbon dynamics of five common forest cover types throughout Massachusetts (Mixed Oak, White Pine, Northern Hardwoods, Hemlock, Mixed Hardwood). We had two primary objectives with this task: (1) to achieve an understanding of Massachusetts forest carbon dynamics and implications of different harvest intensities at the stand level; and (2) to support the forest carbon life cycle accounting analysis (Chapter 6) by providing data on the total carbon recovery rates of forest stands following harvests of varying intensity. Below we summarize the methods used to evaluate forest carbon dynamics and discuss the implications of varying harvest intensities on the carbon volume response by forest stands in Massachusetts.

5.1 FOREST MANAGEMENT AND CARBON SEQUESTRATION

Practices that increase the amount of biomass retained on a given acre over time can be seen as having a carbon benefit. This is particularly true when the removal of the retained biomass (e.g., for pulp wood for paper making) would have generated carbon emissions in a relatively short period of time or emit methane when ultimately disposed. Increased stand-level retention practices consistent with an ecological forestry approach are considered an appropriate mitigation strategy as well. Also appropriate are reduced impact logging practices that minimize soil disturbance and residual damage to stands, thereby reducing mortality and maintaining stand vigor. Under such approaches, late-successional forest structures are seen as beneficial to forest health and resiliency, as well as achieving the biomass levels needed to yield carbon benefits (NCSSF 2008). The relative value of extending rotations is being debated, but there is evidence accumulating that older forests continue to sequester carbon well beyond stand ages we are likely to see in the northeastern forests any time soon (Massachusetts: Urbanski et al., 2007; Globally: Luyssaert et al., 2008). Extending rotation lengths serves to enhance structural complexity, thereby accumulating more biomass on a given acre (Foley et al., 2009). This strategy could also serve to sequester more carbon offsite in long-lived wood products through the production of larger diameter trees suitable for use in these products. However, Nunery and Keeton (2010) showed that even when offsite storage was considered in Northern Hardwood stands, the unmanaged stands still accumulated more carbon over a 160 year time frame. Perez-Garcia et al. (2005) also concluded that offsite storage could not surpass onsite storage unless product substitution was considered. The assumptions made around product conversion efficiencies, decay rates, and the certainty around substitution effects will drive the conclusions about the significance of offsite carbon as a long-term sink associated with forest harvesting (e.g., Van Deusen, in press).

Our modeling of forest carbon dynamics only includes estimates of onsite storage. Chapter 6 incorporates a more complete carbon life cycle accounting of the substitution implications associated with using wood for energy. The role of offsite storage in products is minimal when you consider that only 3.5% of hardwood sawlogs are estimated to be still in use after 100 years in the Northeast (Smith et al., 2006). A significant amount of hardwood sawlogs (28%) is estimated to remain in landfills after 100 years (Smith et al., 2006), but without methane capture technologies in place emissions associated with landfill storage would far exceed the benefits of other offsite storage. Landfill emissions are especially problematic since methane has a Global Warming Potential 25 times worse than carbon dioxide (IPCC, 2007). Without a comprehensive life cycle assessment for products derived from Massachusetts forests we felt it was not productive to speculate on the role of offsite storage, particularly for the time periods we are considering below. More importantly for our analyses however, Chapter 6 assumes that the increase harvest intensity for biomass energy wood doesn't change the disposition of materials that would be harvested absent biomass extraction.

Below we describe the widely-accepted models and inventory data we used to understand the role of forest management in stand-level forest carbon dynamics. Where appropriate, we describe the limitations of the models and data and how they were used to inform the analyses in Chapter 6. Models are a representation of a complex ecological reality and are best used to investigate trends and likely outcomes, not predetermined certainty. Data are generally presented in aggregate to show broad trends, but specific examples are also given to illustrate points.

5.2 INVENTORY DATA AND FOREST CARBON MODELS

Data used in the analyses were based upon Forest Inventory and Analysis (FIA) data from the U.S. Forest Service. We obtained inventory data from the FIA DB version 4.0 Data Mart from 1998–2008.[1] FIA plot data (including tree lists) were imported into the Northeast (NE) Variant of the US Forest Service Forest Vegetation Simulator (FVS)[2] and are accepted as compatible with the model (Ray et al., 2009). FVS is a widely-accepted growth model within current forest carbon offset standards (e.g., Climate Action Reserve Forest Project Protocol 3.1[3] and the Chicago Climate Exchange Forest Offset Project Protocol [4]) and as a tool to understand carbon implications of forest management within the scientific community (e.g., Keeton 2006; Ray et al., 2009; Nunery and Keeton, 2010). The modeling package relies

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1 http://fia.fs.fed.us/tools-data/default.asp

2 http://www.fs.fed.us/fmsc/fvs/

3 http://www.climateactionreserve.org/wp-content/uploads/2009/03/Forest-Project-Protocol-Version-3.1.pdf

4 http://www.chicagoclimatex.com/docs/offsets/CCX_Forestry_Sequestration_Protocol_Final.pdf

on NE-TWIGS (Hilt and Teck, 1989) as the growth and yield model to derive carbon biomass estimates in the Northeast. These growth and yield models are based on data collected by the USFS's Forest Inventory and Analysis unit from the 1950s through the 1980s. Developed by the US Forest Service and widely used for more than 30 years, the FVS is an individual tree, distance independent growth and yield model with linkable modules called extensions, which simulate various insect and pathogen impacts, fire effects, fuel loading, snag dynamics, and development of understory tree vegetation (Crookston and Dixon 2005). FVS can simulate a wide variety of forest types, stand structures, pure or mixed species stands, and allows for the modeling of density dependent factors.

The FVS model modifies individual tree growth and mortality rates based upon density-dependent factors. As would be expected to be observed in nature, the model uses maximum stand density index and stand basal area as important variables in determining density related mortality. The NE Variant uses a crown competition factor CCF as a predictor variable in some growth relationships. Potential annual basal area growth is computed using a species-specific coefficient applied to DBH (diameter at breast height) and a competition modifier value based on basal area in larger trees is computed. In the NE Variant there are two types of mortality. The first is background mortality which accounts for occasional tree deaths in stands when the stand density is below a specified level. The second is density related mortality which determines mortality rates for individual trees based on their relationship with the stand's maximum density. Regeneration in the NE Variant is user-defined (stump sprouting is built in) and we describe the regeneration inputs in more detail below.

The FVS Fire and Fuels Extension includes a carbon submodel that tracks carbon biomass volume based upon recognized allometric equations compiled by Jenkins et al. (2003). The carbon submodel allows the user to track carbon as it is allocated to different "pools." Calculated carbon pools include: total aboveground live (trees); merchantable aboveground live; standing dead; forest shrub and herbs; forest floor (litter, duff); forest dead and down; belowground live (roots); belowground dead (roots). Soil carbon was not included explicitly in this analysis. Our FVS model simulations captured the carbon dynamics associated with the forest floor and belowground live and belowground dead root systems. Mineral soils were not included in our analyses, but appear generally not to be a long-term issue. A meta-analysis published in 2001 by Johnson and Curtis found that forest harvesting, on average, had little or no effect on soil carbon and nitrogen. However, a more recent review (Nave et al., 2010) found consistent losses of forest floor carbon in temperate forest, but mineral soils showed no significant, overall change in carbon storage due to harvest, and variation among mineral soils was best explained by soil taxonomy. It is important to recognize the current scientific uncertainty around the role of timber harvesting in carbon dynamics but the evidence presented to date does not modify our conclusions derived from the modeling.

5.3 MODEL SCENARIOS

FIA data for both private and public lands from inventories between 1998–2008 were imported into a database for manipulation into the FVS model. The most current inventory year from each plot was used in the analysis and grown to the year 2010 using the model described below. Plots were categorized by forest cover type based on tree species list from each plot (Exhibit 5-1).

We selected a subset of the FIA plots that met a condition of having ≥ 25 Metric Tons of Carbon (MTC) per acre of aboveground living biomass ("aboveground live carbon") prior to any harvest in 2010 to represent stands that are typically harvested across the state. This was important to match the assumptions made in the Chapter 3 supply analysis and is consistent with the approach of Kelty et al. (2008). These plots represented a mean aboveground live carbon stocking of 31 MTC/acre (or approximately 124 green tons per acre). We refer to these plots as "operable" stands as they represent the majority of 70-100 year old stands with a likelihood of being harvested in the near term. A total of 88 FIA plots were used for the analyses of operable stands (Mixed Oak n=4; Northern Hardwood n=31; Mixed Hardwood n= 29; Hemlock n=3; White Pine n= 21).

The model scenarios we tested were designed to understand the carbon implications of varying intensity of harvest (i.e., removal rates) including an evaluation of "no management" or "let it grow" scenarios. In particular, we were interested in the implications of harvests that were defined as "biomass" harvests that removed the majority of tops and limbs (65%) and represented higher rates of total removal than that defined as "Business as Usual" (BAU) in supply analysis (Chapter 3). FVS allows the user to

Exhibit 5-1: Cover Type Classification for FIA Plots

Cover Type	Cover Type Code	Dominant Species	Parameter
Mixed Oak	MO	*Quercus* spp. (hickories secondary)	> 50% trees > 5" dbh are *Quercus* spp.
White Pine	WP	Eastern White Pine	> 50% trees > 5" dbh are *Pinus strobus*
Northern Hardwoods	NH	Red and Sugar Maple, Beech, Yellow Birch, Black Birch	> 50% trees > 5" dbh are northern hardwood spp.
Hemlock	HE	Eastern Hemlock	> 50% trees > 5" dbh are *Tsuga canadensis*
Mixed Hardwood	MH	Northern Hardwoods/Mixed Oak	default classification (can contain pine and hemlock)

select and customize forest management scenarios based on input criteria such as target residual basal area (BA), target percent removal, specification of diameter and species preferences, and tops and limbs retention preferences. Twenty scenarios were run using data from all FIA plots representing a range of intensity from no management to a silvicultural clearcut that removed all trees > 2" DBH (Exhibit 5-2). Scenarios are categorized as follows: (1) Unmanaged Accumulation; (2) Business as Usual Harvest (BAU); (3) Biomass Harvests; and (4) Sensitivity Analysis Harvests. The sensitivity analyses were designed to elucidate the carbon dynamics associated with retaining versus removing tops and limbs in biomass harvests and to understand the dynamics of conducting harvests with silvicultural objectives that included promoting crop tree development and moving towards uneven-aged silvicultural systems.

We chose to model carbon accumulation within a period between 2010 and 2100. Modeling on such a time frame comes with a degree of uncertainty and we acknowledge the limitations of this approach. In particular, projections do not include the impacts on carbon accumulation from stochastic natural disturbances, climate change, or the influence of exotic species. However, using these data to understand the potential long-term trajectories is appropriate and can tell us a great deal about response trends.

Exhibit 5-2: Summary of FVS Treatment Scenarios Analyzed

Scenario	Name	Harvest Scenarios	Category	Tops and Limbs Removed From Site (%)	Regeneration Scenario (see Exhibit 5-3)
MS1	Unmanaged	Unmanaged	Unmanaged	0	1
MS2	BAU 32%	Common Partial Harvest (Business As Usual), Thin 25% of stand BA from Above	BAU	0	2
MS3	BAU 32% Light Biomass	BAU with 65% Tops and Limbs Removed	Biomass	65	2
MS4	BAU 32% Heavy Biomass	BAU with 100% of Tops and Limbs Removed	Biomass	100	2
MS5	Heavy Harvest BA 40	Heavy Harvest, Thin from Above to 40 ft2/acre BA	Sensitivity	0	3
MS6	Heavy Harvest BA 40 Light Biomass	Heavy Harvest w/ Light Biomass	Biomass	65	3
MS7	Commercial Clearcut (Tops and Limbs left)	Commercial Clear Cut	Sensitivity	0	4
MS8	Commercial Clearcut	Commercial Clearcut with 65% Tops and Limbs Removed	Biomass	65	4
MS9	Selection Cut	"Quality" Individual Tree Selection (75 ft2/acre BA retained)	Sensitivity	0	2
MS10	Selection Cut Light Biomass	"Quality" Individual Tree Selection (75 ft2/acre BA retained), 65% Tops and Limbs removed	Sensitivity	65	2
MS11	Silvicultural Clearcut	Silvicultural Clearcut No Legacy (>2" DBH trees removed)	Sensitivity	0	4
MS12	Silvicultural Clearcut No Regen	Commercial Clearcut, No Legacy Trees Left, No Regen	Sensitivity	0	x
MS13	DBH BA60	Thinning through diameter classes to BA 60 ft2/acre of trees > 8" DBH	Sensitivity	65	3
MS14	DBH All BA60	Thinning through diameter classes to BA 60 ft2/acre	Sensitivity	65	3
MS15	Biomass BA60	Thin from Above to BA 60 ft2/acre	Biomass	65	3
MS16	BAU 20%	Common Partial Harvest, Thin from Above (15% BA removed = 20% volume)	BAU	0	2
MS17	BAU 20% Light Biomass	Common Partial Harvest, Thin from Above (15 % BA removed = 20% Volume), 65% Tops and Limbs Removed	Sensitivity	65	2
MS18	BAU 35% Light Biomass	Common Partial Harvest, Thin from Above (20% BA removed = 35% volume removed), 65% Tops and Limbs removed.	Sensitivity	65	2
MS19	BAU 40% Light Biomass	Common Partial Harvest, Thin from Above (30% BA removed = 40% volume removed), 65% Tops and Limbs removed.	Biomass	65	2
MS20	BAU 15%	Common Partial Harvest, Thin from Above (10% BA removed = 15% volume)	Sensitivity	0	2

Shorter-term projections (ca. 30 to 50 years) have been verified to have a higher degree of confidence since the impacts of these uncertainties are minimized by low probability of occurrence (Yaussy, 2000). We also focused on the stand-level response following a single harvest event at Time = 0 (i.e., 2010) rather than conduct a more complicated series of repeated harvest entries. We can infer a "sawtooth" response from repeated entries to a target basal area or residual condition, but single entry scenarios provided us the best information to evaluate the short-term impacts and response of stands following "biomass" harvests needed to inform Chapter 6.

The FVS NE Variant does not add regeneration elements by default (except for stump sprouting for appropriate species following harvest). Regeneration inputs were required to more appropriately reflect the behavior of forest stands following harvest. We followed the methods of Nunery and Keeton (2010) and adapted conservative regeneration inputs that were designed to be appropriate to the cover type and disturbance intensity but still within a range of natural variability (Exhibit 5-3). Conceptually, seedling inputs were periodically entered into the simulation throughout the time period to mimic baseline regeneration rates in an unmanaged stand. In harvested stands, larger numbers of seedlings were input immediately post harvest to mimic the pulse of regeneration that would be expected to follow a disturbance. Exhibit 5-3 shows the number of seedling inputs relative to the harvest scenario. Greater removal of overstory trees promotes the opportunity for larger numbers of seedlings to become established. The mix of species in heavier harvests was weighted more heavily to shade intolerant and intermediate shade tolerant species as would be expected following an actual harvest (after Leak et al. 1987 and Leak 2005). Regeneration inputs in harvested stands were then gradually reduced over time to mimic a stand initiation period followed by baseline regeneration. Site indices were inconsistently available for the FIA dataset so we used the default FVS value set to sugar maple with a site index of 56.

Exhibit 5-3: Regeneration Inputs Used in FVS Model Scenarios

Regeneration Group	Harvest Scenarios	Year										
		2015	2025	2035	2045	2055	2065	2075	2085	2095	2105	2115
1	Unmanaged Baseline Regeneration	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000
2	Light Partial Harvest Response	2,500	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000
3	Heavy Partial Harvest Response	5,000	2,500	2,500	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000
4	Commercial Clearcut Response	20,000	5,000	2,500	1,000	1,000	1,000	1,000	1,000	1,000	1,000	1,000

Cover Type	Shade Tolerance			
	Intolerant	Intermediate	Tolerant	Total
HE	16%	21%	63%	100%
MH	33%	40%	27%	100%
MO	23%	43%	34%	100%
NH	18%	54%	28%	100%
WP	32%	31%	37%	100%
Mean	24%	43%	33%	100%

Note: Species were allocated based on proportional representation within each cover type and weighted to reflect a higher proportion of intolerant and intermediate shade tolerant species in the Heavy Partial Harvest and Commercial Clearcut scenarios.

Note: Regeneration is expressed in trees (seedlings) per acre. Inputs based on methods described in Nunery and Keeton (2010) and regeneration response to harvests described in Leak et al. (1987), Hornbeck and Leak (1992), and Leak (2005) (5-3a).

5.4 GENERAL RESULTS AND MODEL EVALUATION

5.4.1 GENERAL RESULTS

All values below are expressed in terms of Metric Tons of Carbon per Acre (MTC/acre). Approximately 50% of dry wood weight is considered to be made up of carbon (or 25% of green wood weight). We also present values either in terms of Total Stand Carbon (TSC) or Aboveground Live Carbon (AGL). AGL is simply the carbon biomass associated with the aboveground elements of a live tree. TSC is comprised of aboveground live and dead trees, belowground live and dead roots, lying dead wood, forest floor, and shrub and herb carbon pools. AGL dynamics reflect behavior foresters would be more accustomed to and are analogous to stand basal area and merchantable volume response. Basal area to AGL relationships are typically more linearly related than AGL and merchantable volume (Ducey and Gunn, unpublished data).

Not surprisingly, unmanaged stands result in greater onsite carbon storage than any of the management scenarios we simulated when both TSC and AGL are considered over the 90 year horizon (Exhibits 5-4a and 5-4b). Here, a range of management scenarios (including unmanaged) are shown to illustrate the response of a light diameter-limit partial harvest, a heavy harvest that removes 65% of the tops and limbs, and a commercial clearcut that removes all trees greater than 5" DBH. The mean values include both public and private landowners, and all cover types are aggregated. These patterns were also observed by Nunery and Keeton (2010) in Northern Hardwood stands and even held true when offsite storage of carbon was considered. There were a few plots where managed stands met or exceeded the unmanaged scenario by 2100. These plots were typically understocked at the time of harvest and a heavy harvest was able to "release" the advanced regeneration and promote the growth of the intolerant and intermediate shade tolerant species that were input following the harvest. These fast growing species begin to decline after 40 to 50 years and it is likely that a decline would be observed beyond our modeling period as a result of mortality in these short-lived species. If longer-living shade tolerant species were present in the pre-harvest canopy or mid-story, it is likely that these species would persist longer than the intolerants in the managed scenario.

Exhibit 5-4a: Total Stand Carbon Accumulation over Time
(see next page)

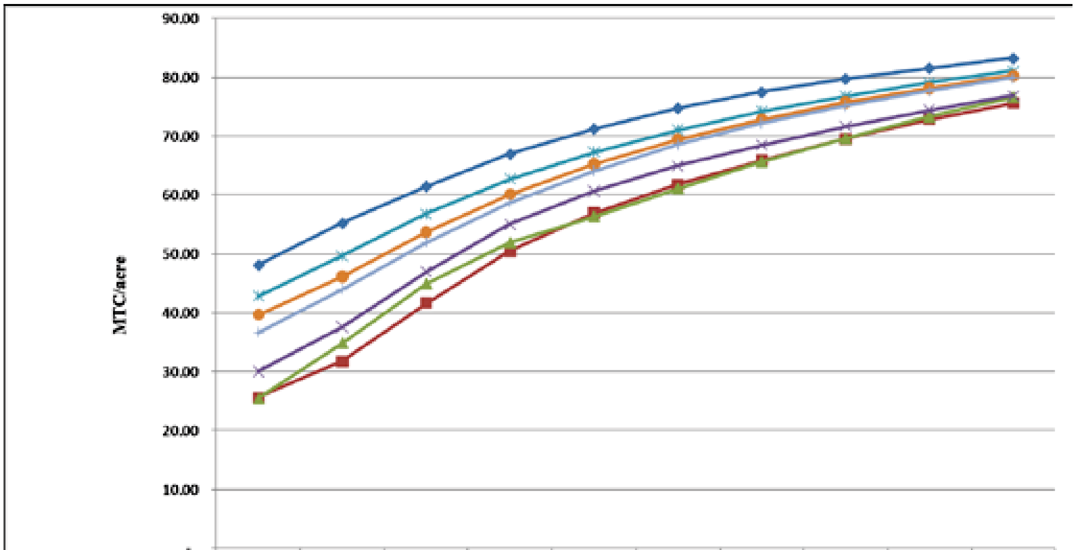
5-4b: Aboveground Live Carbon Accumulation over Time
(see next page)

Light partial harvests in stands that remove larger diameter trees recover slowly and roughly parallel to unmanaged stands, but gradually approach unmanaged volumes over a 90-year period. This is likely because residual mean diameter is still relatively high following the harvest and the associated growth response is slow. These light diameter-limit partial harvests (e.g., BAU 20% and BAU 32%) represent the mean harvest intensity across Massachusetts. The light harvest in the canopy increases the growth rate in the initial ten year period, but very quickly returns to approximately the same as the unmanaged growth rate. Over time these BAU stands approach unmanaged stocking but don't quite catch up after 90 years. This finding is consistent with work in the Harvard Forest by O'Donnell (2007) who found that carbon uptake in live biomass following a light partial harvest recovered quickly after an initial decline to equal the un-harvested control site's carbon uptake rates. If this relationship holds into the future, the onsite stocks would not catch up to the unmanaged site. In contrast, the scenarios we defined as "biomass" harvests (Biomass 40%, Biomass BA40, Biomass BA60) maintain high growth rates for several decades. Because of this increased growth rate, even the heavier harvested stands can reach almost 90% of the volume that could have been achieved in an unmanaged scenario. So, over a long period of time, biomass harvests have an opportunity to recover a large portion of the carbon volume removed during the harvest. However, this assumes no future harvests in the stand as well as an absence of any significant disturbance event. Both are unlikely. This return interval, or cutting cycle, in a silvicultural system will clearly play a role in the recovery of onsite carbon storage over time. If stands are consistently entered prior to achieving complete recovery, the result will be a declining "sawtooth" pattern of growth and recovery of carbon volume stored onsite. With planning and monitoring, uneven-aged silvicultural systems can be implemented that allow adequate time for recovery while maintaining a basal area that promotes quality sawlog production (Hornbeck and Leak, 1992).

Canopy and sub-canopy density plays an important role when the harvest is not heavy enough to reduce the crown completion factors. Heavy harvests create light and space for fast growing intolerant hardwood species to succeed, which can create a pulse of fast growing AGL. The heavy harvest also generates more lying dead wood from the tops and limbs. This may keep the initial post-harvest TSC value high, until this material decays and is lost from subsequent carbon pools. However, this loss is very rapidly recovered by the fast growing species. The curves in Exhibits 5-4a and 5-4b show the general pattern of a faster growth rate in the periods immediately following a harvest event, followed by a gradual slowing at the end of the modeling period. This is not surprising particularly for the unmanaged scenario which would represent plots that are reaching ages around 200 years old by the end of the modeling period. The FIA data that forms the basis of the NE Variant modeling would have had few plots that represented stands of this age, so accumulation behavior this far out in time is uncertain and requires further research (e.g., Keeton et al., In Press).

The Heavy Harvest (BA40) and Commercial Clearcut harvest scenarios behave very similarly to each other. This is largely because the Commercial Clearcut retained trees greater than 5" DBH which effectively brought the stand to 40 ft^2/acre of basal area. Depending upon the density of trees > 5" DBH in the plot, the Heavy Harvest could actually be a heavier harvest than the Commercial Clearcut—which may explain the greater carbon accumulation after 2020. Note that Total Stand Carbon is actually higher for a time in the Commercial Clearcut plots, possibly a product of mortality from the regeneration inputs that are lost through density competition within the smaller stems in that scenario. When we look at the impacts of a Silvicultural Clearcut that removes trees down to 2" DBH, it becomes obvious that there are immediate carbon benefits (AGL) to leaving behind advance regeneration when it is available (Exhibit 5-5). Even though 20,000 seedlings per acre are being input into the stand following harvest, it takes some time before those stems contribute significantly to the AGL, eventually the curve approaches the Commercial Clearcut, but not before 100 years.

Exhibit 5-4a: Total Stand Carbon Accumulation over Time

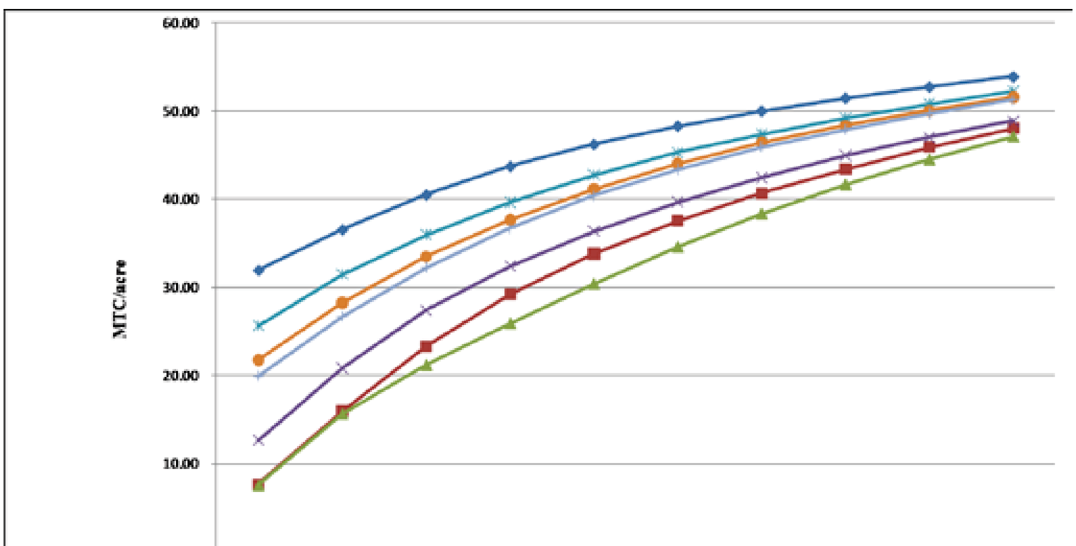


	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Unmanaged	48.09	55.21	61.50	66.97	71.27	74.78	77.52	79.77	81.60	83.27
Biomass BA40	25.61	31.74	41.63	50.44	56.91	61.78	65.90	69.51	72.86	75.60
Commercial Clearcut	25.35	34.88	44.92	51.89	56.36	61.08	65.61	69.61	73.28	76.61
Biomass BA60	30.01	37.58	46.97	55.00	60.62	64.92	68.47	71.63	74.42	76.89
BAU 20%	42.81	49.66	56.82	62.69	67.25	71.05	74.18	76.81	79.05	81.06
BAU 32%	39.62	46.12	53.68	60.08	65.18	69.43	72.84	75.74	78.11	80.27
Biomass 40%	36.67	44.00	51.94	58.74	64.10	68.50	72.17	75.12	77.64	79.87

Year - MTC/acre (Total Stand Carbon)

Note: Plots included are from FIA plots with >25 MTC/acre of Aboveground Live Carbon (pre-harvest) in 2010. Private and public owners and all cover types are aggregated (see Exhibit 5-2 for harvest scenario descriptions).

Exhibit 5-4b: Aboveground Live Carbon Accumulation over Time

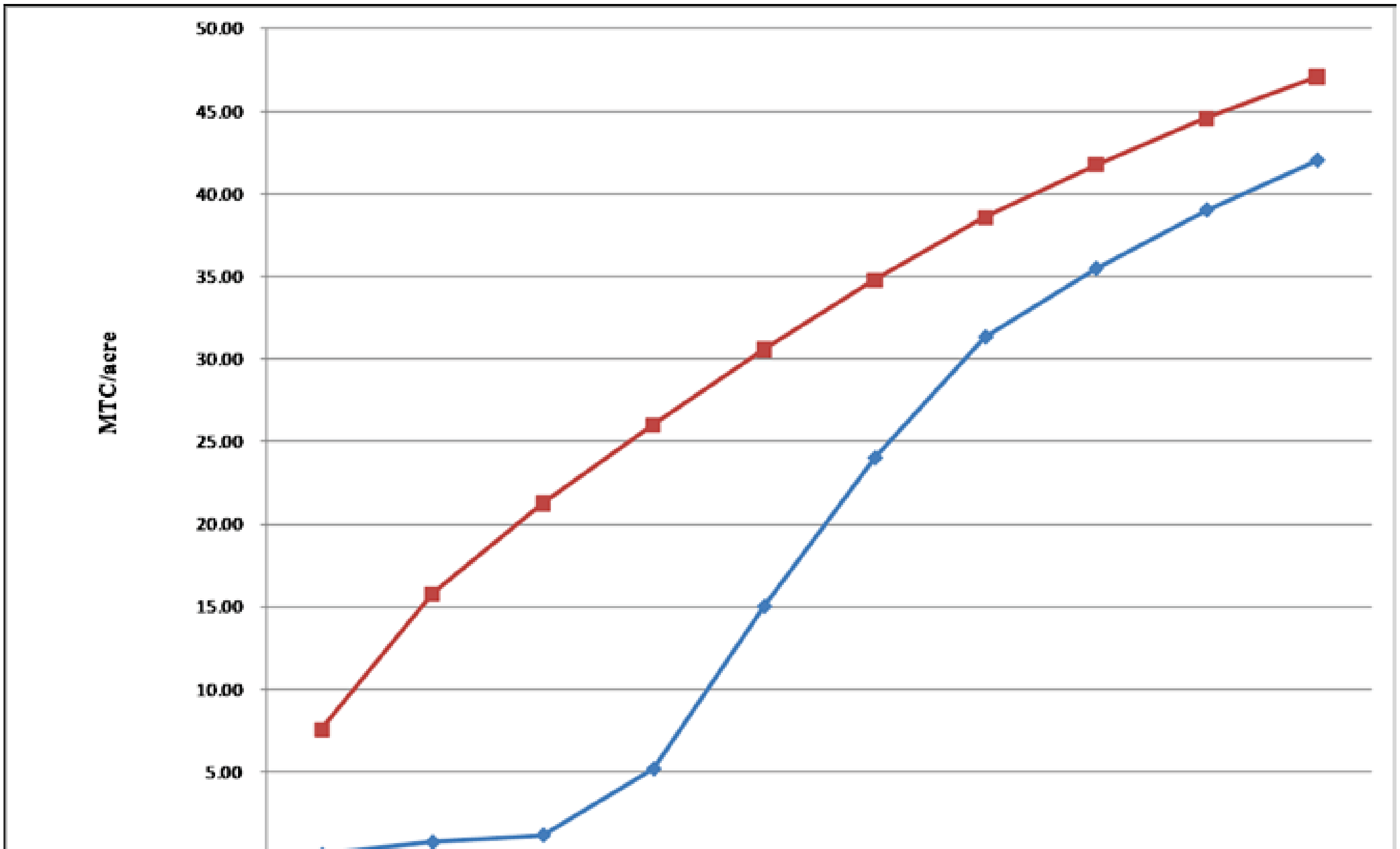


	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Unmanaged	31.91	36.53	40.49	43.70	46.22	48.30	50.02	51.47	52.75	53.90
Biomass BA40	7.66	15.93	23.29	29.22	33.81	37.53	40.66	43.37	45.92	48.01
Commercial Clearcut	7.53	15.65	21.16	25.90	30.39	34.57	38.37	41.62	44.47	47.09
Biomass BA60	12.62	20.83	27.40	32.40	36.38	39.67	42.42	44.91	47.01	48.87
BAU 20%	25.64	31.47	35.98	39.64	42.71	45.27	47.38	49.20	50.80	52.22
BAU 32%	21.76	28.22	33.49	37.63	41.15	44.01	46.38	48.34	50.05	51.56
Biomass 40%	19.93	26.67	32.27	36.68	40.39	43.36	45.88	47.90	49.71	51.26

Year - MTC/acre (Aboveground Live Carbon)

Note: Plots included are from FIA plots with >25 MTC/acre of Aboveground Live Carbon (pre-harvest) in 2010. Private and public owners and all cover types are aggregated (see Exhibit 5-2 for harvest scenario descriptions).

Exhibit 5-5: Aboveground Live Carbon Accumulation Following Clearcut Harvests



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Silvicultural Clearcut	-	0.75	1.17	5.18	15.01	23.99	31.31	35.46	39.00	42.00
Commercial Clearcut	7.57	15.77	21.25	26.01	30.56	34.77	38.56	41.76	44.54	47.07

Note: Comparison is between a Commercial Clearcut (removing trees >5" DBH) vs. Silvicultural Clearcut (removing trees > 2" DBH).

Aboveground Live Carbon typically follows a pattern of faster growth when mean diameters are small and densities are not limiting; then slows down as basal area maximums are reached and the lifespan maximums are approached. This is typical of what would be expected based on principles outlined in Oliver and Larson's classic Forest Stand Dynamics text (1996). Total Stand Carbon provides interesting insight primarily in the short term responses of stands as carbon pools are influenced by material left on the site. Later in the trajectory, the TSC becomes interesting again as mortality occurs and contributions of material to the dead standing and lying dead pools can vary.

5.4.2 COVER TYPE AND OWNERSHIP DIFFERENCES IN CARBON ACCUMULATION

Species response rates can vary depending upon silvical characteristics and this can be illustrated in some variation among cover type responses. Below are some examples of variation among cover types (Exhibits 5-6a through 5-6c). In general, the patterns are similar. The differences occur in terms of starting carbon volume and then become more pronounced near the end of the modeling period. For example, the Hemlock cover type accumulates the greatest amount of carbon over the long term as would be expected from a shade tolerant and long-lived species. However, these curves are based on only 3 plots, so a larger sample might bring it in line with other types. In addition, the future of Hemlock in Massachusetts is highly uncertain given the current status of the Hemlock Woolly Adelgid. For the other cover types, response to harvests (Exhibits 5-6b and 5-6c) generally follows the same trends with the real differences being accentuated late in the model period as with the Hemlock. Though there are minor differences among the cover types, we generally will report the results in Chapter 6 in aggregate.

Likewise, for the purposes of this analysis, we aggregated plots regardless of ownership type (Public and Private). Ownership does not result in major differences in terms of carbon trajectories and response to harvests (e.g., Exhibit 5-7). Minor differences do occur in starting carbon volume, but the plots behave similarly over time. Kelty et al. (2008) documented differences in growth between ownership types but were using two different data sets to make those comparisons (FIA for private lands and MA DCR Continuous Forest Inventory for public lands). Utilizing the Continuous Forest Inventory Plots from the MA DCR proved to be logistically challenging to integrate into FVS with the FIA plots data. Since data were available for both Public and Private lands within FIA, we decided to maintain consistency by only using FIA data.

Exhibit 5-6a: Unmanaged TSC Accumulation by Cover Type (see page 91)

Exhibit 5-6b: BAU 32% Removal TSC Accumulation by Cover Type (see page 91)

Exhibit 5-6c: Heavy Harvest BA40 TSC Accumulation by Cover Type (see page92)

Exhibit 5-7: Ownership Similarities in Carbon Accumulation Over Time by Cover Type (TSC) (see page 92)

5.4.3 REGENERATION CONTRIBUTION TO CARBON ACCUMULATION

Appropriately reflecting a realistic regeneration scenario is an important component of extending the time frame in which the FVS model results can be meaningful. Simply put, regeneration fills space made available by disturbances or natural mortality. In our simulations, we have followed the basic principle that heavier disturbances create more space and light, and therefore allow increasing larger numbers of seedlings to become established. Lighter harvests create less space and light in which regeneration will be successfully established. The successful seedlings will be appropriate to the amount of shade they can tolerate. Regeneration species composition is generally related to species already present within a stand and adjacent stands. But heavy harvests in the NE would typically result in 2/3 of the regenerating species being either shade intolerant or intermediate tolerance. Biologically relevant amounts and species composition were integrated into our approach.

The silvical characteristics of the regeneration are the primary factor contributing to forest carbon dynamics over time. Shade intolerants are typically faster growing species, but they are shorter lived. Thus, they can be responsible for an immediate increase in carbon biomass but will slow and decline after 50–60 years, whereas shade tolerant and intermediate shade tolerant species would persist in the stand and continue accumulating carbon for a longer period. However, Exhibit 5-5 above illustrates that the interaction between starting condition and the amount removed during a harvest are major drivers of carbon accumulation after a harvest.

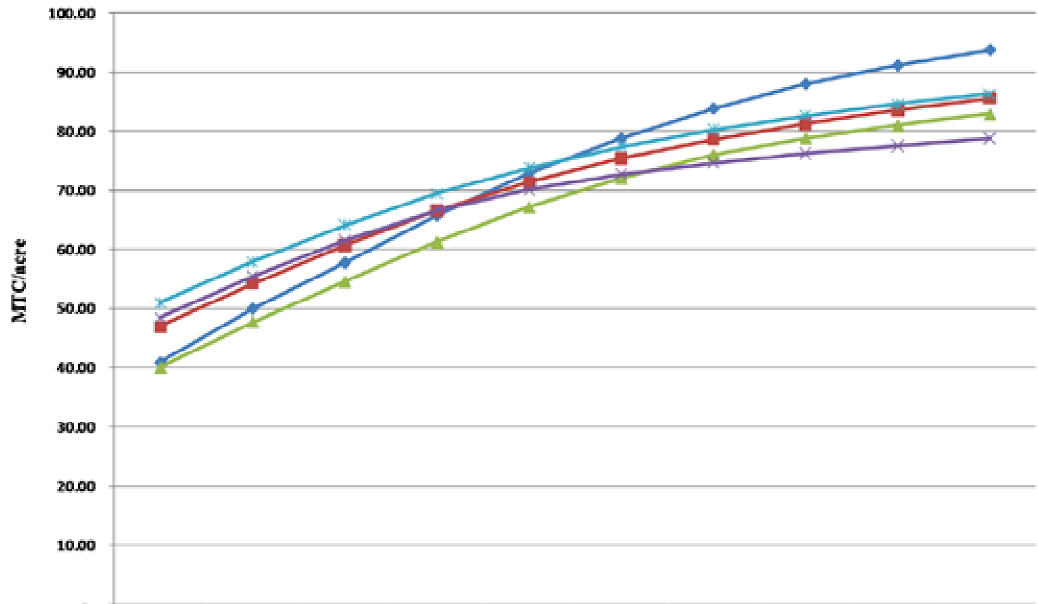
5.4.4 ROLE OF TOPS AND LIMBS IN CARBON BUDGET

We evaluated the carbon implications of the removal of tree tops and limbs during a harvest. We chose to simulate a removal rate of 65% tops and limbs based upon the standards recommended in Chapter 4 and the operability limitations described in Chapter 3. Removal of 65% tops and limbs generates on average between 1.23 MTC/acre and 4.22 MTC/acre depending on the intensity of the overall harvest. This carbon volume decays very rapidly if left on the forest floor, but is compensated for by new growth generally within 10 years following the harvest (Exhibit 5-8). The tops and limbs left in the forest can be observed as a pulse of carbon in the "lying dead" carbon pool, but it moves relatively quickly into the forest floor and ultimately is mostly lost to the atmosphere within a short time period (e.g., Exhibit 5-9). Thus, if tops and limbs are harvested in one scenario, and left in another, Total Stand Carbon in both scenarios will nearly converge within one decade. This recovery of carbon lost from tops and limbs could theoretically be faster if there is significant material left onsite suppressing regeneration. Overall, the model results indicate that the removal of tops and limbs is generally a minor stand level carbon issue; however, as shown in Chapter 6, they can have a significant impact on carbon recovery profiles if they represent a significant proportion of the total harvest.

Exhibit 5-8: Tops and Limbs Contribution to Total Stand Carbon (see page 93)

Exhibit 5-9: Carbon Pool Comparison (see page 93)

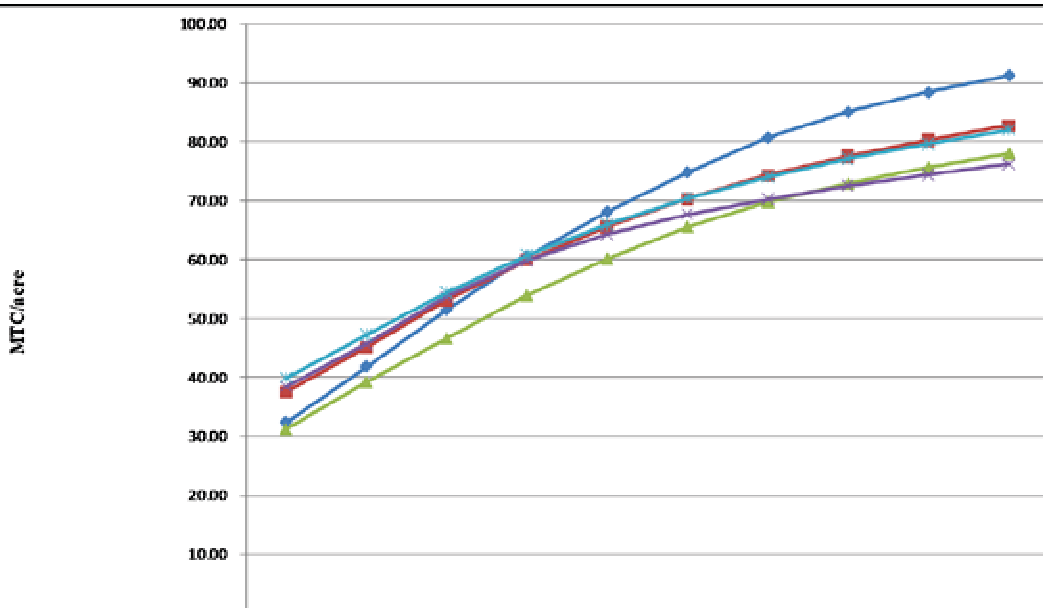
Exhibit 5-6a: Unmanaged TSC Accumulation by Cover Type



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
HE - Unmanaged	40.89	50.01	57.77	65.77	72.94	78.82	83.85	88.02	91.15	93.79
MH - Unmanaged	46.98	54.24	60.72	66.60	71.38	75.46	78.68	81.34	83.57	85.61
MO - Unmanaged	40.02	47.77	54.54	61.30	67.21	72.16	76.12	78.89	81.07	82.98
NH - Unmanaged	48.45	55.44	61.56	66.58	70.11	72.75	74.70	76.31	77.58	78.78
WP - Unmanaged	50.94	57.92	64.15	69.46	73.82	77.40	80.26	82.64	84.61	86.33

Year - MTC/acre Total Stand Carbon

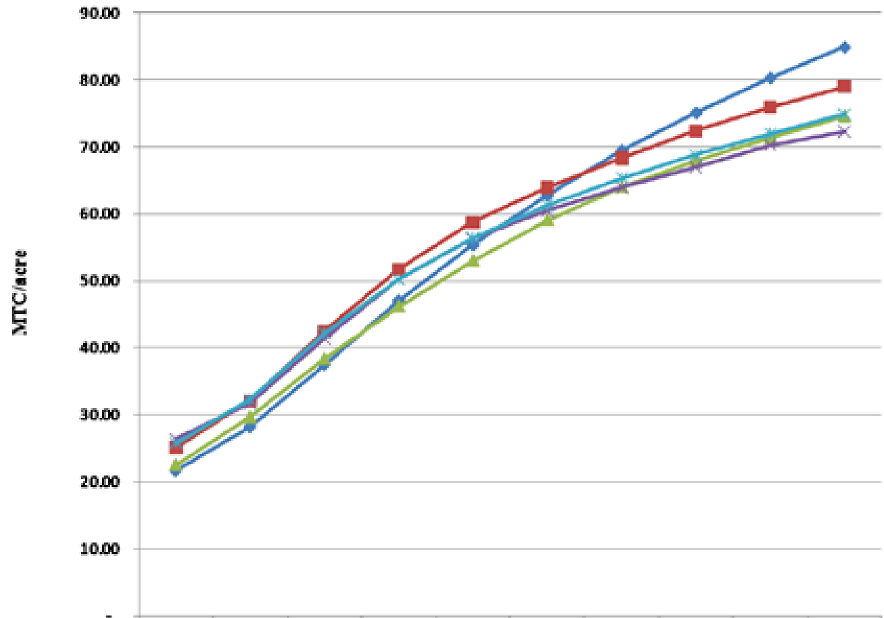
Exhibit 5-6b: BAU 32% Removal TSC Accumulation by Cover Type



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
HE - BAU 32% (65% T&L Removed)	32.40	41.86	51.42	60.49	68.17	74.85	80.73	85.06	88.40	91.32
MH - BAU 32% (65% T&L Removed)	37.48	45.04	53.10	59.98	65.55	70.33	74.30	77.58	80.31	82.72
MO - BAU 32% (65% T&L Removed)	31.19	39.21	46.63	53.88	60.13	65.56	69.75	72.89	75.59	77.99
NH - BAU 32% (65% T&L Removed)	38.46	45.70	53.72	59.86	64.32	67.76	70.26	72.56	74.38	76.19
WP - BAU 32% (65% T&L Removed)	39.98	47.28	54.42	60.79	65.98	70.47	74.07	77.16	79.68	82.00

Year - MTC/acre Total Stand Carbon

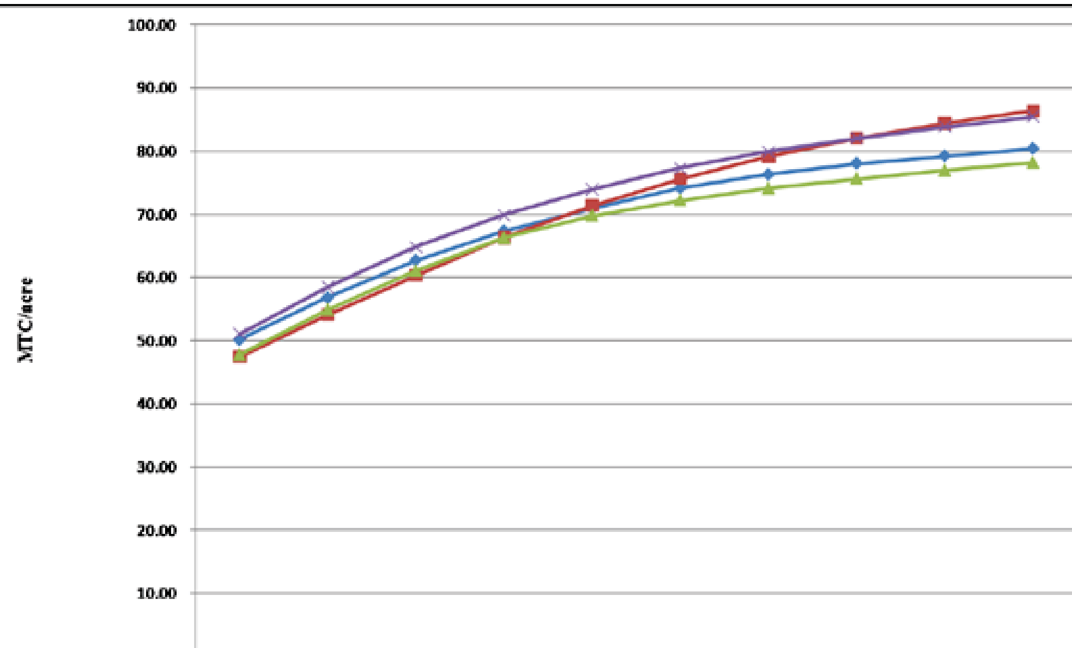
Exhibit 5-6c: Heavy Harvest BA40 TSC Accumulation by Cover Type



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
HE - Heavy Harvest BA40 (65% T&L Removed)	21.60	28.16	37.46	47.03	55.32	62.73	69.47	75.07	80.29	84.87
MH - Heavy Harvest BA40 (65% T&L Removed)	25.09	32.02	42.46	51.75	58.73	63.90	68.28	72.40	75.92	78.92
MO - Heavy Harvest BA40 (65% T&L Removed)	22.53	29.63	38.34	46.19	52.98	59.08	64.02	67.99	71.33	74.52
NH - Heavy Harvest BA40 (65% T&L Removed)	26.36	31.70	41.37	50.25	56.28	60.45	63.91	66.95	70.26	72.27
WP - Heavy Harvest BA40 (65% T&L Removed)	25.73	32.23	42.13	50.20	56.37	61.28	65.32	68.78	71.93	74.83

Year - MTC/acre Total Stand Carbon

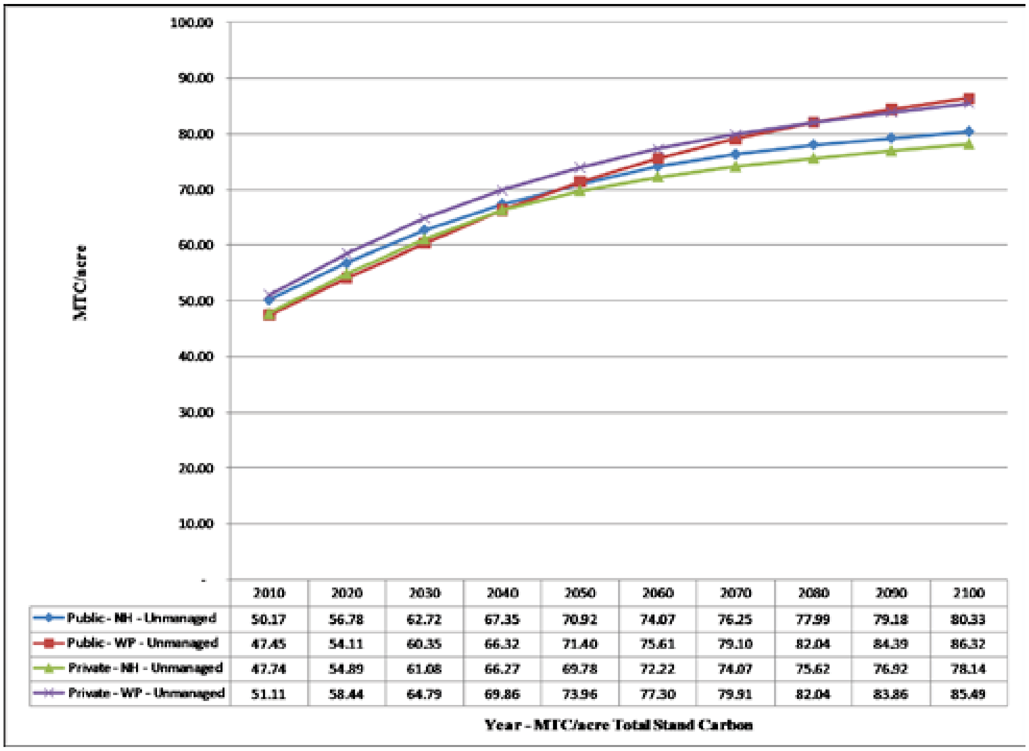
Exhibit 5-7: Ownership Similarities in Carbon Accumulation Over Time by Cover Type (TSC)



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Public - NH - Unmanaged	50.17	56.78	62.72	67.35	70.92	74.07	76.25	77.99	79.18	80.33
Public - WP - Unmanaged	47.45	54.11	60.35	66.32	71.40	75.61	79.10	82.04	84.39	86.32
Private - NH - Unmanaged	47.74	54.89	61.08	66.27	69.78	72.22	74.07	75.62	76.92	78.14
Private - WP - Unmanaged	51.11	58.44	64.79	69.86	73.96	77.30	79.91	82.04	83.86	85.49

Year - MTC/acre Total Stand Carbon

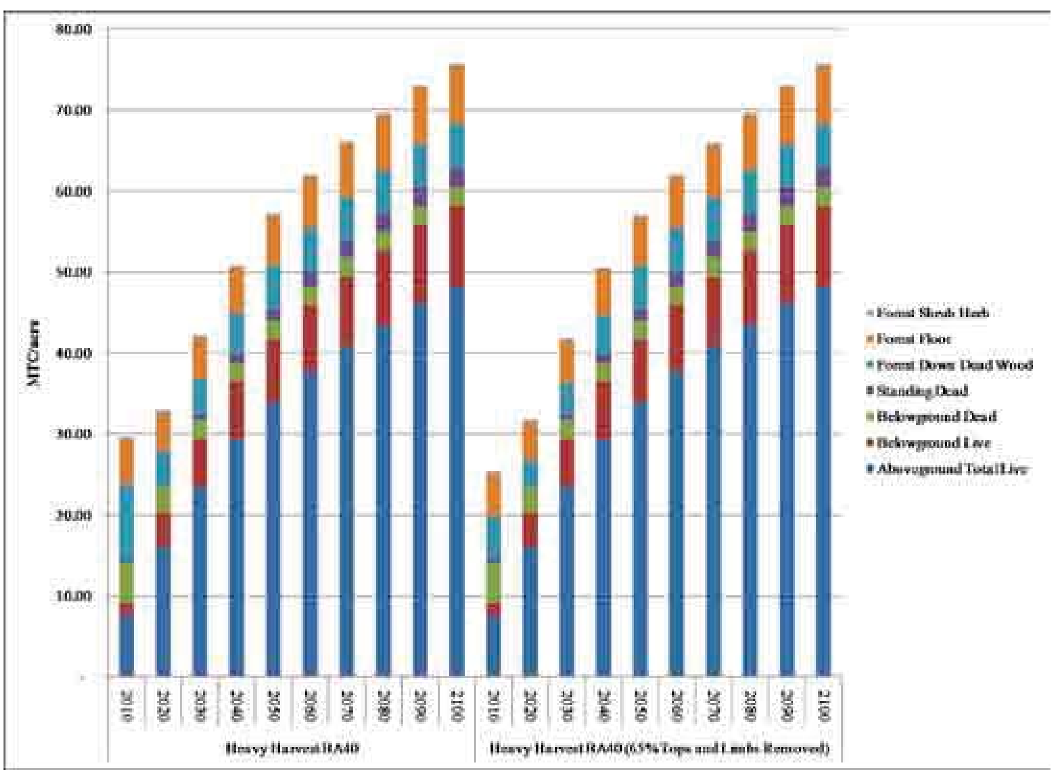
Exhibit 5-8: Tops and Limbs Contribution to Total Stand Carbon



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Public - NH - Unmanaged	50.17	56.78	62.72	67.35	70.92	74.07	76.25	77.99	79.18	80.33
Public - WP - Unmanaged	47.45	54.11	60.35	66.32	71.40	75.61	79.10	82.04	84.39	86.32
Private - NH - Unmanaged	47.74	54.89	61.08	66.27	69.78	72.22	74.07	75.62	76.92	78.14
Private - WP - Unmanaged	51.11	58.44	64.79	69.86	73.96	77.30	79.91	82.04	83.86	85.49

Note: Comparison of harvest scenarios with all tops and limbs retained onsite following harvest versus removing 65% of tops and limbs (BAU 32%, Heavy Harvest BA40, and Commercial Clearcut). Total Stand Carbon values reflect the movement of carbon from tops and limbs into the down dead and forest floor carbon pools over time.

Exhibit 5-9: Carbon Pool Comparison



Note: Carbon pools after a Heavy Harvest (BA40) when 100% of Tops and Limbs are retained vs. 65% removed.

5.5 CONCLUSION

What do we know about modeling carbon accumulation patterns with and without harvesting?

- The basic elements of stand dynamics (and thus carbon dynamics) are the interaction among: space, light, species silvical characteristics (how they grow, regenerate, light tolerance, moisture tolerance), and site characteristics. The FVS model handles the first three elements quite well. We have held the fourth element constant throughout.

- Starting condition matters. No two acres of "natural" forest will be exactly alike. Stand development (particularly in the form of carbon growth and yield) is the reflection of the unique attributes of a given acre, but broad patterns are somewhat predictable based on what we know about the silvical characteristics of individual species and how they interact with each other. Starting diameter distribution (or mean diameter) is a driver of carbon accumulation rates since growth rates will depend on the current diameter of the individual trees making up the stand.

- Basal Area (square feet per acre) in combination with Trees Per Acre (density) is a driver of carbon accumulation rates since it reflects the space available to grow and regenerate.

- Species composition of a plot/stand is also a driver. Differential rates of growth drive differential rates of carbon accumulation. Allometric equations of hardwood vs. softwood are also a factor (e.g., taper, tree architecture).

- The above elements all relate to the influence of stand history on current conditions as well. This history includes the impacts of prior harvests and stand origin (e.g., old field, fire, 1938 hurricane). From a modeling and stand dynamics perspective, stand age (and tree age) also influences biomass/carbon growth rates. Some opportunities exist to "reset" an understocked or degraded stand. Conventional wisdom of foresters often says you would be better off starting over; it appears that can be true if regeneration yields desirable species—but it may just be a carbon/biomass response and the quality species mix for long-term growth may be sacrificed.

- The removal of tops and limbs generally has little impact on stand level carbon dynamics in Massachusetts forests. Tops and limbs that are not removed during a harvest decay quickly, generally within 10 years. If tops and limbs are a small proportion of the total harvest, then new growth will compensate for the removal within 10 years as well.

- Apart from severely understocked or degraded stands, carbon accumulation onsite in unmanaged stands will exceed onsite storage in managed stands in the long term (i.e., greater than 90 years).

- The current "business-as-usual" light harvest in the canopy increases the growth rate in the initial ten-year period, but very quickly returns to approximately the same as the unmanaged growth rate. Over time these BAU stands approach unmanaged carbon stocking but do not quite catch up after 90 years. When considered in the context of the amount of forest harvested annually in Massachusetts there is little impact of harvesting on the onsite forest carbon balance across the state.

- The scenarios we defined as "biomass" harvests (Biomass 40%, Biomass BA40, Biomass BA60) maintain high growth rates for several decades. Because of this increased growth rate, even the heavier harvested stands can reach almost 90% of the volume that could have been achieved in an unmanaged scenario. So, over a long period of time, biomass harvests have an opportunity to recover a large portion of the carbon volume removed during the harvest. However, this assumes no future harvests in the stand as well as an absence of any significant disturbance event. Both are unlikely.

- The FVS NE Variant is an effective tool to evaluate stand-level response of forest carbon to harvesting for relatively long time periods in Massachusetts. The model has known limitations but generally reflects what we know about trends in forest stand dynamics.

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CHAPTER 6

CARBON ACCOUNTING FOR FOREST BIOMASS COMBUSTION

6.1 INTRODUCTION

Greenhouse gas (GHG) emissions from bioenergy systems raise complex scientific and energy policy issues that require careful specification of an appropriate carbon accounting framework. This accounting framework should consider both the short and long term costs and benefits of using biomass instead of fossil fuels for energy generation. In most cases, the carbon emissions produced when forest biomass is burned for energy are higher than the emissions from burning fossil fuels. But over the long term, this carbon can be resequestered in growing forests. A key question for policymakers is the appropriate societal weighting of the short term costs and the longer term benefits of biomass combustion. This chapter provides analysis designed to help inform these decisions.

As discussed in Chapter 1, government policies have reflected a widely-held view that energy production from renewable biomass sources is beneficial from a GHG perspective. In its simplest form, the argument has been that because growing forests sequester carbon, then as long as areas harvested for biomass are remain forested, the carbon is reabsorbed in growing trees and consequently the net impact on GHG emissions is zero.[1] In this context, biomass combustion for energy production has often been characterized as "carbon neutral."

Assumptions of biomass carbon neutrality—the view that forest biomass combustion results in no net increase in atmospheric GHG levels—have been challenged on the grounds that such a characterization ignores differences in the *timing* of carbon releases and subsequent resequestration in growing forests (Johnson, 2008). Burning biomass for energy certainly releases carbon in the form of CO_2 to the atmosphere—in fact, as will be discussed below, per unit of useable energy biomass typically releases more CO_2 than natural gas, oil or coal. In "closed loop" bioenergy systems—for example biomass from plantations grown explicitly to fuel bioenergy facilities—energy generation will be carbon neutral or close to carbon neutral if the biomass plantation represents stored carbon that would not have been there absent the biomass plantation. Net GHG impacts of biomass from sources other than natural forests may also be carbon neutral (or close) where these materials would have quickly entered the atmosphere through decay (e.g., residue from landscaping and tree work, construction waste). But for natural forests where stocks of carbon are harvested for biomass, forest regeneration and growth will not instantaneously recapture all the carbon released as a result of using the woody material for energy generation, although carbon neutrality—resequestering all the forest biomass carbon emitted—may occur at some point in the future if the harvested land is sustainably managed going forward, for example under one of the widely recognized forest certification programs (e.g., FSC, SFI or PEFC). How long this will take for typical Massachusetts forest types and representative energy facilities, and under what conditions, is a primary focus of this study.

[1] Even when lifecycle biomass production emissions are taken into account, the argument is that net impacts on GHG, while perhaps not zero, are at least very low.

6.1.1 BRIEF REVIEW OF PREVIOUS STUDIES

The issue of net GHG benefits from burning forest biomass has been a topic of discussion since the early to mid-1990s. Beginning in 1995, Marland and Schlamadinger published a series of papers that addressed the issue, pointing out the importance of both site-specific factors and time in determining the net benefits of biomass energy (Marland and Schlamadinger, 1995; Schlamadinger and Marland, 1996a, 1996b and 1996c). This work initially was based on insights from a simple spreadsheet model, which evolved over time into the Joanneum Research GORCAM model (Marland et al., undated). A variety of other models are now available for performing similar types of bioenergy GHG analyses. These include CO_2Fix (Schellhaas et al., 2004), CBM-CFS3 (Kurz et al., 2008), and RetScreen (Natural Resources Canada, 2009). Generally these models differ in their choice of algorithms for quantifying the various carbon pools, their use of regional forest ecosystems information, and the methods used to incorporate bioenergy scenarios. Other studies have addressed these issues for specific locations using modeling approaches developed for the conditions in the region (Morris, 2008). Work on the development of appropriate models of biomass combustion carbon impacts continues to be a focus of the Task 38 initiatives of the International Energy Agency (Cowie, 2009).

In general, the scientific literature on the GHG impacts of forest biomass appears to be in agreement that impacts will depend on the specific characteristics of the site being harvested, the energy technologies under consideration, and the time frame over which the impacts are viewed (IEA, 2009). Site-specific factors that may have an important influence include ecosystem productivity, dynamics and disturbance (e.g., dead wood production and decay rates, fire, etc.); the volume of material harvested from a site for biomass; the efficiency of converting biomass to energy; and the characteristics of the fossil fuel system replaced. Recent research has also raised several other site-specific issues. Cowie (2009) cites research at Joanneum on albedo effects, which in some locations have the ability to offset some or potentially all the GHG effects of biomass combustion.[2] The effect of climate change itself on carbon flows into and out of soil and above-ground live and dead carbon pools is another factor that has yet to be routinely incorporated into biomass energy analyses.

Because of the site-specific nature of biomass GHG effects, we have developed an approach to evaluating impacts using available data on the characteristics of regional energy facilities and a forest

[2] This has generally been considered a more serious issue for harvests in forests located at higher latitudes than Massachusetts—areas where harvests interact with longer periods of snow cover to increase reflectivity.

ecosystems model that represents conditions in Massachusetts. In the next section, we discuss the overall carbon accounting framework for our analysis.

6.1.2 CARBON ACCOUNTING FRAMEWORK

Energy generation, whether from fossil fuel or biomass feedstocks, releases GHGs to the atmosphere. The GHG efficiency—the amount of lifecycle GHG emissions per unit of energy produced—varies based on both the characteristics of the fuel and the energy generation technology. However, biomass generally produces greater quantities of GHG emissions than coal, oil or natural gas. If this were not the case, then substituting biomass for fossil fuels would immediately result in lower GHG emissions. The benefits of biomass energy accrue only over time as the "excess" GHG emissions from biomass are recovered from the atmosphere by growing forests. Researchers have recently argued that the carbon accounting framework for biomass must correctly represent both the short term costs and the longer term benefits of substituting biomass for fossil fuel (Hamburg, 2010).[3]

At the most general level, the carbon accounting framework we employ is constructed around comparisons of fossil fuel scenarios with biomass scenarios producing equivalent amounts of energy. The fossil fuel scenarios are based on lifecycle emissions of GHGs, using "CO_2 equivalents" as the metric (CO_2e).[4] Total GHG emissions for the fossil scenarios include releases occurring in the production and transport of natural gas, coal or oil to the combustion facility as well as the direct stack emissions from burning these fuels for energy. Similarly, GHG emissions from biomass combustion include the stack emissions from the combustion facility and emissions from harvesting, processing and transporting the woody material to the facility. Most importantly, both the fossil fuel and biomass scenarios also include analyses of changes in carbon storage in forests through a comparison of net carbon accumulation over time on the harvested acres with the carbon storage results for an equivalent stand that has not been cut for biomass but that has been harvested for timber under a business-as-usual (BAU) scenario. Our approach includes the above- and below-ground live and dead carbon pools that researchers have identified as important contributors to forest stand carbon dynamics.[5]

[3] More broadly, climate and energy policies should consider the full range of alternative sources of energy. Energy conservation and sources such as wind, solar or nuclear have no or very low carbon emissions and may also provide additional, potentially competing, options for reducing GHGs.

[4] These adjustments incorporate the IPCC's normalization factors for methane and nitrous oxides.

[5] Typically wood products would also be included as an important carbon pools but because we assume these products are produced in the same quantities in both the BAU forest management and biomass scenarios, there will be no net change and thus there is no reason to track these explicitly. We also have not modeled soil carbon explicitly as recent papers suggest that this variable is not particularly sensitive to wood harvests (Nave et al., 2010).

The conceptual modeling framework for this study is intended to address the question of how atmospheric GHG levels will change if biomass displaces an equivalent amount of fossil fuel generation in our energy portfolio. With this objective, the modeling quantifies and compares the cumulative net annual change in atmospheric CO_2e for the fossil and biomass scenarios, considering both energy generation emissions and forest carbon sequestration. In the fossil fuel scenarios, there is an initial CO_2e emissions spike associated with energy generation—assumed here to be equivalent to the energy that would be produced by the combustion of biomass harvested from one acre—which is then followed by a drawing down over time (resequestration) of atmospheric CO_2e by an acre of forest from which no biomass is removed for energy generation. For the biomass scenario, there is a similar initial release of the carbon from burning wood harvested from an identical acre of natural forest, followed by continued future growth and sequestration of carbon in the harvested stand.

This process is summarized in the hypothetical example shown Exhibit 6-1 below. Energy emissions represent flows of carbon to the atmosphere and forest sequestration represents capture of carbon that reduces atmospheric levels. We assume the fossil fuel and biomass scenarios produce exactly the same amount of useable energy. The example is based on a fossil fuel facility that generates 10 tonnes of lifecycle C emissions and a BAU (timber cutting but no biomass removals) where total stand carbon (TSC) in all pools is rising by 0.15 tonnes per year. In the biomass scenario, lifecycle bioenergy emissions are 15 tonnes of C and TSC on the forest, which was harvested for both timber and biomass, is increasing by 0.25 tonnes of C per year, a reflection of higher rates of forest growth that can result from increases in sunlight and growing space in the more heavily harvested stand.

The bottom row of Exhibit 6-1 shows the incremental emissions from biomass energy generation (5 tonnes C) and the incremental (beyond a BAU forest management scenario) change in forest carbon sequestration (0.1 t/C/y or 1 tonne of carbon per decade). The cumulative net change (referred to hereafter as the carbon "flux") in atmospheric C is equivalent for the two feedstocks at the point in time where cumulative TSC increases, above and beyond the accumulation for the fossil fuel scenario, just offset the incremental C emissions from energy generation. In the example this occurs at year 2060 when the forest has sequestered an additional 5 tonnes of C, equivalent to the initial "excess" biomass emissions. Before that time, cumulative carbon flux is higher for the biomass scenario, while after 2060 the biomass scenario results in lower cumulative atmospheric C flux. In this comparison, not until after 2060 would the biomass energy option become better than the fossil fuel with respect to impact on GHGs in the atmosphere. Furthermore, in the example full carbon neutrality would not be achieved, assuming no change in growth rates, until five decades after 2110, at which point the entire 15 tonnes of biomass energy emissions will have been recovered in new forest growth.

Exhibit 6-1: Carbon Accounting Framework (tonnes-carbon)

Scenario	Energy Generation Emissions	Forest Stand Cumulative Total Carbon Accumulation									
Year	**2010**	**2020**	**2030**	**2040**	**2050**	**2060**	**2070**	**2080**	**2090**	**2100**	**2110**
Biomass	-15	2.5	5.0	7.5	10.0	12.5	15.0	17.5	20.0	22.5	25.0
Fossil	-10	1.5	3.0	4.5	6.0	7.5	9.0	10.5	12.0	13.5	15.0
Net Change	-5	1.0	2.0	3.0	4.0	5.0	6.0	7.0	8.0	9.0	10.0

Adoption of this conceptual framework allows a useful and potentially important reframing of the biomass carbon neutrality question. From a GHG perspective, environmental policymakers in Massachusetts might prefer biomass to fossil fuels even if biomass combustion is not fully carbon neutral—that is even if biomass burning increases carbon levels in the atmosphere for some period of time. For example, it is possible that over some policy-relevant time frame burning biomass for energy could result in cumulatively lower atmospheric CO_2e levels than generating the same amount of energy from coal, oil or natural gas—although these levels may still represent an increase in GHGs relative to today's levels. Rather than focusing all the attention on the carbon neutrality of biomass, our approach illustrates that there is a temporal component to the impacts of biomass GHG emissions to the atmosphere. The questions then become: (1) do policymakers seek to promote an energy source that could benefit the atmosphere over the long term, but that imposes increased GHG levels relative to fossil fuels in the shorter term (perhaps several decades); and (2) do the long term atmospheric benefits outweigh the short term costs?

A useful way to understand the relative carbon dynamics is to isolate the key drivers of net carbon flux. From this perspective, the incrementally greater amount of CO_2e associated with biomass energy is the relevant starting point. Following on the terminology developed by Fargione et al. (2008), we refer to these incremental emissions as the biomass "carbon debt."

In addition, we introduce the concept of "carbon dividends," which represent the longer term benefits of burning biomass. In the example in Exhibit 6-1, these dividends can be thought of as the reductions in future atmospheric carbon represented in the years after the carbon debt has been recovered (i.e., after 2060). For example, by 2100 all 5 tonnes of excess C from biomass burning have been recovered plus another 4 tonnes (the dividend) that reflects additional reductions in emissions beyond what would have resulted if only fossil fuel had been used to generate energy.

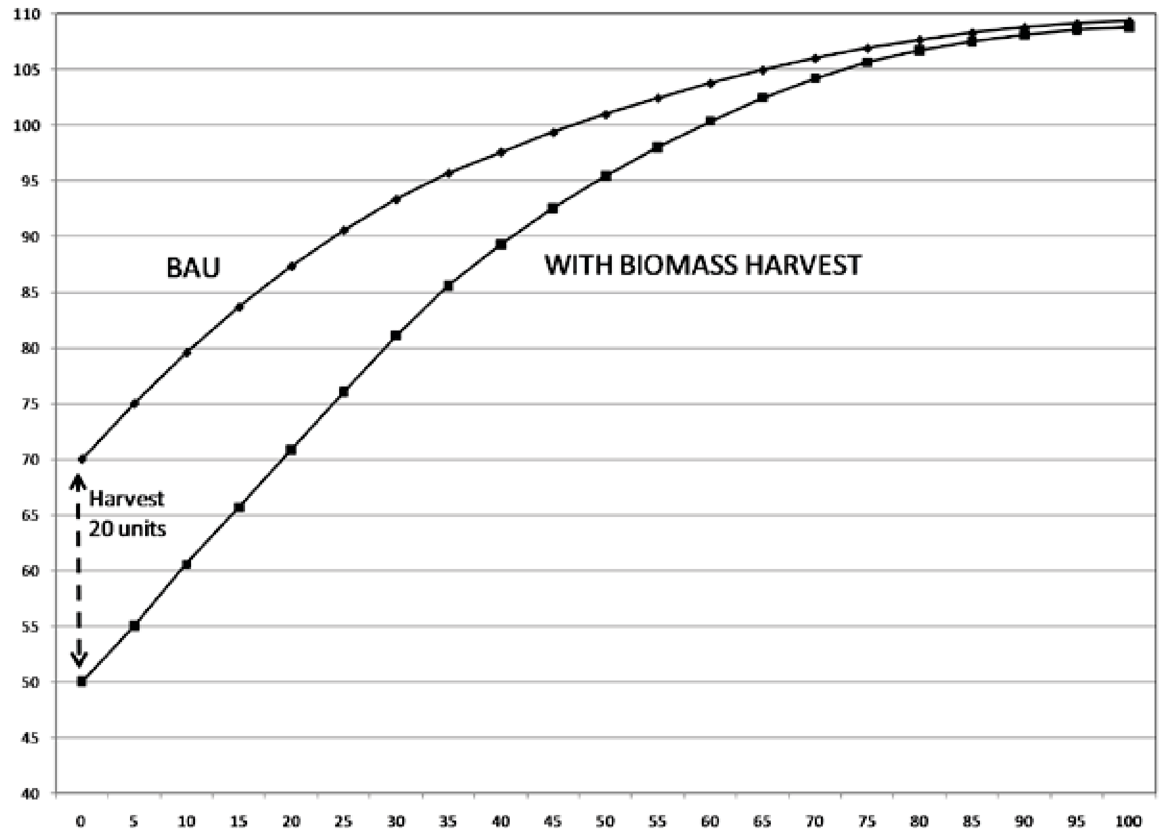
Graphically, the concepts of carbon debt and carbon dividend are illustrated in Exhibit 6-2. Exhibit 6-2a shows hypothetical carbon sequestration profiles for a stand harvested in a "business as usual" timber scenario and the same stand with a harvest that augments the BAU harvest with removal of 20 tonnes of additional carbon. Exhibit 6-2b shows the net carbon recovery profile for the biomass versus BAU harvest. This represents the incremental growth of the stand following the biomass harvest (relative to the BAU harvest) that is needed to recover the biomass carbon debt and begin accruing carbon dividends (calculated as the difference in growth between the biomass and BAU harvests). In the example, the carbon debt (9 tonnes) is shown as the difference between the total C harvested for biomass (20 tonnes) and the C released by fossil fuel burning (11 tonnes) that produces an equivalent amount of energy.

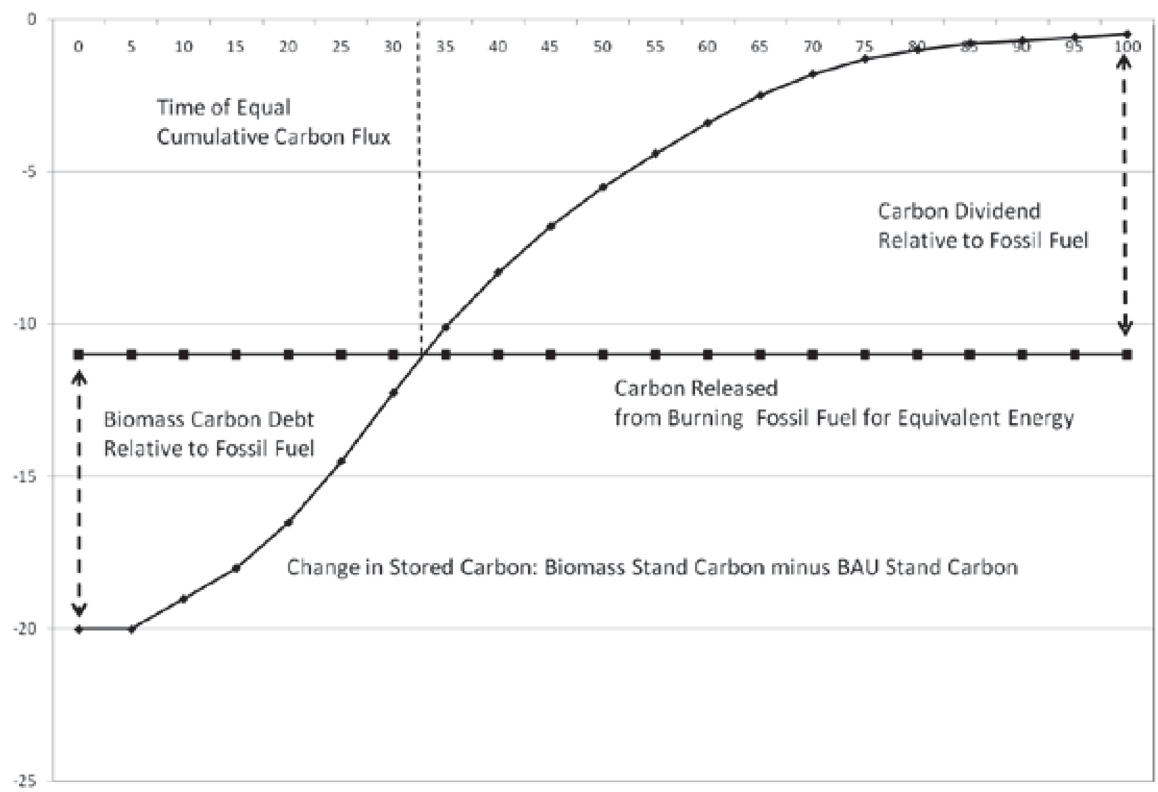
Exhibit 6-2a and 6-2b: Total Stand Carbon and Carbon Recovery Times (tonnes carbon) (see next page)

The carbon dividend is defined in the graph as the fraction of the equivalent fossil fuel emissions (11 tonnes) that are offset by forest growth at a particular point in time. In the example, after the 9 tonne biomass carbon debt is recovered by forest growth (year 32), atmospheric GHG levels fall below what they would have been had an equivalent amount of energy been generated from fossil fuels. This is the point at which the benefits of burning biomass begin to accrue, rising over time as the forest sequesters greater amounts of carbon relative to the BAU. Throughout this report we quantify these dividends as the percentage of the equivalent fossil fuel emissions that have been offset by forest growth. By approximately year 52, the regrowth of the stand has offset an additional 6 tonnes of emissions beyond what was needed to repay the carbon debt—representing an offset (or dividend) equal to 55% of the carbon that would have been emitted by burning fossil instead of biomass feedstocks.[6] In this context, a 100% carbon dividend (almost achieved in year 100 in the example) represents the time at which all 20 tonnes of emissions associated with burning biomass have been resequestered as new forest growth. In a benefit-cost analytical framework, decisionmakers would decide whether the tradeoff of higher initial atmospheric carbon levels—occurring in the period before the carbon debt is fully recovered—is an acceptable cost given the longer term benefits represented by the carbon dividends.

6 The carbon dividend, expressed as the percentage of the equivalent fossil fuel emissions offset by the growing forest, is calculated as the 6 tonnes of reduction (beyond the debt payoff point) divided by the 11 tonnes of fossil fuel equivalent that would have been needed to generate the energy produced by burning wood that released 20 tonnes of carbon.

Exhibit 6-2a and 6-2b: Total Stand Carbon and Carbon Recovery Times (tonnes carbon)





To see why carbon debt is an important driver of impacts, consider the hypothetical case where a biomass fuel's lifecycle CO_2e emissions from electricity production are one gram less per megawatt-hour (MWh) than that of coal (i.e., the carbon debt is negative). All else equal, one would prefer biomass from a GHG perspective since the emissions are initially lower per unit of energy, and this is the case even if one ignores that fact that cumulative net carbon flux to the atmosphere will fall further in the future as carbon is resequestered in regenerating forests. In the example, biomass would not be immediately carbon neutral, but would still have lower emissions than coal and would begin to accumulate carbon dividends immediately.

From an atmospheric GHG perspective, the policy question only becomes problematic when CO_2e emissions from biomass are above that of the fossil fuel alternative (i.e., where the carbon debts for biomass are positive). Because wood biomass emissions are typically higher than coal, oil and natural gas at large-scale electric, thermal or CHP facilities, this is in fact the decision policymakers face.

Framing the problem this way shifts the focus away from total emissions, allowing the net carbon flux problem to be viewed in purely incremental terms. *In our forest carbon accounting approach, the question then becomes how rapidly must the forest carbon sequestration rate increase after a biomass harvest in order to pay back the biomass carbon debt and how large are the carbon dividends that accumulate after the debt is recovered?* The debt must be paid off before atmospheric GHG levels fall below what they would have been under a fossil fuel scenario. After that point, biomass energy is yielding net GHG benefits relative to the fossil fuel scenario.

In this framework, the net flux of GHGs over time depends critically on the extent to which the biomass harvest changes the rate of biomass accumulation on the post-harvest stand. If the rate of total stand carbon accumulation, summed across all the relevant carbon pools increases very slowly, the biomass carbon debt may not be paid back for many years or even decades, delaying the time when carbon dividends begin to accumulate. Alternatively, for some stands, and especially for slow-growing older stands, harvesting would be expected to increase the carbon accumulation rate (at least after the site recovers from the initial effects of the harvest) and lead to relatively more rapid increases in carbon dividends. Determining the time path for paying off the carbon debts and accumulating carbon dividends is a principle focus of our modeling approach.

In this context, it is also important to note that the point at which the cumulative carbon flux from biomass just equals the cumulative flux from fossil fuels (the point at which the biomass carbon debt is paid off) is not necessarily the point at which a policymaker is indifferent between the biomass and fossil fuel scenarios. For example, the policymaker might only be indifferent at the time when the discounted damages resulting from the excess biomass emissions just equals zero—this is the point in time at which early damages due to increased GHG levels from biomass are just offset by lower biomass damages in later years when net cumulative GHG flux from biomass is below that of the fossil fuel alternative. In this case, longer time periods are needed to reach the point defined as "fully-offset damages." The higher the discount rate—indicative of a greater preference for lower GHG levels in the near-term, the longer the time to reach the point of fully-offset damages.

6.1.3 OTHER CONSIDERATIONS: LANDSCAPE OR STAND-LEVEL MODELING

A key question in developing the conceptual framework for biomass GHG analysis is whether to analyze the problem at the level of the individual stand or across the entire landscape affected by biomass harvests. A recent formulation of the biomass carbon neutrality argument focuses on the forested landscape across the entire wood supply zone for a biomass plant—as opposed to individual harvested stands—and suggests that as long as landscape-scale forest growth is in excess of harvests, then biomass is embedded in the natural carbon cycle of the forests and is causing no net increase in GHG emissions (Miner, 2010). In our view, however, this landscape approach to carbon neutrality is incomplete because it does not fully frame the issue with respect to the carbon sequestration attributes of the forested landscape in a "business as usual" scenario. In general, the carbon accounting model should be premised on some knowledge of how lands will be managed in the future absent biomass harvests, and this becomes a critical reference point for analyzing whether burning biomass for energy results in increased or decreased cumulative GHG emissions over time.

Consequently, appropriate characterization of the BAU baseline is essential to the development of an accurate carbon accounting model of forest biomass combustion. In the case of the landscape argument for carbon neutrality, the conclusion that biomass burning has no net impact on GHG emissions does not account for the fact that in the absence of biomass harvests, the forests would likely have continued to sequester carbon anyway.[7] Therefore, a well-framed landscape analysis needs to consider the net carbon emissions of biomass burning relative to the BAU scenario of continued carbon accumulation by forests across the landscape. Framing the problem this way does not necessarily negate the landscape carbon neutrality argument—it simply recognizes that the landscape level carbon accounting problem is a more complicated one. However, when a complete representation of the baseline is taken into account, the landscape-scale and the

[7] This assumes that additional biomass stumpage revenues will not dramatically alter the acreage devoted to commercial forestry activities. We believe this is a reasonable assumption given the current low prices for biomass stumpage. At $1 to $2 per green ton, few, if any, landowners would see enough change in revenue from biomass sales to alter their decisions about whether to keep forest land or sell it to someone who is looking to change the land use (e.g., a developer). As a result, we do not address the carbon issues associated with conversion of natural forests to energy plantations. We also do not address "leakage" issues that might arise if productive agricultural land is converted to energy plantations and this leads to clearing forests somewhere else to create new cropland.

stand-level frameworks may yield the same result. The following simplified numerical example provides an illustration of why this is the case.

The example assumes an integrated energy/forest system made up of three carbon pools—the forest, atmosphere, and fossil fuel pools—each initially containing 1000 tonnes of carbon. In addition, we assume burning biomass releases 50 percent more emissions than burning fossil fuels for an equivalent level of energy production—close to the estimate of carbon debts when comparing biomass and coal-fired electricity generation. Finally, we specify that an average forest's total stand carbon across the above- and below-ground carbon pools increases by 5% per year, or 50 tonnes in our example.

In year one of a coal-fired electric scenario, we assume energy production at a level that transfers 10 units of carbon from the fossil fuel pool to the atmosphere. In the same year, the forest removes 50 tonnes of carbon from the atmosphere. The net values for each pool after one year are:

- **Fossil Fuel Carbon Pool:** 990 tonnes (1000 tonnes–10 tonnes released from energy production)
- **Forest Carbon Pool:** 1050 tonnes (1000 tonnes + 50 tonnes forest sequestration)
- **Atmospheric Carbon Pool:** 960 tonnes (1000 tonnes+ 10 tonnes emissions–50 tonnes forest sequestration).

Alternatively, we consider a change in energy production that replaces fossil fuel with biomass, in this case releasing 15 tonnes of carbon versus 10 tonnes in the equivalent energy fossil scenario. We also assume that cutting the forest does not reduce total carbon sequestration (i.e., that the harvested areas of the forest still add carbon at the 5 percent rate).[8] At the end of the first year, the carbon pools are as follows:

- **Fossil Fuel Carbon Pool:** 1000 tonnes (no change)
- **Forest Carbon Pool:** 1035 tonnes (1000 tonnes–15 tonnes biomass + 50 tonnes forest sequestration)
- **Atmospheric Carbon Pool:** 965 tonnes (1000 tonnes + 15 tonnes emissions–50 tonnes forest sequestration).

In the example, it is true that forest growth across the landscape exceeds the amount of biomass harvested (50 tonnes of new sequestration versus 15 tonnes of biomass removals)—the condition under which advocates of landscape-level carbon neutrality would argue that biomass burning is embedded in a natural cycle in which forest sequestration (50 t-C/y) exceeds removals for biomass (15 t-C/y). But it is also true that the initial effect of switching to biomass is to increase atmospheric carbon levels, in this case by 5 tonnes. The result makes clear that when the BAU baseline is correctly specified, the net change in GHG from biomass is equivalent to the biomass carbon debt, and therefore that carbon neutrality is not achieved immediately.

Introducing the assumption that additional stands are harvested in subsequent years to provide fuel for a biomass plant—while adding greater complexity to the analysis—does not alter the basic conclusions as long as stands are harvested randomly (e.g., stands with rapid carbon recovery rates are no more or less likely to be harvested than stands with slower carbon recovery). For each additional year of harvests, a carbon debt is incurred and these are additive over time. Similarly, the period required to pay off the debt is extended one year into the future for each additional year of harvests. Finally, the longer-term dividends are also additive and will accumulate over time as greater quantities of fossil fuel emissions are offset by forest growth.

The one area where landscape scale analysis might alter conclusions about carbon debts and dividends is a situation where the stands with more rapid carbon recovery profiles can be scheduled for harvest sooner than the slower recovery stands. This has the potential to accelerate the time to debt payoff and the onset of the carbon dividends. To implement such an approach, one would need to be able to identify the characteristics of the rapid carbon recovery stands and be able to influence the scheduling of harvests across the landscape. Detailed analysis to clearly identify rapid recovery stands is beyond the scope of the analysis in this report. Nonetheless, we would like to note that, while harvest scheduling may be possible for large industrial forest ownerships, it would be difficult to accomplish across a landscape like Massachusetts that is fragmented into many small ownerships. For this report, we have confined our focus to stand level analyses, which should provide useful indicators of the timing and magnitude of carbon debts and dividends in Massachusetts.

6.2 TECHNOLOGY SCENARIOS AND MODELING ASSUMPTIONS

6.2.1 OVERVIEW OF TECHNOLOGIES AND APPROACH

To illustrate the relative carbon life-cycle impacts associated with various energy scenarios, we compare the emission profiles for a representative set of biomass energy generation facilities relative to their appropriate fossil fuel baselines. Our analysis considers the following technologies:

- **Utility-Scale Electric:** A utility-scale biomass electric plant (50 MW) compared to a large electric power plant burning coal or natural gas.
- **Thermal Chips:** A thermal generation facility relying on green biomass chips relative to a comparable facility burning fuel oil (#2 or #6) or natural gas.

[8] This is likely a conservative scenario for the first year after harvest when the stand is recovering from the impacts of the cut. Assuming a lower than 5% rate of carbon growth on these acres would lower the overall average across the landscape to below 5%; the assumptions made above therefore may overstate the amount of carbon in the forest pool and understate the carbon in the atmosphere.

- **Thermal Pellets:** A thermal generation facility relying on wood pellets relative to a comparable facility burning fuel oil or natural gas.
- **CHP:** A combined heat and power (CHP) facility compared to a similar facility burning oil or natural gas.

We selected these scenarios to illustrate the range of likely wood-based bioenergy futures that we judge to be feasible in the short- to mid-term in Massachusetts. This choice of technologies reflects differences in scale, efficiency and fuel choice. The emission profiles of more advanced technologies—such as cellulosic ethanol production and biomass pyrolysis—are not modeled based on lack of commercial demonstrations, scale requirements that make development in Massachusetts unlikely, or because of a lack of available GHG emissions data.

As detailed in our conceptual framework, each scenario is made up of two primary components: a stand-level forest carbon model and an energy facility GHG emissions model. In the fossil fuel scenarios, we assume the stand is harvested for timber but not for biomass. We then track the total amount of C in the stand's various carbon pools—including above- and below-ground live and dead wood—over a 90-year time frame. For the biomass scenarios, consistent with the supply analysis discussed in Chapter 3, we assume a heavier harvest that removes additional material in the form of logging residues and low-quality trees. For each scenario, we then model the change in total stand carbon over the same 90-year time frame in order to provide comparisons of net changes in total stand-level carbon relative to the baseline "no biomass" scenario. The energy facility emissions model is designed to take into account both the direct stack emissions of energy generation as well as the indirect emissions that come from producing, processing and transporting fuels to the facility. These are expressed as (1) biomass carbon debts, which denote the incremental percentage of carbon emissions due to harvesting and combusting wood relative to the lifecycle GHG emissions of the alternative fossil fuel, and (2) biomass carbon dividends which are the longer term benefits from reducing GHGs below fossil baseline levels. For each scenario, the combined forest and energy carbon models provide an appropriate accounting for the emissions from energy production and the carbon sequestration behavior of a forest stand that has been harvested (1) only for timber or (2) for both timber and biomass.

The details of the forest harvest scenarios are described below, followed by a discussion of the GHG modeling process for energy facilities.

6.2.2 FOREST HARVEST SCENARIOS

We take the individual stand as the basis for our carbon accounting process. For the fossil fuel baseline scenarios, we assume a "business as usual" forest management approach where the stand is harvested for timber but not for biomass. The model provides a dynamic baseline for comparisons with the biomass alternative. The scenarios are summarized in Exhibit 6-3 below and include two alternative BAU specifications, one a relatively heavy cut that removes approximately 32% of the above-ground live biomass, and a lighter BAU that removes 20%. The heavier BAU is intended to represent the case where the landowners who decide to harvest biomass are the ones who cut more heavily in the BAU. The lighter harvest BAU represents a scenario where the distribution of landowners harvesting biomass is spread more evenly across the full range of landowners who currently harvest timber, as specified in the Massachusetts Forest Cutting Plan data discussed in Chapter 3. We assume in the BAU that all logging residues are left in the forest.

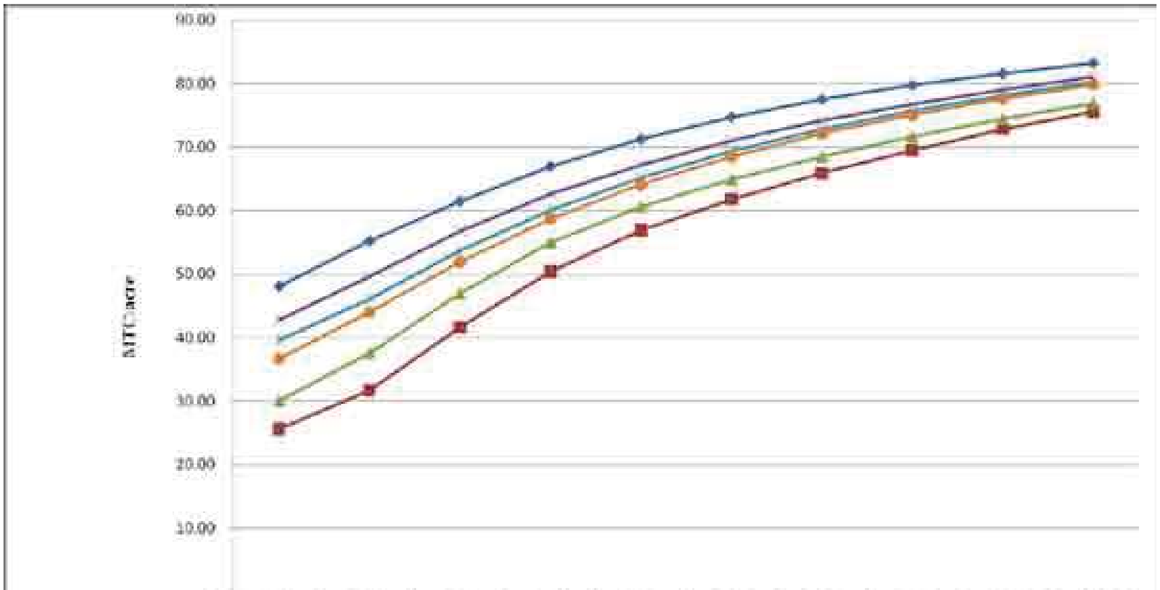
Using the FVS model, described in Chapter 5, we quantify changes in total stand carbon by decade through an evaluation of carbon in the above- and below-ground live and dead carbon pools for the following six biomass harvest scenarios. Carbon recovery profiles represent averages for a set of 88 plots in the Massachusetts FIA database with an initial volume of more than 25 tonnes of carbon per acre in the above-ground live pool.

Exhibit 6-3: BAU and Biomass Harvest Scenarios

Harvest Category	Description	Carbon Removed (tonnes)	Above-Ground Live Carbon Harvested (%)	Logging Residues Left On-Site (%)
BAU 20%	Lighter BAU removal	6.3	20	100
BAU 32%	Heavier BAU removal	10.2	32	100
Biomass BA60	Moderate biomass removal: BAU & Biomass removal down to 60 ft2 of stand basal area	19.3	60	35
Biomass 40%	Lighter biomass removal: BAU plus biomass removal equals 40% stand carbon	12.0	38	35
Biomass BA40	Heavier biomass removal: BAU & Biomass removal down to 40 ft2 of stand basal area	24.3	76	35

The results of the FVS analysis provide profiles of total stand carbon and above-ground live carbon over time for the BAU and biomass harvest scenarios. These are graphed on the next page in Exhibits 6-4 and 6-5.

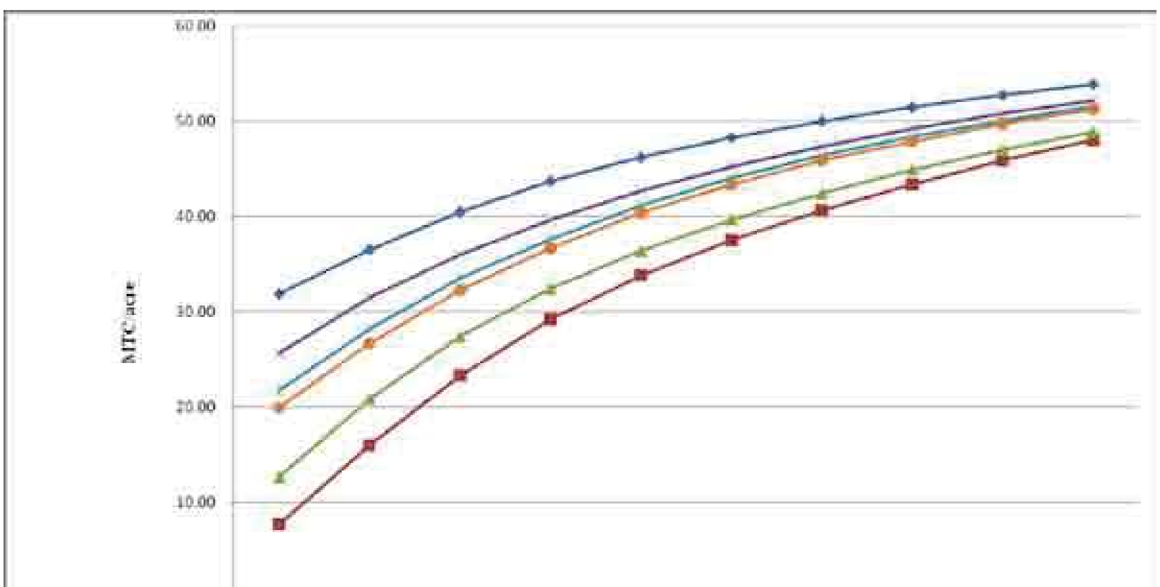
Exhibit 6-4: Total Stand Carbon



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Unmanaged	48.09	55.21	61.50	66.97	71.27	74.78	77.52	79.77	81.60	83.27
Biomass BA40	25.61	31.74	41.63	50.44	56.91	61.78	65.90	69.51	72.86	75.60
Biomass BA60	30.01	37.58	46.97	55.00	60.62	64.92	68.47	71.63	74.42	76.89
BAU 20%	42.81	49.66	56.82	62.69	67.25	71.05	74.18	76.81	79.05	81.00
BAU 32%	39.62	46.12	53.68	60.08	65.18	69.43	72.84	75.74	78.11	80.27
Biomass 40%	36.67	44.00	51.94	58.74	64.10	68.50	72.17	75.12	77.64	79.87

Year - MTC/acre (Total Stand Carbon)

Exhibit 6-5: Above-Ground Live Stand Carbon



	2010	2020	2030	2040	2050	2060	2070	2080	2090	2100
Unmanaged	31.91	36.53	40.49	43.70	46.22	48.30	50.02	51.47	52.75	53.90
Biomass BA40	7.66	15.93	23.29	29.22	33.81	37.53	40.66	43.37	45.92	48.01
Biomass BA60	12.62	20.84	27.40	32.40	36.38	39.67	42.42	44.91	47.01	48.87
BAU 20%	25.64	31.47	35.98	39.64	42.71	45.27	47.38	49.20	50.80	52.22
BAU 32%	21.76	28.22	33.49	37.63	41.15	44.01	46.38	48.34	50.05	51.56
Biomass 40%	19.93	26.67	32.27	36.68	40.39	43.36	45.88	47.90	49.71	51.26

Year - MTC/acre (Aboveground Live Carbon)

Due to model constraints, the FVS analyses rely on "thin-from-above" harvest strategies to simulate both BAU and biomass harvests, although we conducted some limited analysis of the sensitivity of the results to alternative assumptions. For all the biomass harvests, we assume 65% of the logging residues are removed from the forest, with the remainder left on the ground.

The results were analyzed to determine how the stands harvested for biomass responded relative to their response in the BAU scenario. This analysis is designed to show relative rates of recovery of forest carbon stocks following biomass harvests.

6.2.3 BIOMASS AND FOSSIL FUEL GHG EMISSIONS

To estimate biomass carbon debts relative to fossil fuel technologies, we assembled estimates of GHG emissions per unit of energy produced by each technology. These estimates included both the direct combustion emissions as well as the indirect emissions related to feedstock production, processing and transportation. To the extent that data were available, we work in CO_2 equivalents (CO_2e), a metric that considers other greenhouse gases (e.g., methane from coal mines) and expresses them in terms of the amount of CO_2 that would have an equivalent global warming effect. The emissions estimates for both the biomass and fossil fuel technologies are shown below in Exhibit 6-6, where they have been converted to kilograms of carbon per energy unit.

Exhibit 6-6: Carbon Emission Factors by Technology* Kilograms per Unit of Energy**

Scenarios	Biomass	Coal	Oil (#6)	Oil (#2)	Natural Gas
Utility-Scale Electric	**Kilograms/MWh**				
Fuel Prod & Transport	7	9			34
Fuel Combustion	399	270			102
Total	406	279			136
Thermal	**Kilograms/MMBtu**				
Fuel Prod & Transport	1		6	6	6
Fuel Combustion	35		27	25	17
Total	36		33	31	23
CHP	**Kilograms/MMBtu**				
Fuel Prod & Transport	1		7	6	6
Fuel Combustion	35		29	27	18
Total	36		35	33	24

** As discussed below, emissions factors for pellets are characterized relative to the thermal technology using green chips which is shown in this table.*

*** Sources and calculations for these data are described in the text.*

Emissions from Biomass Harvest, Processing and Transportation: For the biomass technologies, we include estimates of the CO_2e releases associated with harvesting, processing and transporting the biomass fuel to a bioenergy facility. For green chips (delivered to a large-scale electric, thermal or pellet facility), the estimates are based on releases of CO_2 associated with diesel fuel consumption in each of these processes. We estimated harvest and chipping costs using the U.S. Forest Service's Fuel Reduction Cost Simulator (also used to estimate harvesting costs for the wood supply analysis and described in Chapter 3). We assumed chips were transported 100–120 miles (round-trip) to the combustion facility, using trucks carrying 25–30 green tonnes with an average fuel efficiency of 5 mpg. Our results were verified for consistency with other relevant studies including: CORRIM (2004); Department of Forest Resources, University of Minnesota (2008); Finkral and Evans (2008); and Katers and Kaurich (2006).

Indirect CO_2e emissions make a very small contribution to the overall life-cycle emissions from biomass energy production, generally on the order of 2%. A simple way to understand this is as follows. Diesel consumption in harvesting and processing forest biomass is typically less than one gallon (we have calculated an average of 0.75 gallons per green ton based on the sources described above). Diesel consumption in transport is also assumed to be less than one gallon (we have calculated 0.85 gallons per green ton). The combustion of a gallon of diesel releases 22 pounds of CO_2, while the combustion of a ton of green wood (45% moisture) releases one ton of CO_2 [9]; thus, CO_2 emissions per gallon of diesel are equivalent to about 1% of stack emissions. The amount of carbon dioxide released per MWh or per MMBtu will of course depend on the green tonnes of wood required, but the ratio between indirect CO_2e emissions and combustion emissions will remain the same.

Lifecycle Emissions from Utility-Scale Electric: For these facilities, all emissions are initially calculated as CO_2e /MWh of electrical output, and then expressed as C/MWh. The biomass estimate is based on analysis of electricity generation and wood consumption from a set of power plants in this region with efficiencies in the 20% to 25% range. These data have been compiled from a combination of information from company websites and financial reports. On average, these plants release about 1.46 tonnes of CO_2 (399 kg of C) per MWh. When combined with the indirect emissions discussed above, lifecycle CO_2e for biomass plants total approximately 1.49 tonnes per MWh (or 406 kg of C).

The comparable data for natural gas and coal have been developed by NREL (Spath and Mann, 2000 and Spath et al., 1999) and include the full lifecycle CO_2e emissions. On a per MWh basis,

9 A bone-dry ton of wood is assumed to be 50% carbon. A green ton of wood with 45% moisture weighs 1.82 tons. Thus, the ratio of green wood (45% moisture) to its carbon content is 3.64 (or 1.82 / 0.5). This is essentially the same as the ratio of a ton of carbon dioxide to its carbon content (3.67, equal to the ratio of the molecular weight of CO2 to C, or 44/12). So, the combustion of one green ton of wood releases one ton of CO2.

lifecycle CO_2e emissions for a large (505 MW) combined-cycle natural gas power plant are approximately 0.5 tonnes (136 kg of C) per MWh, of which 75 percent results from the combustion facility itself and 25 percent is from gas production and transportation. The comparable lifecycle estimate for a large coal generating station is approximately 1.0 tonne (279 kg of C) per MWh, with 97 percent of the emissions attributable to the generating station emissions and the remainder to mining and transportation of the coal. The natural gas plant was assumed to be very efficient at 48% due to the combined-cycle technology, while the coal plant was closer to average efficiency at 32%. These plants were selected to bracket the range of emissions of fossil fuel plants relative to their biomass electric counterparts.

We note that co-firing of biomass with coal represents another technology variant for electric utilities. The emissions characteristics of co-firing biomass with coal are expected to similar to those from a stand-alone utility scale biomass electricity plant since the biomass combustion efficiency will be similar in both types of operations. As long as this is the case, the results for utility-scale biomass electricity are indicative of the emissions characteristics of biomass emissions at electricity plants using co-firing.

Lifecycle Emissions from Thermal Facilities: All emissions for these facilities are expressed as C/MMBtu of thermal output. Biomass is based on a typical thermal plant with 50 MMBtu's per hour of capacity and 75% efficiency, which has heat input of 120,000 MMBtu/yr (see Chapter 2 for a more detailed description of this pathway and technology). Assuming the gross heating value of oven-dry wood to be 8,500 Btu's/lb, the total lifecycle estimate for carbon emissions is 36 kg/MMBtu.

Emissions data for heating oil and natural gas thermal plants were developed assuming that the typical capacity of the plants was also 50 MMBTH (these technologies and pathways are described in Chapter 2). The oil facilities were assumed to run at 80% efficiency, while the natural gas plants were assumed to be more efficient at 85%. We consider oil facilities that use distillate fuel oil (#2 or #4) and residual fuel oil (#6). The majority of the commercial and industrial facilities in Massachusetts use distillate oil (about 70%), but it is possible that wood biomass may compete more directly with plants burning residual fuel oil. For natural gas, indirect emissions were calculated using the same percentages available in the NREL analysis of electric power plants. Indirect emissions from oil are based on estimates from the National Energy Technology Laboratory (Gerdes, 2009). Lifecycle carbon emissions were calculated to be 33 kg/MMBtu for #6 fuel oil, 31 kg/MMBtu for #2 fuel oil, and 23 kg/MMBtu for natural gas. Because of the differences in relative combustion efficiencies, the gap between biomass and fossil fuel technologies for thermal facilities is smaller than the gap for utility-scaled electric facilities.

Lifecycle Emissions from Pellet Applications: Emissions for thermal pellet applications require the addition of emissions from plant operations and for transport and distribution of pellets from the plant to the final consumer. The limited analysis that we have seen for these operations (for example, Katers and Kaurich, 2006) suggest that the increased efficiencies in boiler combustion achieved with pellets approximately offsets most of the increased emissions from plant operations and additional transport of pellets from the plant to their final destination.

Lifecycle Emissions from CHP Facilities: Emissions for CHP facilities are also expressed on the basis of MMBtu of heat output, in which electrical energy is converted to a Btu equivalent. The analysis of these operations depends critically on the mix of thermal and electrical output in the plant design. In general, thermal-led facilities tend to relative emissions profiles that are similar to their thermal counterparts, while electric-led facilities more closely resemble the emissions profiles of electric power plants. While some variations can result from the scale of facilities, the specifics of the design, and the type of heat recovery systems employed, the utility-scale electric and dedicated thermal technologies provide approximate bounds for the wide range of possibilities for CHP facilities.

Carbon Debt Summary: Exhibit 6-7 below summarizes the carbon debts for biomass relative to each technology and fuel. These are expressed as the percentage of total biomass-related emissions accounted for by the incremental GHG releases from biomass relative to a specific fossil fuel and technology combination. For example, using the data from Exhibit 6-6, we calculate the 31% for coal electric as ((406–279)/406)*100.

Exhibit 6-7: Carbon Debt Summary Table*
(Excess Biomass Emissions as % of Total Biomass Emissions)

Scenarios	Coal	Oil (#6)	Oil (#2)	Natural Gas
Electric	31%			66%
Thermal		8%	15%	37%
CHP		2%	9%	33%

** See text for pellet applications.*

It is clear from this table that carbon debt depends on both the choice of fuel (and hence its heating value) and the choice of technology. Carbon debt for biomass compared to natural gas in electric power is much higher than the carbon debt in the thermal scenario. These differences are attributable to the relative efficiencies of the technologies in each scenario—natural gas electric power has a large advantage in this case due to the assumed use of combined-cycle technology.

Carbon debts for CHP raise another important issue when comparing biomass fuel with other technological alternatives. While comparisons of biomass CHP and CHP using oil or natural gas may be straightforward, there are no data on how much fossil-fuel based CHP capacity is now operating in Massachusetts and could potentially be a candidate for replacement. Nevertheless, this comparison may still be useful in assessing the relative carbon merits of constructing a new biomass CHP plant or a new fossil fuel-fired CHP plant. On the other hand, it is interesting to note that if biomass CHP facilities were developed, it is likely that they would replace a mix of independent thermal and electric applications. Since a large amount of heat is wasted in producing stand-alone electricity, these comparisons may show biomass CHP with no carbon debt at the outset. For example, if thermal-led biomass CHP at a commercial location replaces a current mix of heat from oil and power from coal, then total carbon emissions generated at the new site are likely to decline relative to the fossil scenario as long as a significant percentage of the waste heat is utilized. In contrast, if natural gas is consumed in the current energy mix, the situation may be reversed.

6.3 FOREST BIOMASS CARBON ACCOUNTING RESULTS

6.3.1 INTRODUCTION

As discussed in the conceptual framework section, our carbon accounting analysis for biomass focuses on biomass carbon debt, biomass carbon dividends and the number of years until debts are paid off and dividends begin accumulating. These are a function of the bioenergy technology as well as the biophysical characteristics of the forest and management practices used. The transition from debt to dividend occurs at the point when the atmospheric carbon level resulting from the lifecycle biomass emissions falls to the point where it just equals the level resulting from lifecycle fossil fuel emissions.[10]

To examine the carbon debts, dividends and the timing of the transition from one to the other, we analyzed a wide array of integrated energy technology/forest management scenarios. These consider the impacts of potential differences in (1) energy technology and efficiency and (2) the biophysical characteristics of the forest and assumptions about the intensity and type of silvicultural approach used for harvests in both the BAU and biomass scenarios.

Our analysis approaches the problem by establishing integrated technology and forest scenarios that we find to be representative of average or typical conditions and management practices. Energy technologies are characterized in terms of typical lifecycle carbon emissions. Representative forest carbon recovery paths are estimated using FVS model simulations averaged across 88 actual forest stands that are included in the U.S. Forest Service's system of FIA sampling plots in Massachusetts. Overall these analyses provide guidance on the range of *average* forest carbon recovery times for each technology. It is important to note, however, that care should be exercised when translating these average results into policy. Our concern is primarily the result of three factors. First, energy technologies are continually evolving and the characteristics of any specific project proposal could differ from the typical existing configurations that we have analyzed. Second, our lack of knowledge of how stands will be harvested in response to increased demand for forest biomass may introduce substantial uncertainty in the projections of forest carbon recovery rates. Third, modeling the carbon dynamics of forest stands is complex, and although our analysis provides indications of broad general trends, these are subject to considerable uncertainty about stand-level changes in carbon pools.

In the remainder of this chapter, the presentation of results in organized around three principal topics:

- How do choices about biomass technology and assumptions about the fossil fuel it will replace affect carbon recovery times?
- How do forest management choices with respect to harvest intensity and silvicultural practice interact with the biophysical properties of forests to determine carbon recovery profiles?
- What are the carbon dividend levels associated with the various biomass energy scenarios?

To answer these questions, we first present data from our modeling of the various energy/forest scenarios. We then summarize our overall conclusions and discuss some considerations regarding how our results are most appropriately interpreted and used in energy and environmental policymaking processes.

6.3.2 ENERGY TECHNOLOGY AND CARBON DEBT RECOVERY

A key insight from our research is the wide variability in the magnitude of carbon debts across different biomass technologies. This results from the way specific lifecycle GHG characteristics of a bioenergy technology combine with the GHG characteristics of the fossil fuel energy plant it replaces to determine carbon debts. As shown in Exhibit 6-7, carbon debts for situations where biomass thermal replaces oil-fired thermal capacity can be as low as 8%, whereas the debt when biomass replaces combined-cycle natural gas in large-scale electricity generation can range as high as 66%.

Exhibit 6-8 illustrates how debt payoff varies with technology, with detailed supporting numbers included in the table in Exhibit 6-9. The scenario represented in this exhibit is one that assumes a relatively heavy BAU harvest of timber—32% removal of above-ground live carbon using a diameter limit partial harvest—and a biomass harvest that extends the diameter limit approach to removal of all trees down to a residual basal area of 60 ft^2 per acre. Exhibit 6-8(a) illustrates the FVS model results for total stand carbon in stands harvested only for timber (BAU) and for the same stands where the BAU harvest is augmented by the additional removals of biomass including the harvest of 65% of all tops and limbs. Exhibit 6-8(b) captures the relative differences in growth between the two stands, indicating an initial harvest of 38 green tons of biomass.[11] For these scenarios,

[10] Offsetting of earlier damages from higher biomass GHG levels would require additional years of lower GHG levels (or dividends) in the biomass scenario. Full carbon neutrality would not be achieved until the point at which the entire release of carbon from burning biomass has been resequestered in the forest carbon pools.

[11] This relative difference in growth is derived by subtracting the BAU recovery curve from the biomass harvest recovery curve in Exhibit 6-8(a) In this case, the relationship in Exhibit 6-8(b) can be interpreted as the incremental growth in the stand harvested for biomass relative to growth of the BAU stand. Only through this incremental growth will carbon debts be recovered.

the graph shows that post-harvest biomass stands sequester carbon more rapidly than BAU stands harvested only for timber. In this scenario, the biomass harvest removed an additional 9.1 tonnes of above-ground live carbon from the stand (and resulted in the loss of another 0.5 tonnes of below ground carbon). After one decade of growth, the total carbon in the biomass stand has increased by approximately 1.1 tonnes compared to the BAU stand and continues to increase to a cumulative total 6.2 tonnes of carbon after 90 years. At this point in time, the biomass stand has recovered approximately 65% of the carbon removed from the stand and used for biomass energy generation (6.2 tonnes versus 9.6 tonnes harvested).

Exhibit 6-8(a): Forest TSC Sequestration Rates under Scenario 1 (tonnes carbon)

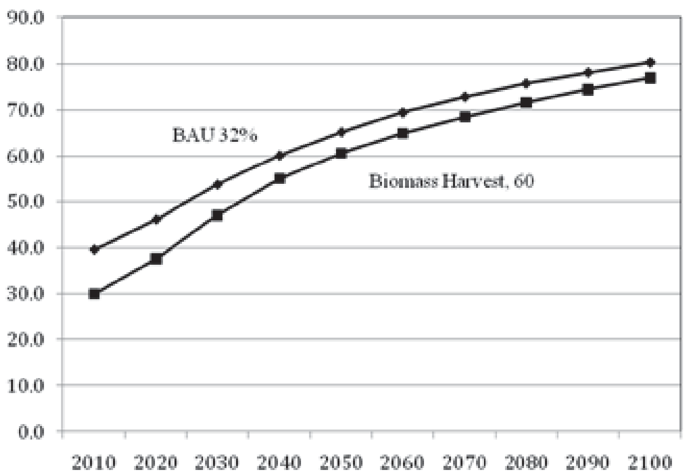
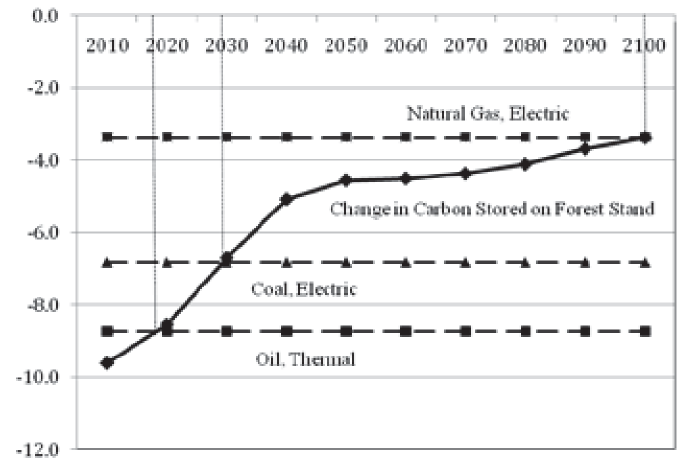


Exhibit 6-8(b) also indicates the time required on average for the stands to recover the carbon debt for various technologies. Oil-fired thermal facilities are represented by the horizontal line indicating that for the equivalent level of energy production they emitted about 12% less carbon than a thermal biomass plant when full lifecycle carbon emissions are taken into account.[12] The intersection of the thermal-oil emissions line and the forest carbon recovery curve identifies the year in which the carbon debt is fully recovered in this scenario—about 10 years for replacement of oil-fired thermal capacity with biomass. At that time, the net atmospheric levels of GHGs are equivalent for the biomass and fossil fuel technologies. Prior to that point, biomass resulted in higher GHG levels, but in later years biomass GHG levels are lower than those for fossil fuels because the forest continues to remove relatively greater amounts of the carbon than the stand in the BAU scenario. These are the benefits we characterize as carbon dividends.

12 This represents an average of residual fuel oil (#6) and distillate fuel oil (#2).

Exhibit 6-8(b): Carbon Recovery Rates under Scenario 1 (tonnes carbon)



The carbon debt recovery periods are also plotted in Exhibit 6-8(b) for biomass replacement of coal and natural gas electricity generation. The results make clear that technologies with higher carbon debts have longer payoff times, indicative of carbon dividends that do not appear until further in the future. Technology scenarios with shorter payoff times have lower GHG impacts than scenarios with higher carbon debts. In general, the analysis indicates that thermal carbon debts can be substantially lower than debts from large-scale electricity generation.

Our analyses also considered the carbon debt characteristics of wood pellet technology and CHP systems. In general, we find that carbon debts associated with burning pellets in thermal applications do not differ significantly from debts resulting from use of green wood chips. The differences relate primarily to location of GHG emissions associated with water evaporation from green wood rather than the overall magnitude of the lifecycle GHG emissions. For CHP, carbon debts generally fall somewhere between those of thermal and large-scale electric, depending upon whether the CHP plant is designed to optimize thermal or electric output; however, in our cases, initial carbon debts are shown to be lower than thermal because all waste heat is fully utilized and some reductions in the gross efficiency of oil and gas are recognized due to higher electrical efficiencies.

The technology scenario rankings described above generally hold true as long as the forest management and silvicultural practices are the same for the various energy generation technologies (however, as demonstrated below in Section 6.3.3.4, this may not be the case if harvesting methods preclude the removal and use of tops and limbs). Within this general hierarchy, however, the absolute and relative timing of carbon recovery for the different technologies will vary depending on the specific harvesting assumptions and results from the forest modeling process (discussed in detail in Section 6.3.3 below).

In interpreting the technology/carbon debt results, it is important to recognize that the carbon debts discussed above are based on average levels of GHG emissions per unit of energy for typical energy generation systems readily available today.[13] Biomass energy technology, however, is evolving and there are technologies that have yet to be commercialized in the U.S. that are more efficient and thus produce less GHG emissions per unit of useable energy—for example the biomass CHP gasification technologies discussed in Chapter 2. Bioenergy proposals based on new technologies with lower carbon debts are feasible and have the potential to reduce GHG impacts and associated carbon debts.

6.3.3 FOREST MANAGEMENT AND CARBON RECOVERY

Within the broad context of biomass carbon debts and dividends for specific technologies, the timing of carbon recovery is a direct function of two factors related to forests and forest management—(1) the biophysical characteristics of Massachusetts forests and (2) assumptions about the intensity and type of silvicultural approach used for harvests in both the BAU and biomass harvest scenarios.

As described in Chapter 5, we rely on FIA data for basic biophysical information about Massachusetts forests, and we evaluate carbon dynamics using the U.S. Forest Service FVS model. The FIA data are intended to provide a set of forest stands that is representative of the range of forest cover types, tree size distributions, species growth characteristics, and per-acre wood inventories across Massachusetts. For presentation and analysis purposes we generally characterize our results as carbon recovery rates averaged across the 88 stands in our FIA database that are at a stage in their development that makes them available for biomass harvests (i.e., stands with greater than 25 tonnes of carbon in the above-ground live carbon pool). This approach provides a reasonable basis for capturing the impact on carbon debt recovery of differences in the biophysical characteristics of the forests.

Assumptions about the nature of forest management in both the BAU and biomass harvest scenarios also have important impacts on the timing of the transition from carbon debt to carbon dividends. In order to analyze biomass harvest scenarios, we need to specify the BAU harvest level, the incremental amount of material removed in the biomass cut, the percentage of tops and limbs left on-site, and the silvicultural approaches used to harvest the material. For all scenarios, the biomass carbon calculations assume that in the absence of biomass demand, landowners will continue to manage their forests for timber and other wood products. To establish the BAU baseline, we define both the silvicultural practice used in harvesting the wood and the total quantity removed in the baseline harvest. Generally speaking, our knowledge of logging practices in the state suggests a relatively high probability that landowners would apply diameter limit, partial harvest approaches, removing the largest and best quality trees in the stand. Chapter 3 indicates that based on Forest Cutting Plan data, average harvests historically have removed between 4.5 and 6 tonnes of carbon per acre (approximately 20 to 25 green tons). Using FVS, we modeled this baseline through a removal of 20% of above-ground live stand carbon using a “thin from above” silvicultural prescription.

We also analyzed an alternative baseline in which we assume a significantly heavier BAU harvest, one that removes approximately 32% of the above-ground live carbon. We include this BAU to account for uncertainty regarding which landowners will be more likely to harvest biomass. This scenario would be consistent with the assumption that landowners who have harvests that are heavier than statewide averages would be more likely to harvest biomass.

We then created three biomass harvest options, designed to model light, medium and heavy biomass cuts, all of which include the removal of 65% of all tops and limbs. These were combined with the two BAUs to generate six scenarios representing the impact of different management and harvest assumptions on the timing of the transition from carbon debt to carbon dividends. The results for the six scenarios are summarized in the table included as Exhibit 6-9 (next page). For each scenario, the table shows the quantity of carbon removed in the biomass harvest (i.e., the carbon removal incremental to the harvest in the timber only BAU) and statistics on the recovery by decade of this carbon through growth of the stand. For each scenario, the first row provides the difference in tonnes of total stand carbon between the BAU stand and the biomass stand in years 10 through 90. The second row indicates the tonnes of carbon recovered by the biomass stand relative to the BAU. The third row presents the cumulative percentage of the original biomass carbon recovered by decade.[14]

6.3.3.1 Impacts of Alternative BAUs

The results graphed in Exhibit 6-10 demonstrate that carbon recovery times are somewhat, but not highly, sensitive to assumptions about the volume of timber removed in the BAU harvest. The graph shows carbon recovery curves for Scenarios 1 and 5, the light and heavy BAU harvests, followed by a medium-intensity biomass cut, in this case removal via a diameter limit cut of biomass down to a residual stand basal area of 60 ft^2. The results indicate that the heavier BAU results in a somewhat, but not dramatically, more rapid recovery of carbon in the stand following the biomass harvest. Carbon debts resulting from biomass replacement of coal-fired electricity capacity would take about 20 years in the heavy BAU case, and about 25 years in the light BAU scenario. After these points in time, carbon dividends begin to accrue because atmospheric GHG levels are below those that would have resulted had an equivalent amount of energy been generated using fossil fuel.

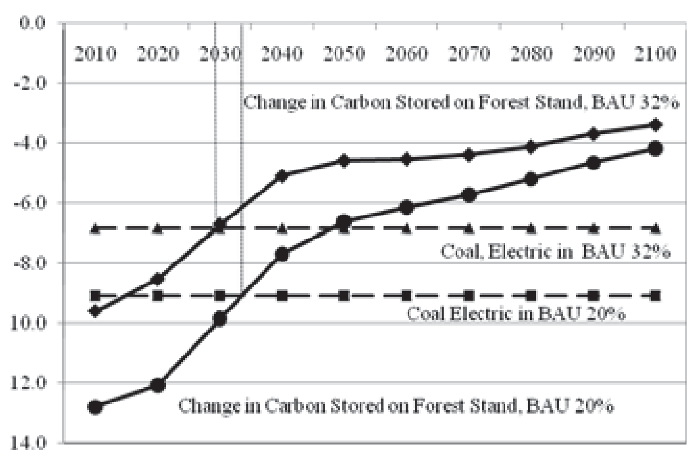
13 In the case of large-scale electricity generated by natural gas, the scenario here assumes a very efficient combined-cycle technology, and this provides a high-end estimate of carbon debts compared to biomass replacement at less efficient natural gas facilities.

14 For example, in Scenario 1, in year 1 the harvest resulted in an initial loss of 9.6 tonnes of total stand carbon (of which 9.1 tonnes is above-ground live carbon). By year 10, the difference in total stand carbon has narrowed to 8.5 tonnes, the relative differences in stand carbon accumulation between the two stands. In this case the biomass stand accumulated an additional 1.1 tonnes of carbon more than the BAU stand (9.6 tonnes minus 8.5 tonnes). This represents recovery of 11.1% of the original carbon removed in the biomass harvest (1.1/9.6).

Exhibit 6-9: Graph of Carbon Recovery Times for Scenarios 1 and 5 (tonnes carbon)

Scenario			BAU vs. Biomass Total Stand Carbon Difference by Year								
Number	**Description**	**Harvest**	**10**	**20**	**30**	**40**	**50**	**60**	**70**	**80**	**90**
	BAU32%-BioBA60	9.1	8.5	6.7	5.1	4.6	4.5	4.4	4.1	3.7	3.4
1	CumRecovered		1.1	2.9	4.5	5.0	5.1	5.2	5.5	5.9	6.2
	%Recovery		**11.1**	**30.2**	**47.1**	**52.5**	**53.1**	**54.5**	**57.2**	**61.6**	**64.8**
	BAU32%-Bio40%	1.8	2.1	1.7	1.3	1.1	0.9	0.7	0.6	0.5	0.4
2	CumRecovered		0.8	1.2	1.6	1.9	2.0	2.3	2.3	2.5	2.6
	%Recovery		**28.1**	**41.0**	**54.6**	**63.4**	**68.5**	**77.3**	**79.0**	**84.1**	**86.4**
	BAU32%-BioHHBA40	14.1	14.4	12.1	9.6	8.3	7.7	6.9	6.2	5.3	4.7
3	CumRecovered		-0.4	2.0	4.4	5.7	6.4	7.1	7.8	8.8	9.3
	%Recovery		**-2.6**	**14.0**	**31.2**	**41.0**	**45.4**	**50.5**	**55.5**	**62.5**	**66.7**
	BAU20%-Bio40%	5.7	5.7	4.9	4.0	3.2	2.6	2.0	1.7	1.4	1.2
4	CumRecovered		0.0	0.8	1.8	2.5	3.2	3.7	4.0	4.3	4.5
	%Recovery		**0.7**	**13.4**	**28.5**	**41.5**	**51.3**	**60.1**	**65.3**	**69.9**	**73.5**
	BAU20%-BioBA60	13.0	12.1	9.9	7.7	6.6	6.1	5.7	5.2	4.6	4.2
5	CumRecovered		0.7	3.0	5.1	6.2	6.7	7.1	7.6	8.2	8.6
	%Recovery		**5.6**	**23.0**	**39.9**	**48.2**	**52.1**	**55.4**	**59.5**	**63.8**	**67.4**
	BAU20%-BioHHBA40	18.0	17.9	15.2	12.3	10.3	9.3	8.3	7.3	6.2	5.5
6	CumRecovered		-0.7	2.0	5.0	6.9	7.9	8.9	9.9	11.0	11.7
	%Recovery		**-4.2**	**11.7**	**28.8**	**39.9**	**46.1**	**51.9**	**57.6**	**64.0**	**68.3**

Exhibit 6-10: Graph of Carbon Recovery Times for Scenarios 1 and 5 (tonnes carbon)



6.3.3.2 Impacts of Alternative Biomass Harvest Intensities

Next we examined the impact of varying the intensity of the biomass harvest on carbon debt recovery. Exhibit 6-9 shows the impact of the light, medium and heavy biomass harvests when combined with the heavy harvest BAU and the comparable results when a lighter BAU harvest is assumed.

The results suggest that for very light biomass harvests, the time required to pay off the carbon debt and begin accumulating dividends is relatively rapid. This is evident in Scenario 2—a heavy BAU coupled with a light biomass harvest—where only 3 tonnes of biomass carbon is removed. In this example, both oil-thermal and coal-electric debts are recovered in the first decade and natural gas electric debts are paid back in approximately 50 years. As discussed in Section 6.3.3.4 below, the rapid recovery occurs because the small removal is comprised of a much greater proportion of logging residues that would have been left on the ground to decay in a BAU harvest. This relatively large magnitude of the decay losses in the BAU results in a rapid recovery of lost carbon in the biomass harvest. Such light harvest, however, would not necessarily produce the supplies forecast in Chapter 3 and may not be the economic choice of landowners.

As harvest intensity increases, however, recovery times become longer. Scenarios 1, 4 and 5, where biomass harvests range from 5.7 to 13.0 tonnes of carbon, all have carbon recovery profiles that are longer than Scenario 2, although all three show steady progress in the recovery of carbon debts. In the three scenarios, oil-thermal debts are recovered roughly between years 10 and 20 and coal-electric debts are recovered between years 20 and 30. For Scenarios 3 and 6, where the biomass removal is close to what would be considered a clearcut, the stand harvested for biomass actually loses carbon relative to the BAU stand in the first decade, creating a delay in carbon recovery that persists for many decades. This may be the result of complex interactions between regeneration and woody debris decay in the years immediately following harvest, although in the case of these more extreme harvests, we may be pushing the model to an extreme case where its results are simply less robust. Given the low likelihood that most biomass harvests will be in the form of clearcuts (see Chapter 3), we do not view the uncertainties in the Scenario 3 and 6 results as having great relevance to the overall patterns of carbon recovery.

6.3.3.3 Impacts of Alternative Silvicultural Prescriptions

The impact of different silvicultural prescriptions has been more difficult to evaluate using the FVS model. The present set of scenarios uses a thin-from-above strategy linked to residual stand carbon targets for all harvests. These types of harvests tend to open the canopy and promote more rapid regeneration and growth of residual trees. While this silvicultural approach may provide a reasonable representation of how a landowner who harvests stands heavily in a BAU is likely to conduct a biomass harvest, it is less likely that someone who cuts their land less heavily would continue to remove canopy trees for biomass (unless they had an unusual number of canopy cull trees remaining after the timber quality trees are removed). More likely in this case is that the landowners would harvest the BAU timber trees and then selectively remove poor quality and suppressed trees across all diameter classes down to about 8 inches. We hypothesized that this type of harvest would result in a slower recovery compared to thinning from above. Unfortunately, the complexity of this type of harvest was difficult to mimic with FVS.

Although project resources were not adequate to manually simulate this type of harvest for all FIA stands, we did conduct a sensitivity analysis for two stands with average volumes. For each of these stands we simulated a BAU harvest removing 20% of the stand carbon, followed by removal of residual trees across all diameter classes above 8 inches down to basal areas similar to the target in Scenario 4. For these two stands, the results, shown in Exhibit 6-11, do indicate a slowing of carbon recovery profiles relative to Scenario 4, although two stands are not enough to draw any conclusions about average impacts of this silvicultural prescription. What can be said is that stands harvested in this manner will probably recover carbon more slowly than would be suggested by Scenario 4; how much more slowly on average we did not determine; it is clear however that on a stand-by-stand basis the magnitude of the slowdown can vary considerably.

6.3.3.4 Impacts of Harvesting Methods and the Role of Tops and Limbs

The harvest and use of tops and limbs for biomass can have an important influence on carbon recovery times and profiles: tops and limbs decay quickly if left in the forest and so their use comes with little carbon "cost" which tends to shorten carbon recovery times. Conversely, if tops and limbs from a biomass harvest of cull trees were left in the woods to decay, this "unharvested" carbon would delay recovery times, effectively penalizing wood biomass relative to fossil fuels. Tops and limbs are available from two "sources" in our biomass harvest scenarios: (1) the material left behind following an industrial roundwood harvest in a BAU scenario and (2) tops and limbs from standing trees harvested specifically for bioenergy in the biomass harvest scenarios.

As discussed in the wood supply analysis in Chapter 3, the harvest of tops and limbs would likely be economical only when harvested with whole-tree systems. Biomass harvested in this manner can be used for any type of bioenergy technology. However, biomass can also be harvested with traditional methods or cut-to-length methods when these systems are preferred due to operating restrictions and/or landowner preferences. These roundwood operations tend to be more costly, but yield higher-quality bole chips that are preferred by thermal, CHP and pellet facilities. Importantly, leaving tops and limbs behind as forest residues would increase carbon recovery times for bioenergy technologies that utilize the bole chips that are produced. The discussion that follows helps to demonstrate how the use of tops and limbs affects our carbon recovery results.

The carbon recovery times in the six scenarios presented in Exhibit 6-9 are all based on the assumptions that 100% of tops and limbs are left in the forest in the BAU scenarios and 65% of all tops and limbs (from both the BAU and the incremental biomass harvest) are harvested in the biomass scenarios. These carbon recovery times (for the three BAU32 scenarios) are compared with the carbon recovery times when all tops and limbs are left in the forest in Exhibit 6-12.

Exhibit 6-11: Carbon Recovery Times
Alternative Harvest Analyses (tonnes carbon)

Scenario			BAU vs. Biomass Total Stand Carbon Difference by Year								
Number	Description	Harvest	10	20	30	40	50	60	70	80	90
	BAU20:Bio40DBH	7.5	8.1	6.6	3.6	2.3	1.7	0.5	-0.2	-0.7	-0.9
1	CumRecovered		-0.6	0.9	3.9	5.2	5.8	7.0	7.8	8.2	8.5
	%Recovery		-9.6	15.1	63.5	84.6	94.8	113.9	126.4	133.6	137.8
	BAU20:Bio40	5.9	6.0	4.4	2.4	2.1	3.3	1.6	1.8	-0.5	0.2
1	CumRecovered		0.0	1.5	3.5	3.8	2.7	4.4	4.2	6.5	5.8
	%Recovery		-0.3	25.6	59.2	64.4	44.7	73.7	70.2	108.9	97.1
	BAU20:Bio40	4.2	4.3	4.4	4.3	3.2	1.3	1.6	0.4	0.6	0.0
2	CumRecovered		-0.1	-0.3	-0.1	0.9	2.9	2.6	3.8	3.5	4.2
	%Recovery		-2.7	-6.4	-3.1	22.6	68.6	62.5	90.4	84.4	100.9
	BAU20:Bio40	6.4	6.0	5.1	4.1	3.5	1.9	2.0	0.0	0.5	0.4
2	CumRecovered		0.4	1.3	2.2	2.8	4.4	4.4	6.3	5.9	5.9
	%Recovery		6.1	20.4	34.8	44.6	69.5	69.1	99.4	92.3	93.5

When tops and limbs are left on-site, all three scenarios show net carbon losses between the initial period and the 10-year mark; in addition, carbon losses in year 10 are substantial relative to the recovery levels in the scenarios in which tops and limbs are taken and used for bioenergy. Scenario 2 (the lightest biomass harvest) shows the greatest impact from not utilizing tops and limbs, with carbon recovery times delayed by about three decades (about 50% of the original biomass harvest was comprised of tops and limbs). Thus, if BAU32 was followed by a light biomass harvest of only roundwood for use by a thermal facility, carbon debt recovery would require 20 to 30 years (when compared to oil-based thermal), rather than occurring in less than 10 years when tops and limbs are taken in whole-tree harvests.

In contrast, in the heavier biomass harvests, recovery times are extended only about ten years. In Scenario 1, the carbon debt incurred by replacing oil thermal by biomass thermal would be recovered in 20 years instead of the 10 years indicated when tops and limbs are utilized. In Scenario 3, carbon debt recovery times for replacement of oil thermal are extended from 20 years to 30 years.

Finally, it is interesting to consider the "harvest" and use of just tops and limbs. While this may not be directly applicable to forest management in Massachusetts (due to poor markets for pulpwood and limited opportunities for log merchandizing), it may be representative of situations involving non-forest biomass sources, such as tree trimming/landscaping or land clearing. The results in this case (also shown in Exhibit 6-12) indicate rapid recovery, with nearly 70% of the carbon losses "recovered" in one decade. Thus, all bioenergy technologies—even biomass electric power compared to natural gas electric—look favorable when biomass "wastewood" is compared to fossil fuel alternatives.

Exhibit 6-12: The Impact of Tops and Limbs on Carbon Recovery Times in BAU32

Number of Years from Initial Harvest					
	10	20	30	40	50
Scenario 1					
Original (with T&L)	11%	30%	47%	53%	53%
No T&L	-9%	11%	31%	38%	38%
Scenario 2					
Original (with T&L)	28%	41%	54%	63%	68%
No T&L	-12%	-4%	16%	31%	39%
Scenario 3					
Original (with T&L)	-3%	14%	31%	41%	45%
No T&L	-22%	-6%	14%	25%	31%
Tops and Limbs Only	68%	87%	93%	96%	97%

6.3.3.5 Impacts of Differences in Stand Harvest Frequencies

A final factor that merits consideration in interpreting the modeling results is the effect of harvest frequencies on the timing of the transition of carbon debt to carbon dividend. Frequent re-entry to the stand to remove biomass has the general effect of extending carbon recovery times. For example, if a stand is re-entered before the time at which carbon levels have recovered to the point where atmospheric concentrations are equivalent to those from fossil fuel burning, a new carbon debt is added to what remains of the initial one and the period required for that stand to reach the equivalent flux point is extended. Conversely, if a second harvest is not conducted until after the stand has begun contributing to actual reductions in GHG levels relative to a fossil fuel scenario, net benefits in the form of carbon dividends will have been positive; additional benefits will depend on the amount of carbon debt incurred in the second harvest and the growth rate of the forest following the additional removal.

As a result of this effect, it is clear that carbon recovery times are sensitive to the frequency at which a landowner chooses to harvest. Data on frequency of harvests indicates landowners who manage for timber typically cut their stands relatively frequently, which suggests our estimated carbon recovery times may be shorter than would actually occur in practice; as a result actual times to the to pay off carbon debts and begin accumulating carbon dividends may be longer.

6.3.3.6 Carbon Dividends

Beyond the point in time when the carbon debt is paid off, and as long as the total carbon recovery rates of stands harvested for biomass are at least as high as the recovery rates in the BAU stands, the carbon dividends from biomass energy continue to accumulate. This means that in the years after the point of carbon debt repayment, there will be less carbon in the atmosphere than had a comparable amount of energy been generated with fossil fuel. As long as the stand harvested for biomass is accumulating carbon faster than the BAU stand, this benefit—lower GHG concentrations relative to the fossil fuel scenario—continues to increase. Even if the two stands ultimately reach a point where carbon accumulates at the same rates, there continues to be a dividend in the form of an ongoing reduction in GHG levels from what they would otherwise have been. As a result, the magnitude of carbon dividends varies depending on the year in which they are evaluated. Exhibit 6-13 indicates the year in which the carbon debt is paid off and provides estimates of the percentage carbon dividend in 2050 and 2100, 40 and 90 years respectively after the modeled biomass harvest.[15]

As discussed in more detail in Section 6.1.2, the carbon dividends in the table indicate the extent to which burning biomass has

[15] FVS simulations become increasingly uncertain as they are extended over long time periods. We believe 90-year simulations represent a reasonable length of time for providing insights into long-term carbon recovery effects.

reduced GHG levels beyond what they would have been had the same energy been generated from fossil fuels. For example, if a biomass thermal plant with an initial carbon debt of 15% emitted 150 tonnes of lifecycle carbon, and the harvested forest recovered an incremental 115 tonnes of carbon over 60 years compared to a BAU scenario, the carbon dividend is 73%. This indicates that the biomass carbon debt has been completely recaptured in forest carbon stocks and in addition GHGs have been reduced by 73%[16] from what they would have been if fossil fuels had been used to generate the equivalent amount of energy. In this context, a carbon dividend of 100% indicates that biomass combustion has achieved full carbon neutrality—all the energy emissions from biomass burning have been fully offset in the form of newly sequestered carbon.

As was the case for carbon debt payoff, the dividend levels clearly indicate benefits are strongly a function of the fossil technology that is being replaced. Where whole-tree harvesting is used, replacement of oil-fired (#6) thermal by biomass thermal results in carbon dividends in excess of 38% by 2050 even in the slowest carbon recovery scenario. These reductions in GHG levels relative to a fossil fuel baseline rise to greater than 60% by 2100. With the exception of biomass replacement of natural gas electric capacity, carbon dividends after 90 years always result in fossil fuel offsets that exceed 40%. These dividends, however, are potentially reduced if stands are re-entered and additional material is harvested prior to the 90-year reference point discussed above. Carbon dividends are consistently low (and in one case negative) for biomass replacement of natural gas electricity generation.

Another way of comparing the relative contributions of carbon debts and carbon dividends is to estimate the difference in cumulative net atmospheric carbon emissions between using biomass and fossil fuel for energy at some future point in time. Due to the importance of demonstrating progress in reducing greenhouse gas emissions by 2050 as part of the Massachusetts Global Warming Solutions Act, we have provided such a comparison for our six harvest scenarios in Exhibit 6-14.

Conceptually, the analysis is perhaps best understood as follows. In the first year, a bioenergy plant consumes a specified volume of wood and establishes a carbon debt relative to the amount of carbon that would have been released in generating the same amount of energy from a fossil fuel alternative. The pattern is then repeated each year and continues until the year 2050. We then calculate the total difference in atmospheric carbon in 2050 from each harvest year and sum the results. For example, the difference in carbon from the first year is simply equal to our estimate of the carbon dividend in year 2050, 40 years after our initial harvest. The difference in carbon from the second year is the carbon dividend that we observe after 39 years, the difference in carbon from the third year is the carbon dividend that we observe after 38 years, etc. The process continues until the last year (2050) at which time the difference in carbon is equal to the difference in year one, or in other words, it is equal to the initial carbon debt.[17] This allows us to compute the total carbon "savings" from burning biomass for a 40-year period, and then compare this value with the total amount of carbon that would have been released by using fossil fuel. When expressed in this manner, the concept is identical to our carbon dividend; however, rather than calculating a dividend at a single point in time, we now have measured the cumulative dividend in 2050, which indicates the total net change in atmospheric carbon at that time due to 40 years of biomass use.

The cumulative dividend net of forest carbon resequestration results from these calculations are shown in Exhibit 6-14: a value of 0% indicates that the carbon dividends during the 2010–2050 period have exactly offset the carbon debt; a positive value indicates that the cumulative carbon dividends have more than offset the carbon debts and have reduced atmospheric carbon compared to what would have been the case had fossil fuels been used (for example, 22% for oil (#6), thermal in harvest scenario 1 indicates that atmospheric carbon is 22% lower in 2050 due to the replacement of oil with biomass); a negative value indicates that total carbon dividends have not yet offset the cumulative debt levels (for example, -13% for natural gas, thermal in harvest scenario 1 indicates that there is still 13% more carbon in the atmosphere in 2050 as a result of having replaced a natural gas thermal plant with biomass and operating it for 40 consecutive years.

Several key observations can be made from these results: (1) the percentage carbon dividend for the entire 2010–2050 period is significantly less than the single year percentage dividend in 2050 that was based only on emissions in 2010 (shown in Exhibit 6-13, next page)—the dividend resulting from only the initial year of emissions will always be the maximum because our empirical analysis has shown that forest carbon resequestration is generally an increasing function (at least after the first few decades); (2) cumulative carbon dividends are positive for oil (#6), thermal for all harvest scenarios; using biomass to displace residual fuel oil in thermal applications would result lower atmospheric carbon levels by an average of about 20% in 2050; (3) cumulative carbon dividends are mostly negative in 2050 for the three other fossil fuel technologies indicating that 40 years is not sufficient for biomass to reduce atmospheric carbon levels using these technology/fuel combinations.

Finally, it should be noted that extending this analysis beyond 2050 will continue to show higher cumulative dividends over

[16] Carbon dividend = (total carbon recovered – carbon debt)/(total carbon emissions –carbon debt) or (115 –(0.15*150))/(150-(150*0.15)) = 73%

[17] Mathematically, there are several ways to compute these values: 1) sum the carbon differences in 2050 for each harvest year, as described above; 2) sum the total carbon released from biomass (net of forest carbon recapture) from 2010–2050 and compare this with the total carbon released from 40 years of burning fossil fuel; or, equivalently, 3) sum the total excess carbon generated from burning biomass (the excesses prior to the point of equal carbon flux) and compare these with the sum of carbon reductions relative to fossil fuel during the phase when dividends are positive.

Exhibit 6-13: Carbon Debt and Dividends

Harvest Scenario	Fossil Fuel Technology	Carbon Debt Payoff (yr)	Carbon Dividend	
			2050	2100
1	Oil (#6), Thermal	7	47%	58%
	Coal, Electric	21	32%	46%
	Gas, Thermal	24	26%	41%
	Gas, Electric	>90	-38%	-9%
2	Oil (#6), Thermal	3	64%	75%
	Coal, Electric	12	54%	68%
	Gas, Thermal	17	50%	65%
	Gas, Electric	45	7%	35%
3	Oil (#6), Thermal	14	38%	62%
	Coal, Electric	30	21%	52%
	Gas, Thermal	36	13%	47%
	Gas, Electric	89	-61%	3%
4	Oil (#6), Thermal	10	53%	76%
	Coal, Electric	27	40%	70%
	Gas, Thermal	31	34%	67%
	Gas, Electric	59	-22%	39%
5	Oil (#6), Thermal	15	46%	64%
	Coal, Electric	25	31%	54%
	Gas, Thermal	28	24%	49%
	Gas, Electric	86	-41%	6%
6	Oil (#6), Thermal	15	39%	66%
	Coal, Electric	32	22%	56%
	Gas, Thermal	37	14%	52%
	Gas, Electric	85	-59%	11%

time. When cumulative dividends through 2100 are considered (Exhibit 6-15), they are higher than the results shown for 2050, although these longer term results will overstate benefits if biomass comes from forests that are harvested more than once or experience significant mortality-causing natural disturbance during the 2010–2100 period.

Exhibit 6-14: Cumulative Carbon Dividends: 2010 to 2050

Harvest Scenario	Fossil Fuel Technology			
	Oil (#6), Thermal	Coal, Electric	Gas, Thermal	Gas, Electric
1	22%	-3%	-13%	-110%
2	34%	11%	3%	-80%
3	8%	-22%	-34%	-148%
4	15%	-13%	-24%	-129%
5	16%	-11%	-22%	-126%
6	7%	-25%	-36%	-153%

Exhibit 6-15: Cumulative Carbon Dividends: 2010 to 2100

Harvest Scenario	Fossil Fuel Technology			
	Oil (#6), Thermal	Coal, Electric	Gas, Thermal	Gas, Electric
1	40%	19%	12%	-63%
2	56%	42%	36%	-18%
3	31%	8%	0%	-86%
4	43%	24%	17%	-54%
5	37%	16%	9%	-69%
6	31%	8%	-1%	-86%

The interpretation of the carbon dividend results should recognize that neither carbon dividends nor carbon debts provide direct indications of the associated environmental benefits or damages. This would require a detailed analysis of the actual climate impacts of increased GHG levels in the period before carbon debts are paid off and lower GHG levels after that point in time. Potential non-linearity in the climate damage functions make such formal benefit-cost analysis challenging and beyond the scope of this study; consequently we leave this analysis to other researchers. Nonetheless, information on initial carbon debts, dividends accrued up to a point 90 years in the future, and estimates of the number of years needed to pay off carbon debts and begin accruing benefits should help inform the development of biomass energy policies.

6.3.4 DISCUSSION OF RESULTS

The analyses presented above make clear that technology choices for replacing fossil fuels, often independent of any forest management considerations, play an important role in determining the carbon cycle implications of burning biomass for energy. The choice of biomass technology, and the identification of the fossil capacity it replaces, will establish the initial carbon debt that must be recovered by forest growth above and beyond BAU growth. These carbon debts vary considerably across technologies. For typical existing configurations, replacement of oil-fired thermal systems with biomass systems leads to relatively low carbon debts. Carbon debts for large-scale electrical generation are higher. Because of its much lower GHG emissions per unit of useable energy, replacing natural gas for either thermal or electric applications results in significantly higher carbon debts than incurred in replacing other fossil fuels.[18] The carbon recovery profile for combustion of wood pellets is roughly similar to burning green wood chips in terms of total lifecycle GHG emissions. CHP facilities, particularly those that optimized for thermal rather than electricity applications, also show very low initial carbon debts.

While the relative ranking of technologies by their carbon recovery times provides useful insights on relative carbon emissions per unit of useable energy, the specific time required in each case to pay off carbon debts and begin realizing the benefits of biomass energy, represented in this study by the carbon dividends, depends on what happens in the forests harvested for biomass fuel. The results of our analyses provide some broad insights into biomass carbon dynamics but are also subject a number of uncertainties that are difficult to resolve.

A key finding of our work is that the magnitude and timing of carbon dividends can be quite sensitive to the forest management practices adopted by landowners. Carbon recovery times can differ by decades depending upon assumptions about (1) the intensity of harvests; (2) the silvicultural prescriptions and cutting practices employed; (3) the fraction of the logging residues removed from the forest for biomass; and (4) the frequency

18 Cowie (2009) draws similar conclusions in a recent presentation of work on IEA Bioenergy Task 38.

at which landowners re-enter stands to conduct future harvests. If the landowners responding to demands for increased biomass are the same ones who harvest their lands heavily today, then it is probably reasonable to assume that carbon debts are recovered relatively rapidly, along the lines suggested by our Scenario 1. In this case, the transition from debt to dividends that results from replacing oil-fired thermal with biomass is between 10 and 20 years and the biomass coal-electric transition occurs after 20 to 30 years. But if the response is more evenly distributed across all landowners and the biomass harvests are more heavily focused on removal of suppressed and understory cull trees, we expect that recoveries would likely be slower. How much slower, and the impact on subsequent carbon dividends, cannot be predicted without a better understanding than we currently have about future landowner forest management practices. While detailed landowner surveys might improve our understanding of this issue, this uncertainty cannot be completely resolved until we can observe actual landowner behavior in response to increased biomass demand.

Finally, it is important to emphasize that after the point in time where GHG levels are equivalent for biomass and fossil fuels, biomass energy provides positive reductions in future GHG levels. Over time, under some scenarios these carbon dividends can become substantial, reducing GHGs by up to 85% in some scenarios relative to continued fossil fuel use. But the key question remains one of the appropriate weighting of near-term higher GHG levels with long-term lower ones. Policymakers will need to sort out these issues of societal time preferences and weight near term higher GHG emissions against longer term lower ones.

6.4 FINAL CONSIDERATIONS

The Massachusetts Department of Energy Resources has indicated that it hopes this study will provide valuable information to help guide its decisions on biomass energy policy. The study discusses a complex subject that is technically challenging and inevitably we have not been able to resolve all critical uncertainties. Policymakers should carefully weigh the significant uncertainties that remain, as well as other factors not addressed by our study, in deciding whether to encourage or discourage biomass development. In light of that, we conclude with some general observations on how the results of our carbon accounting analyses should be interpreted by policymakers and the public at large.

- As suggested in the discussion of carbon recovery, we have used average and/or typical values for GHG emissions from biomass and fossil fuel energy facilities. With continually evolving technology, biomass developers may be able to demonstrate lower GHG emissions per unit of useable energy. This can be expected to reduce carbon debts and change the overall time required to pay off these debts through forest growth. Consequently, our carbon debt and dividend conclusions should be viewed as representative of typical or average conditions today, a state of affairs that will likely change in the future given the evolution of technologies.

- Our carbon analysis considers only biomass from natural forests. Tree care and landscaping sources, biomass from land clearing, and C&D materials have very different GHG profiles. Carbon from these sources may potentially enter the atmosphere more quickly and consequently carbon debts associated with burning these types of biomass could be paid off more rapidly, yielding more immediate dividends. Our results for biomass from natural forests likely understate the benefits of biomass energy development relative to facilities that would rely primarily on these other wood feedstocks.

- Our analyses of recovery of carbon recovery by forests have focused primarily on average or typical forest conditions in Massachusetts. The responses of individual stands vary around these average responses, with some stands recovering carbon more rapidly and others less rapidly than the average. Due to the complexity of responses at the individual stand level, this study has not been able to isolate the characteristics of rapidly recovering stands using FVS. Should better data become available on this topic, it might be possible to design and implement forest biomass harvest policies that accelerate the average carbon recovery times reported here.

- Some landowners may face alternative BAU baselines that we have not considered, and this raises issues about generalizing our results too widely—particularly beyond Massachusetts and New England. We have used the historical harvest trends in Massachusetts as the basis for our BAUs and we believe this is the most likely future for landowners in the Commonwealth. However, we cannot rule out other BAU scenarios that could change the carbon recovery results in important ways. For example, if no biomass plants are sited in Massachusetts, will landowners actually face an alternative BAU where they can sell this material to out-of-state energy facilities? If so, GHG impacts are likely the same as if the material were used in state. Or is there an alternative BAU for an out-of-state facility that sells renewable energy to Massachusetts—for example bioenergy facilities in Maine that may be competing for biomass supplies that would otherwise go to paper production and enter the GHG system relatively more quickly? The existence of alternative baselines would result in different carbon debts and recovery profiles than those that we have identified for Massachusetts.

- Views about how long it will take before we have truly low or no carbon energy sources play a critical role in biomass policy decisions. If policymakers believe it will take a substantial amount of time to develop and broadly apply low or no carbon sources of energy, they may be more inclined to promote the development of biomass. Conversely, if they think that no or low carbon alternatives will be available relatively soon, say in a matter of one or two decades, they may be less inclined to promote development of biomass, especially for applications where carbon debts are relatively higher and where longer payoff times reduce future carbon dividends.

- Concerns about the relative importance of short- versus long-term consequences of higher carbon emissions may also play a role in how one interprets the results of this study. Those who believe that short-run increases in GHG levels need to be avoided at all costs will be less likely to favor biomass development than those focused on the potentially quite significant, but longer term benefits of reduced GHG levels that could ultimately result from biomass development.

In light of all these factors, we stress that our work should be viewed as providing general indicators of the time frames for recovery of biomass carbon and the key factors that influence these estimates. Uncertainties remain and we have tried to be transparent about them. For the variety of reasons discussed above, the carbon recovery and dividend profile for a specific facility is likely to deviate from the average facilities analyzed in this report. As such, we suggest that new energy and environmental policies that rely on insights from this study should clearly take into account the impacts of the various uncertainties embedded in the report's analytic framework, assumptions and methods.

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APPENDIX 1-A

FEDERAL, STATE AND REGIONAL BIOMASS ENERGY POLICIES

The following summary of federal and select state policies and incentives related to the development of biomass energy facilities addresses the following areas:

- A summary of relevant federal policies affecting the development of biomass energy;
- A review of relevant regional policies and regulatory initiatives impacting the development of biomass energy;
- A summary of current policies in the State of Massachusetts that relate to renewable energy and biomass facilities as well as state policies related to sustainable forestry issues; and
- A review of notable biomass policies and incentives in other states, with a particular focus on renewable energy, forest sustainability, carbon regulation, and climate change issues.

The information presented here is drawn from several sources including work prepared for the Biomass Energy Resources Center by Shems Dunkiel Raubvogel & Saunders PLLC, research conducted by Charles Niebling of New England Wood Pellet, analysis conducted by the Biomass Thermal Energy Council, analysis conducted by Jesse Caputo of the Environmental and Energy Study Institute, and analysis provided by the Pinchot Institute for Conservation.

This discussion includes a historical review of prior federal policies under the Public Utility Regulatory Policy Act of 1978 (PURPA), which spurred development of many existing biomass energy facilities in the U.S.

I. Federal Policies & Incentives Relevant to Biomass

Federal incentives for renewable energy (including biomass) have taken many forms over the past four decades. The focus of most of these programs has been on encouraging renewable electricity generation and, more recently, production of renewable transportation fuels, such as ethanol with little attention to or investment in the thermal energy sector. Consequently, biomass as an energy source is being primarily directed into the large scale production of liquid biofuels and/or large scale electric generation. In addition, existing renewable energy policy provides little or no connection to efficiency requirements, sustainable forestry provisions or carbon sequestration goals.

As discussed below, federal policy initially focused on encouraging renewable electricity generation by requiring utilities to purchase electricity from renewable energy generators at a fixed cost through the Public Utility Regulatory Policy Act (PURPA).

More recently, federal policy has shifted towards encouraging renewable energy through tax incentives and direct grants—with the primary focus on renewable transportation fuels and renewable electricity generation.

A. Historical Review of Major Federal Policies Incentivizing Biomass Development

Development of biomass energy facilities in the U.S. in the last four decades has been largely driven by federal energy policies and incentives designed to encourage renewable energy development and diversification of energy sources. Historically speaking, the most important of these federal policies was the Public Utility Regulatory Policy Act (PURPA). PURPA was passed in 1978, primarily in response to the sharp spike in oil prices during the 1970s, and embodied a national effort to reduce reliance on foreign oil and diversify domestic energy generation.

To achieve these goals, PURPA contained several provisions specifically designed to spur development of renewable energy generation in the U.S. Chief among these provisions was the requirement that utilities purchase the power output from certain small renewable energy generators—known as "qualified facilities" (QF)—at the utility's "avoided cost." The certainty of these guaranteed, highly favorable rates led to a dramatic increase in renewable energy generation, including an estimated three-fold increase in biomass facilities in the 1980s and early 1990s.

But the spike in biomass facilities developed under PURPA was relatively short lived and market conditions and regulatory changes have limited the value and application of the "avoided cost" provisions of PURPA. Deregulation efforts in the 1990s also led to increased competition among energy generators in many parts of the U.S., opening the grid to a greater number of small or independent power producers. Due to the perceived increase in competition in power markets, Congress revised PURPA in 2005 and, combined with subsequent regulator action, PURPA no longer serves as a significant incentive for the development of biomass facilities in the US.

B. Current Federal Policies Related to Biomass Energy Development

Current federal policies and incentives for renewable energy facilities take many different forms. This review focuses on incentives relevant to biomass power or combined heat and power vs. the production of liquid biofuels, which is beyond the scope of this project. These incentives have moved away from the "guaranteed cost" approach implemented under PURPA, and now rely primarily on either (1) federal tax incentives, or (2) direct federal grants or loans from federal agencies. Specific examples of these two types of incentives are summarized below.

Federal Tax Incentives

Overall, existing federal tax incentives for renewable energy focus on electric power generation and the production of liquid biofuels. Consequently, biomass feedstocks are being directed preferentially towards these types of energy applications. In addition, existing federal tax incentives provide little or no connection to efficiency requirements, sustainable forestry provisions or carbon sequestration goals.

1. Production Tax Credit (IRS Code Section 45)

The Renewable Energy Production Tax Credit (PTC) provides a tax credit for owners or operators of qualifying renewable electric generation facilities for the first ten years of operation. Qualifying resources include both "closed-loop biomass" and "open loop biomass" facilities that sell power to the public. Co-fired units (those burning both fossil fuel and biomass) are not eligible. The 2009 American Recovery and Reinvestment Act recently extended the PTC for projects placed into service from the end of 2010 through the end of 2013. The benefit of this production tax credit can only be realized by an entity with sufficient taxable income to take advantage of the credit; the PTC will not provide an incentive to entities that do not pay federal taxes unless they partner with other entities with federal tax exposure. This program is not subject to annual appropriations, but does need to be extended every year.

2. Business Energy Investment Tax Credit (IRS Code Section 48)

The Business Energy Investment Tax Credit (ITC) provides a credit based on the value of the investment in certain types of electrical generation and combined heat and power (CHP) biomass facilities and was also recently expanded to apply to general closed and open loop biomass facilities. The CHP ITC is a 10 percent tax credit for the first 15MW of a system up to 50MW. The CHP ITC extends through December 31, 2016. The 2009 ARRA also expanded the availability of the ITC to other closed loop and open loop biomass facilities (besides CHPs) that are otherwise eligible for the PTC. Under this new provision, the owner of a biomass facility that qualifies for the PTC may elect to claim an ITC in lieu of the PTC.

3. Grant in Lieu of Investment Tax Credit

The 2009 ARRA also created a new program that allows taxpayers eligible for the ITC to elect to receive a grant from the U.S. Treasury. This is technically a direct federal grant, not a tax credit, but is covered here for sake of continuity with the related ITC and PTC provisions. This cash grant may be taken in lieu of the federal business energy investment tax credit (ITC). Eligible CHP property includes systems up to 50 MW in capacity that exceed 60 percent energy efficiency. The efficiency requirement does not apply to CHP systems that use biomass for at least 90 percent of the system's energy source.

4. Clean Renewable Energy Bonds (CREBs) (IRS Code Section 54)

The Clean Renewable Energy Bonds (CREBs) program was created by the Energy Policy Act of 2005. The program provides "tax-credit" bonds to renewable energy projects developed by governments or electric coops. The bonds are awarded to eligible entities on a competitive basis by the IRS. Both closed-loop and open-loop biomass facilities are eligible for the program. Unlike typical bonds, which pay interest to the bondholder, the tax-credit bonds provide bondholders a credit against their federal income tax, effectively providing the issuer of the bonds a 0% loan with the federal treasury covering the interest payments. The 2009 ARRA allocated an additional $1.6 billion for this program.[1]

5. Five-Year Modified Accelerated Cost Recovery System (MACRS) (IRS Code Section 168)

Certain biomass facilities are also eligible for the Modified Accelerated Cost-Recovery System (MACRS). Under the MACRS program, businesses may recover investments in certain properties through accelerated depreciation deductions. At the present time, combined heat and power facilities powered by biomass are in the five-year accelerated depreciation class for this program.

6. New Market Tax Credits

Although not specific to biomass projects, The New Markets Tax Credit (NMTC) Program could potentially provide an additional tax incentive for biomass facilities, depending on the location of the facility, and potentially, on the clients the facility serves. The purpose of the NMTC program is to encourage development that would benefit low income people and populations. It provides a tax credit against Federal income taxes for taxpayers making qualified equity investments in designated Community Development Entities (CDEs). The potential application of this tax credit program to any particular project is very site specific. A map of NMTC-qualifying areas in western Massachusetts can be found at http://www.ceimaine.org/content/view/215/233/. $13.4 billion in NMTC have been finalized or committed by May 2009 out of $19.5 billion awarded through 2008. An additional $1.5 billion was awarded in May 2009.

Federal Grants and Loans

The second major category of incentives is direct grants and loans from federal agencies including primarily the Department of Agriculture (USDA) and the Department of Energy (DOE). Some of the relevant programs from each agency are discussed below. The major portion of these funds are available through the Department of Energy, with the Exception of USDA's Biomass Crop Assistance Program (as discussed below). While there are several important programs at USDA that address smaller scale biomass energy options, these initiatives generally have low appropriations levels and, in many cases, have never been funded. By contrast the DOE programs generally focus on large scale production of liquid biofuels and/or electric generation and are funded at much higher levels than the array of USDA programs. Again, this creates incentives for certain biomass energy applications—biofuel production and electricity generation—at the federal level.

A. USDA Grant & Loan Programs

[1] http://www.taxalmanac.org/index.php/Sec._54._Credit_to_holders_of_clean_renewable_energy_bonds.

The majority of relevant USDA biomass programs are based on provisions of the 2008 Farm Bill. The relevant portions of the bill are focused on encouraging development of renewable biomass facilities. The Farm Bill specifically includes biomass in the definition of renewable energy, and defines "renewable biomass" broadly as "any organic matter that is available on a renewable or recurring basis from non-Federal land" and certain materials from public lands, if harvested during preventative treatments to reduce hazardous fuels, address infestation, or restore "ecosystem health." The following specific programs may provide incentives for biomass facilities and projects.

1. The Rural Energy for America Program (Sec. 9007 of 2008 Farm Bill)

The Rural Energy for America Program (REAP) provides financial assistance to rural communities in order for them to become more energy independent through increased production of renewable energy and energy efficiency. Grants and loan guarantees are available for energy efficiency and renewable energy investments (including biomass) for agricultural producers and rural small businesses. Grants may be up to 25% of project cost (up to a maximum of $500,000 for renewable energy projects), loan guarantees are capped at $25 million/loan and grants and loan guarantees together may be up to 75%. A portion of grants are reserved for small projects.

2. The Rural Energy Self-Sufficiency Initiative (Sec. 9009 of 2008 Farm Bill)

Authorizes a new program to provide financial assistance to increase energy self-sufficiency of rural communities. Provides grants to conduct energy assessments, formulate plans to reduce energy use from conventional sources, and install integrated renewable energy systems. Integrated renewable energy systems are defined as community-wide systems that reduce conventional energy use and incorporate renewable energy use. Federal-cost share for any grant is limited to 50% of project cost. The 2008 bill authorizes appropriations of $5 million annually for FY 2009-12.

3. Biomass Crop Assistance Program (BCAP) (Sec. 9011)

Created in the 2008 Farm Bill, BCAP is an innovative program intended to support establishment and production of eligible crops for conversion to bioenergy, and to assist agricultural and forest landowners with collection, harvest, storage, and transportation (CHST) of these eligible materials to approved biomass conversion facilities (BCF).

The program pays for up to 75% of establishment costs of new energy crops. In addition, farmers participating in a selected BCAP project area surrounding a qualifying BCF can collect 5 years of payments (15 years for woody biomass) for the establishment of new energy crops. An additional matching payment of up to $45/ton (on a $1 to $1 basis) to assist with collection, harvest, storage and transportation (CHST) of an eligible material to a BCF will also be available for a period of 2 years.

However, the launch of this new program has proved problematic. Rather than focusing funding on the front-end of the program, establishment of new energy crops, the Farm Service Agency (FSA) announced funds for the back-end of the program (via a Notice of Funding Availability (NOFA) for the Collection, Harvest, Storage and Transportation (CHST). It also interpreted CHST as an "entitlement" and allowed payment for a broad range of agricultural and forested materials delivered to an approved BCF.

The result amounted to a substantial, new subsidy for the existing wood market with significant market impact. Large numbers of existing biomass conversion facilities (led by lumber, pellet and paper mills currently burning wood for their own energy use without a federal subsidy) submitted applications to USDA to be approved as qualifying facilities. Consequently, funds obligated (though not yet spent) for BCAP through the end of March 2010 soared to over $500 million, more than seven times BCAP's estimated budget of $70 million in the 2008 Farm Bill. The USDA now estimates BCAP costs at $2.1 billion on CHST from 2010 through 2013.

The proposed rule for BCAP was announced February 8 with a final rule anticipated late summer 2010.

4. Forest Biomass for Energy (Sec. 9012)

Authorizes new competitive research and development program to encourage use of forest biomass for energy. To be administered by USDA's Forest Service; priority project areas include:

- developing technology and techniques to use low-value forest biomass for energy production
- developing processes to integrate energy production from forest biomass into biorefineries
- developing new transportation fuels from forest biomass
- improving growth and yield of trees intended for renewable energy

Authorizes appropriation of $15 million annually for FY 2009-12.

5. Community Wood Energy Program (Sec. 9013)

The Community Wood Energy Program is administered by the USDA and provides grants of up to $50,000 to state and local governments to develop community wood energy plans. Once a plan has been approved, qualified applicants may request up to 50% matching grants toward the capital costs of installing biomass energy systems. The Farm Bill authorizes $5 million per year from FY 2009 through FY 2012 for this program, but to date, no funds have actually been appropriated.

6. Business and Industry Guaranteed Loan Program

The Business and Industry Guaranteed Loan Program administered by USDA Rural Development. The purpose of the B&I Guaranteed Loan Program is to improve, develop, or finance business, industry, and employment and improve the economic and environmental climate in rural communities. A borrower may be a cooperative, corporation, partnership, or other legal entity organized and operated on a profit or nonprofit basis; an Indian tribe on a Federal or State reservation or other federally recognized tribal group; a public body; or an individual. A borrower may be eligible if they are engaged in a business that will reduce reliance on nonrenewable energy resources by encouraging the development and construction of renewable energy systems.

7. Rural Business Enterprise Grants (RBEGs)

The RBEG program provides grants for rural projects that finance and facilitate development of small and emerging rural businesses (defined as those that will employ 50 or fewer new employees and have less than $1 million in projected gross revenues). The program is not specific to biomass projects, but biomass projects could benefit from the grants.

B. Department of Energy Grant & Loan Programs

1. Renewable Energy Production Incentive

The Renewable Energy Production Incentive (REPI) provides financial incentive payments for electricity generated and sold by new qualifying renewable energy generation facilities. Qualifying facilities- including biomass facilities—are eligible for annual incentive payments for the first 10-year period of their operation, subject to the availability of annual appropriations in each Federal fiscal year of operation. This program serves as an alternative to the PTC for entities that are not eligible to take advantage of that tax program (i.e. entities that do not have federal tax liabilities).

2. DOE Loan Guarantee Program

Title XVII of the Energy Policy Act of 2005 authorizes DOE to provide loan guarantees for energy projects that reduce air pollutant and greenhouse gas emissions. DOE recently released its second round of solicitations for $10 billion in loan guarantees for energy efficiency, renewable energy, and advanced transmission and distribution projects under EPACT 2005 with a primary focus on transportation and electric generation. The final regulation provides that the DOE may issue guarantees for up to 100 percent of the amount of a loan. The 2009 ARRA extended the authority of the DOE to issue loan guarantees and appropriated $6 billion for this program. Under this legislation, the DOE may enter into guarantees until September 30, 2011.

3. Energy Efficiency and Conservation Block Grants

The Energy Efficiency and Conservation Block Grant (EECBG) Program provides federal grants to local government, Indian tribes, states, and U.S. territories to improve energy efficiency and reduce energy use and fossil fuel emissions in their communities. Activities eligible for funding include energy distribution technologies that significantly increase energy efficiency, including distributed generation, CHP, and district heating and cooling systems. A total of $3.2 billion was appropriated for the EECBG Program for fiscal year 2009. This funding will generally flow directly to states or local municipalities and is typically awarded on a competitive basis.

4. Sec. 471, Energy Independence and Security Act of 2007

Sec. 471 of the 2007 Energy Bill authorizes a program for Energy Efficiency and Sustainability Grants for implementing or improving district energy systems, combined heat and power applications, production of energy from renewable resources, developing sources of thermal energy and other applications. These funds would leverage investments by eligible public sector entities including institutions of high education, local governments, municipal utilities and public school districts. The Act authorizes $250 million for grants and $500 million for loans under this program for FY2009-2013 with maximum grants limited to $500,000. The program has never been funded.

5. Other ARRA Programs and Funding Opportunities Specific to Combined Heat and Power Facilities.

In addition to these major programs, the 2009 ARRA authorized a number of small grant and loan programs through DOE, some of which apply to potential biomass facilities including CHP and thermal district energy facilities. Of these, two grant opportunities were particularly relevant to biomass energy applications.

DOE-FOA-0000044, issued through the National Energy Technology Laboratory, provided $156 million for deployment of CHP systems, district energy systems, waste energy recovery systems, and efficient industrial equipment. Approximately 350 responses were submitted representing $9.2 billion in proposed projects with a $3.4 billion federal share, a demand far in excess of the available funding. DE-FOA-0000122, provided $21 million for community renewable energy development, with awards going to 5 projects nationwide.

The Department of Energy also has other solicitations specifically for combined heat and power systems. For example, the Industrial Technologies Program (ITP), part of DOE's Office of Energy Efficiency and Renewable Energy, recently released a funding opportunity for or up to $40 million in research, development and demonstration of combined heat and power (CHP) systems, based on annual appropriations, not ARRA funds.

II. Review and summary of Massachusetts state policies relevant to biomass energy and sustainable forestry.

Massachusetts has implemented policies to increase the use of biomass to meet energy needs in the electricity sector, the transportation sector, and the building heating sector, although state policies are focused primarily on implementing biomass to replace fossil fuels in the electricity and transportation sectors. Combined with the state's regulatory scheme designed to implement the Regional Greenhouse Gas Initiative (RGGI) (which sets an emissions cap on fossil fuel fired electrical generation systems of 25 megawatts or greater), this has created significant incentives in the state driving biomass towards larger scale electric generation capacity vs. smaller scale or thermal applications. A recent exception to this trend is the Massachusetts Green Communities Act of 2008 which established new Renewable and Alternative Energy Portfolio Standards (RPS and APS) which allow eligible CHP units to receive credits for useful thermal energy. This program promotes the installation and effective operation of new CHP units for appropriate residential, commercial, industrial, and institutional applications. It does not, however, eliminate or counterbalance the overall focus on encouraging development of the biomass electric power sector.

Following is a summary of the range of statutory and regulatory provisions that directly address biomass in Massachusetts, with an emphasis on biomass policy within the electricity sector.

A. Biomass in Renewable Energy Policy

1. Electricity

Massachusetts has two regulatory schemes that directly impact the incentives for developing biomass-fueled electricity in the state. The first is the Massachusetts Renewable Portfolio Standard (RPS), which is administered by the Department of Energy Resources (DOER), and the second is the implementation of the state's membership in the Regional Greenhouse Gas Initiative (RGGI), which is administered by the Department of Environmental Protection (DEP). We discuss RGGI and the Massachusetts regulatory scheme implementing RGGI in Part III, in the context of regional biomass policy initiatives. In this section of the paper, we discuss the implications for biomass energy under the RPS program regulations as currently written, recognizing that DOER has suspended RPS review of all proposed biomass-fueled electricity generators pending completion of the Manomet study.

The Massachusetts RPS program currently mandates that all retail electricity suppliers must include minimum percentages of RPS Class I Renewable Generation, RPS Class II Renewable Generation, and RPS Class II Waste Energy in the retail electricity they sell to consumers. For 2010, the Class I requirement is 5 percent, the Class II Renewable requirement is 3.6 percent, and the Class II Waste requirement is 3.5 percent. The definition of "eligible biomass fuel" under the RPS program is:

> *Fuel sources including brush, stumps, lumber ends and trimmings, wood pallets, bark, wood chips, shavings, slash and other clean wood that are not mixed with other unsorted solid wastes; by-products or waste from animals or agricultural crops; food or vegetative material; energy crops; algae; organic refuse-derived fuel; anaerobic digester gas and other biogases that are derived from such resources; and neat Eligible Liquid Biofuel that is derived from such fuel sources.*

It is notable that this definition contains no "sustainability" requirement. The RGGI definition, by contrast, does contain such a requirement, though the criteria for sustainability in that definition are not fleshed out at this time. This definition also includes liquid biofuels, which are expressly excluded from the definition of "eligible biomass" for purposes of the Massachusetts RGGI program.

Biomass facilities may qualify as RPS Class I or Class II generation units as long as they are classified as "low-emission, advanced biomass Power Conversion Technologies using an Eligible Biomass Fuel." Both the Class I and Class II RPS regulations also allow generators that co-fire to qualify as RPS Renewable Generation as long as certain requirements are met. This provision in the RPS program is analogous to the biomass exemption from carbon dioxide emissions accounting in the RGGI program.

In 2008, the Massachusetts Green Communities Act established new Renewable and Alternative Energy Portfolio Standards (RPS and APS) allowing Combined Heat and Power facilities to be included as an eligible technology provided the thermal output of a CHP unity is used in Massachusetts. APS eligible CHP units receive credits for the useful thermal energy of a CHP unit delivered to Massachusetts end-uses, subject to the formula included in the regulations. The DOER rules issued for this program will, for the first time in the Commonwealth, promote the installation and effective operation of new CHP units for appropriate residential, commercial, industrial, and institutional applications.

There are two other regulatory programs, aside from the DOER process for RPS approval, which could address the sustainability and the carbon neutrality of biomass-fueled electricity generation. The first is the Energy Facilities Siting Board review process for generation facilities, and the second is the Massachusetts Environmental Policy Act (MEPA).

All electricity generation facilities proposed in Massachusetts must be approved by the Energy Facilities Siting Board within the Department of Public Utilities. The Board reviews the environmental impacts of generation facilities to ensure that the plans for the facility are consistent with current health and environmental protection policies and the commonwealth's energy policies; and that the plans minimize environmental impacts and related mitigation costs. The Board is also responsible for adopting performance standards for emissions from generating facilities. The Board also has the authority to preempt other state agency or local regulatory bodies that pose hurdles to electricity facility siting. In making such decisions, the board has already has a track record of taking issues of carbon neutrality and sustainable fuel supplies into account.

The other regulatory vehicle for screening the sustainability and carbon neutrality of biomass electric generation facilities is environmental impact review through MEPA. However, as MEPA review is only mandatory for any new electric facility with a capacity of 100 MW or more, it may not have a great deal of promise for effective implementation of regulatory goals for biomass because facilities are unlikely to meet this size threshold. Further, the process is "informal," and "MEPA and [its implementing regulations] do not give the Secretary the authority to make any formal determination regarding . . . consistency or compliance" with "any applicable Federal, municipal, or regional statutes and regulations."

2. Transportation and Heating

The focus of Massachusetts policy on biomass in the transportation and heating sectors seems to be on liquid biofuels. In 2006, the commonwealth instituted a policy requiring the use of a minimum percentage of biofuels in state vehicles and instituting a pilot study on the use of biofuels in heating systems in state buildings. Additionally, the commonwealth created the "Advanced Biofuels Task Force" in late 2007 to explore how Massachusetts could accelerate use of advanced biofuels.[2] The Task Force issued a report, which explores the environmental life cycle of biofuels, and contains recommendations heavily focused on the transportation sector, in the spring of 2008. (Advanced Biofuels Task Force, 2007)

Following the report's publication, the commonwealth passed the Clean Energy Biofuels Act, which exempts cellulosic biofuels from the state gasoline tax, requires transportation diesel and home heating oil to contain 2-5% of cellulosic biofuels from 2010-2013, and requires the commonwealth to develop a low-carbon fuel standard that will reduce transportation GHG emissions by 10%. DOER has been implementing the Biofuels Act through regulations related to the tax exemptions for cellulosic biofuels. The proposed regulation includes a definition of "Lifecycle Greenhouse Gas Emissions" and eligibility criteria for the tax exemption that include requirements for the reductions in lifecycle GHG emissions achieved by eligible biofuels compared to fossil fuels.

B. Biomass and Forestry

Massachusetts has a statutory framework as well as administrative regulations addressing forest harvesting. By statute, the Commonwealth recognizes that:

> *the public welfare requires the rehabilitation, maintenance, and protection of forest lands for the purpose of conserving water, preventing floods and soil erosion, improving the conditions for wildlife and recreation, protecting and improving air and water quality, and providing a continuing and increasing supply of forest products for public consumption, farm use, and for the woodusing industries of the commonwealth.*

Accordingly, it is the policy of the Commonwealth that:

> *all lands devoted to forest growth shall be kept in such condition as shall not jeopardize the public interests, and that the policy of the commonwealth shall further be one of cooperation with the landowners and other agencies interested in forestry practices for the proper and profitable management of all forest lands in the interest of the owner, the public and the users of forest products.*

[2] Advanced Biofuels Task Force. (2007). Retrieved 2010 from http://www.mass.gov/?pageID=eoeeaterminal&L=4&L0=Home&L1=Energy%2c+Utilities+%26+Clean+Technologies&L2=Alternative+Fuels&L3=Clean+Energy+Biofuels+in+Massachusetts&sid=Eoeea&b=terminalcontent&f=eea_biofuels_biofuels_report&csid=Eoeea

The Massachusetts Department of Conservation and Recreation (DCR) is the regulatory agency charged with administering timber harvesting on public and private forest lands. DCR has jurisdiction over all commercial forest cutting that produces more than 25,000 board-feet or 50 cords on any parcel of land. Under the regulations, any landowner planning a cut within DCR's jurisdiction must complete a "forest cutting plan." Proposed cuts that include a clearcut exceeding 25 acres are subject to additional regulatory process mandated by the Massachusetts Environmental Policy Act.

In addition to administering the Forest Cutting Practices regulations DCR has joined with DOER to form the Sustainable Forest Bioenergy Initiative (SFBI). The goal of the SFBI is to "provide research and development on forest management and market infrastructure needs, and enable the state to provide the resources necessary to develop the biomass supply market." The Initiative has produced a number of technical reports regarding woody biomass energy, woody biomass supply in the state, forest harvesting systems for biomass production, economic impact analyses, and silvicultural and ecological considerations for forest harvesting.

Documents produced under the SFBI state that the "carbon dioxide produced by burning wood is roughly equal to the amount absorbed during the growth of the tree." Other documents estimate between 500,000 and 890,000 dry tons of biomass from public and private forests located in the state can be sustainably harvested per year, and that the demand for woody biomass from forestry is approximately 526,000 dry tons per year. The SFBI has carried out extensive state-specific work on biomass energy and forest sustainability issues relevant to this study.

<u>C. Other Massachusetts Incentives Related to Renewable or Alternative Energy Development and Biomass</u>

The following paragraphs comprise a set of tax incentives and other programs available in Massachusetts that may have an impact on biomass development in the Commonwealth.

<u>1. Renewable Energy Trust Fund</u>—Two separate public benefits funds to promote renewable energy and energy efficiency for all customer classes. The renewable energy fund, known as the Massachusetts Renewable Energy Trust (MRET), is supported by a surcharge on customers of all investor-owned electric utilities and competitive municipal utilities in Massachusetts. The Massachusetts Technology Collaborative (MTC), a quasi-public research and development entity, administers the MRET with oversight and planning assistance from the Massachusetts Department of Energy Resources (DOER) and an advisory board. The MRET may provide grants, contracts, loans, equity investments, energy production credits, bill credits and rebates to customers. The fund is authorized to support a broad range of renewable energy technologies including low-emission advanced biomass power conversion technologies using fuels such as wood, by-products or waste from agricultural crops, food or animals, energy crops,

biogas, liquid biofuels; and combined heat and power (CHP) systems less than 60 kilowatts (kW).

2. Large Onsite Renewables Initiative (Massachusetts Renewable Energy Trust Fund)—Program funds support grid-tied renewable-energy projects (excluding PV) greater than 10 kilowatts (kW) in capacity that are located at commercial, industrial, institutional and public facilities that will consume more than 25% of the renewable energy generated by the project on-site. The applicant and project site must be a customer of a Massachusetts investor-owned electric distribution utility or a municipal utility that pays into the Renewable Energy Trust. Grant awards may be used to facilitate the installation of renewable-energy projects on existing buildings (retrofits) or in conjunction with new construction/ major renovation projects, including green buildings.

3. Business Expansion Initiative—The Massachusetts Technology Collaborative (MTC), as administrator of the state's Renewable Energy Trust Fund, offers loans to support renewable energy companies entering or expanding within the manufacturing stage of commercial development. Companies that currently, or plan to, manufacture renewable energy technology products in Massachusetts are generally eligible. Products may be new or existing, or a combination of the two.

4. Clean Energy Pre-Development Financing Initiative (Massachusetts Technology Collaborative)—Offers grants and loans to support the development of grid-connected renewable energy systems in New England. Eligible technologies or resources include wind energy; naturally flowing water and hydroelectric power; landfill gas; anaerobic digestion; and low-emission, advanced power-conversion technologies using "eligible biomass fuel." Biomass and wind energy projects must have a minimum capacity of three megawatts (MW), and hydroelectric, landfill gas and digester gas projects must have a minimum capacity of 250 kilowatts (kW). Projects must be designed to lead to the development of new renewable grid-connected generating capacity for the wholesale electricity market. Therefore, more than 50% of the renewable energy produced must be provided to the wholesale market.

5. Massachusetts Technology Collaborative (MTC) - Sustainable Energy Economic Development (SEED) Initiative—Provides financial assistance to support renewable-energy companies in the early stage of development. Applicants are companies that generally have a unique technology but have not yet demonstrated commercial viability to an extent sufficient to attract venture capital. Awards of up to $500,000 are available as a convertible loan on a competitive basis. Since 2004, the Massachusetts Renewable Energy Trust has invested over $4.9 million in Massachusetts-based renewable energy companies through the SEED Initiative.

6. Net Metering—The state's investor-owned utilities must offer net metering. Municipal utilities may do so voluntarily. (The aggregate capacity of net metering is limited to 1% of each utility's peak load.

7. The Biomass Energy Policy and Market Development Program (U.S. Department of Energy's State Energy Program)—The Biomass Energy Policy and Market Development Program promoted biomass with a comprehensive biomass energy policy initiative to improve the policy and market conditions and foster biomass economic development. The project informed the Renewable Portfolio Standard eligibility criteria for biomass projects and forestry management, assessed the regional woody biomass resource, and evaluated the potential for rural economic development. It increased the use of biofuels and biodiesel for building heating through outreach, formal collaboration with other state agencies to formalize comprehensive biomass energy policy and implementation plan, engaging with public and private sectors to inform policy discussions and understand and address issues, promote project activities within state agencies and private market to adopt bioenergy fuels, legal review and input, outreach policy and project development to industry, municipalities, concerned citizens, and renewable energy developers.

8. Alternative Energy and Energy Conservation Patent Exemption (Corporate)—Corporate excise tax deduction for (1) any income received from the sale or lease of a U.S. patent deemed beneficial for energy conservation or alternative energy development by the Massachusetts Department of Energy Resources, and (2) any income received from the sale or lease of personal property or materials manufactured in Massachusetts and subject to the approved patent.

9. Alternative Energy and Energy Conservation Patent Exemption (Personal)—Personal income tax deduction for any income received from a patent deemed beneficial for energy conservation or alternative energy development. The Massachusetts Commissioner of Energy Resources determines if a patent is eligible.

10. Biodiesel Blend Mandate (Massachusetts Session Law 206)—All diesel motor vehicle fuel and all other liquid fuel used to operate motor vehicle diesel engines must contain at least 2% renewable diesel fuel by July 1, 2010; 3% renewable diesel fuel by July 1, 2011; 4% renewable diesel fuel by July 1, 2012; and 5% renewable diesel fuel by July 1, 2013. For these purposes, eligible renewable diesel fuel includes diesel fuel that is derived predominantly from renewable biomass and yields at least a 50% reduction in lifecycle greenhouse gas (GHG) emissions relative to the average lifecycle GHG emissions for petroleum-based diesel fuel sold in 2005. The Massachusetts Department of Energy Resources must also study the feasibility, benefits, and costs of applying the percentage mandates on a statewide average basis rather than for every gallon of diesel motor fuel sold.

11. Biofuels Incentives Study (Massachusetts Session Law 206)—A special commission is established to study the feasibility and effectiveness of various forms of incentives to promote the development and use of advanced biofuels in Massachusetts, including, but not limited to, production credits, the production and harvesting of woody biomass, feedstock incentives and direct consumer credits for the use of advanced biofuels in various applications. The commission reported the results of its investigation and recommendations in March 2009.

12. Massachusetts - Green Power Purchasing Commitment—In April 2007, Massachusetts Governor Deval Patrick signed Executive Order 484, "Leading by Example: Clean Energy and Efficient Buildings." This order establishes energy targets and mandates for state government buildings and directed state government agencies to procure 15% of annual electricity consumption from renewable sources by 2012 and 30% by 2020. This mandate may be achieved through procurement of renewable energy supply, purchase of renewable energy certificates (RECs), and/or through the production of on-site renewable power. Only renewable sources that qualify for the Massachusetts renewable portfolio standard (RPS) are eligible.

13. Boston - Green Power Purchasing—In April 2007, Boston Mayor Thomas Menino issued an executive order that established a green power purchasing goal of 11% for the city government, and a goal of 15% by 2012. The executive order also requires all existing municipal properties to be evaluated for the feasibility of installing solar, wind, bio-energy, combined heat and power (CHP), and green roofs and set goals for greenhouse gas emissions reductions, recycling, green building, vehicle fuel efficiency, biofuels use, and the development of the Boston Energy Alliance, a non-profit corporation dedicated to implementing large-scale energy efficiency, renewable energy, and demand response projects citywide.

III. Review and summary of regional policy and regulatory initiatives impacting development of biomass energy.

A. Regional Greenhouse Gas Initiative

Massachusetts is a member of the Regional Greenhouse Gas Initiative (RGGI), a group of ten New England and Mid-Atlantic states that has agreed to cap greenhouse gas (GHG) emissions from the generation of electric power and to lower this cap over time. Under the RGGI agreement, each participating state has been assigned a certain number of carbon dioxide allowances, serving as that state's emissions cap. The individual states are responsible for assigning carbon allowances to the covered emissions sources within the state, and to adopt rules to implement the emissions accounting, trading, and monitoring necessary to achieve the initial cap and subsequent reductions of GHG emissions. Eight of the ten participating states, including Massachusetts, exempt biomass-fueled electricity generation from carbon dioxide emissions accounting such that any carbon dioxide emitted from biomass-fueled processes is not counted against that state's carbon cap. The RGGI emissions cap applies to fossil fuel-fired electricity generators with a capacity of 25 megawatts or greater.

As a consequence of this program, Renewable Energy Credits are issued in Massachusetts (and the other RGGI states) for biomass-fueled electric power generation, providing a significant incentive and market driver for large scale biomass-fueled electric power generation over other uses such as thermal, Combined Heat and Power, or smaller scale applications.

In addition to the complete exemption from the RGGI system for generators whose fuel composition is 95 percent or greater biomass, the RGGI Model Rule and all participating states except for Maine and Vermont provide partial exemptions for facilities that co-fire with smaller percentages of biomass. This partial exemption provides that any carbon dioxide emissions attributable to "eligible biomass" may be deducted from a facility's total carbon dioxide emissions when calculating whether the facility's emissions are within its carbon-allowance budget.

The Model Rule defines "eligible biomass" as follows:

> *Eligible biomass includes sustainably harvested woody and herbaceous fuel sources that are available on a renewable or recurring basis (excluding old-growth timber), including dedicated energy crops and trees, agricultural food and feed crop residues, aquatic plants, unadulterated wood and wood residues, animal wastes, other clean organic wastes not mixed with other solid wastes, biogas, and other neat liquid biofuels derived from such fuel sources. Sustainably harvested will be determined by the [participating state's designated regulatory agency].*

In Massachusetts, the regulation defines "eligible biomass" identically except that it deletes the language "and other neat liquid biofuels." Additionally, the Massachusetts definition states, "Liquid biofuels do not qualify as eligible biomass." It is unclear why the Massachusetts regulators decided to eliminate liquid biofuels from the definition, especially since liquid biofuels are included in the "eligible biomass fuel" definition in Massachusetts' RPS program. As illustrated in Table 1, below, several other states similarly exclude liquid biofuels from their RGGI definitions of "eligible biomass." In Massachusetts, the Department of Environmental Protection is charged with defining the sustainable harvesting criteria for sustainable harvesting of biomass under RGGI.

Exhibit A-1: Summary of biomass provisions in the RGGI implementing regulations of the ten participating RGGI states.

State	Allows deduction for biomass-attributable emissions	Includes liquids as eligible biomass	Uses December 2008 Model Rule for biomass calculation	Uses January 2007 Model Rule for biomass calculation
Massachusetts	X			X
Connecticut	X		X	
Delaware	X	X	X	
Maine				
Maryland	X	Not found	X	
New Hampshire	X	X	X	
New Jersey	X	X	X	
New York	X		X	
Rhode Island	X		X	
Vermont				

B. Midwestern Greenhouse Gas Reduction Accord and Western Climate Initiative

While RGGI is the only fully developed and implemented regional cap and trade program for GHG emissions reductions in the United States, several Midwestern states and the Canadian province of Manitoba have joined together to achieve GHG emissions reductions through their own regional cap and trade system. The Midwestern agreement is called the Midwestern Greenhouse Gas Reduction Accord (Accord), and in June 2009, the Accord's Advisory Group issued a set of recommendations for emissions reductions targets and for designing a regional cap-and-trade system. The Advisory Group recommended that a broader set of sectors be included in the emissions reduction program than RGGI covers, such that the program would cover not only electricity generation, but also industrial sources, fuels serving residential, industrial, and commercial buildings, and transportation fuels. However, the recommendations include an exemption for carbon dioxide emissions "from the combustion of biomass or biofuels, or the proportion of carbon dioxide emission from the combustion of biomass or biofuels in a blended fuel," which essentially mirrors the RGGI exemption.

After the Advisory Group recommendations were published, the Accord issued a draft Model Rule in October 2009 The rule contains a definition of "eligible biomass" that is exactly identical to the RGGI Model Rule definition, including the liquid biofuels measure. Additionally, the Accord's Model Rule includes the same provision allowing a GHG source to deduct all biomass-attributable GHG emissions from its total GHG emissions when determining compliance with the source's GHG allowance budget. The Accord's Model Rule does not, however, contain any provision detailing how the biomass-attributable GHG emissions are to be calculated.

Similar to RGGI and the Midwestern Accord, several western states and Canadian provinces have joined in the Western Climate Initiative to enact similar GHG emissions reductions through a cap-and-trade system. The WCI, like the Accord, recommends that the program cover not just electricity, but also transportation, industrial and commercial fossil fuel combustion, industrial process emissions, and residential fuel use. Further, the WCI has issued draft program recommendations, which include a recommendation that "biomass determined by each WCI Partner jurisdiction to be carbon neutral" should not be included in the cap-and-trade program, except for reporting purposes. Further, the recommendations state that "[c]arbon dioxide emissions from the combustion of pure biofuels, or the proportion of carbon dioxide emissions from the combustion of biofuel in a blended fuel" would not be included in the program. The WCI recommendations also indicate that each participating jurisdiction "will assess whether and how to include upstream emissions from biofuel and fossil fuel production." These recommendations, unlike the RGGI Model Rule or the Accord's recommendations and Model Rule, exhibit more caution regarding the carbon neutrality of biomass fuel use.

IV. Review and summary of outstanding state policies impacting development of biomass energy, with a focus on renewable energy, forest sustainability and climate issues.

This section provides a summary of relevant policies in several states with notable approaches to biomass development, with a particular focus on renewable energy incentives, forest sustainability and climate change issues. Specifically, this section: characterizes the state-level approach to biomass usage in general; reviews the typical basket of state policies that address biomass; highlights some outstanding state policies with regard to biomass; and concludes with a listing of relevant state policies. It is based on a review of eleven states' policies regarding biomass: Arizona, California, Connecticut, Maryland, Minnesota, Missouri, Oregon, Pennsylvania, Vermont, Washington, and Wisconsin.

The thrust of state policies promoting biomass and/or biofuels is focused on electric generation and less so on transportation and thermal. All surveyed states have numerous policies, programs and/or incentives to promote electric generation from renewable sources of energy, including biomass. A few states have policies to support the use of biomass/biofuels for transportation (California, Minnesota, Oregon, Pennsylvania, Washington, and Wisconsin) and/or for thermal production (Arizona, Connecticut, Missouri, Oregon, Pennsylvania, Vermont, Washington, and Wisconsin).

Typically, states include biomass as one of a number of sources of renewable energy in a variety of policies and programs aimed at increasing electric generation from renewable energy. A common method to advance biomass electric generation policies is via renewable portfolio standards, which typically mandate that utilities provide a certain percentage of renewably generated electricity by a certain date. Other common state policies supportive of biomass electric generation are net metering programs; public benefits funds which, among other activities, distribute grants and/or loans for biomass research and/or development; other grant and/or loan programs for biomass research and/or development; power purchasing programs at the state and/or local level; and a variety of tax incentives. The range of tax incentives includes: production tax incentives such as energy production tax credits, or deductions or exemptions for installing certain types of biomass manufacturing systems; sales tax incentives for purchasing qualifying equipment for harvesting, transportation, and manufacturing or processing of biomass; personal tax incentives such as income tax credits and deductions for installation of certain types of renewable energy systems; and property tax incentives such as exemptions, exclusions and credits for property (including equipment) used for the siting of qualifying manufacturing facilities or the transport of biomass.

States with large sources of biomass supply, such as Minnesota, Missouri, Oregon, Washington and Wisconsin, also tend to have biomass-specific policies or programs in addition to general programs such as renewable portfolio standards. These states are also likely to have biomass working groups or a biomass program (Connecticut, Minnesota, Oregon, Pennsylvania, and Vermont). Some have produced biomass reports, including woody biomass supply assessments. (Arizona, California, Minnesota, Oregon, Vermont, Washington, and Wisconsin). These reports typically focus more on biomass promotion and less on sustainability, and some discuss the linkage between biomass utilization and climate change. Finally, some states have produced woody biomass harvesting guidelines that focus on best management practices for harvesting woody biomass in an ecologically sensitive and sustainable manner (Minnesota, Missouri, Pennsylvania, and Wisconsin). All such harvesting guidelines are voluntary, guidance only.

The following state programs stand out regarding the sustainable utilization of biomass for renewable energy generation:

The Vermont Energy Act of 2009 aims to expand the market for renewable-energy technologies in Vermont in a number of innovative programs that address the issue from different directions. Its key elements include: clarification that the Clean Energy Development Fund's grants and loans also apply to thermal energy projects (discussed further below); a standard offer for renewable energy (discussed further below); incentives that allow utilities to recover permitting costs for renewable energy; pilot downtown-community renewable-energy projects in two towns, Montpelier and Randolph (Village Green Program); improvements to residential- and commercial-building standards; provision for the creation of clean energy assessment districts so that towns, cities, and incorporated villages can use municipal bonds to finance residential renewable-energy or energy-efficiency projects; and limitations on the power of municipalities and deeds to prohibit residential installation of renewable-energy and energy-efficiency devices, such as solar panels, residential wind turbines, and clothes lines.

The Vermont Clean Energy Development Fund, Vermont's principle renewable energy incentive program, has provided millions of dollars to wind, solar, biomass, and other renewable energy projects in the form of grants and loans over the past several years. The Vermont Energy Act of 2009 clarified the scope of the CEDF to include thermal energy and geothermal resources, including combined heat-and-power systems, which sets Vermont's program apart from most state programs. Grant funding is available to four categories of projects: pre-project financial assistance, small-scale systems (microturbines, fuel cells, and CHP), large-scale systems, and special demonstration projects. Proposed projects must have an electric generation component and be grid-connected; off-grid projects and thermal projects (except CHP systems) are not eligible. There is a special funding opportunity in 2009 for municipalities, public schools, and colleges to explore renewable energy projects and feasibility up to $5,000. Low-interest fixed-rate loans are available to individuals, companies, nonprofits and municipalities for purchasing land and buildings for qualifying projects, purchasing and installing machinery and equipment, and providing working capital. Eligible clean electric-energy technologies include solar, wind, biomass, fuel cells and combined heat and power.

The **Vermont Standard Offer for Qualifying SPEED Resources** was enacted as part of the 2009 Vermont Energy Act. It requires all Vermont retail electricity providers to purchase electricity generated by eligible renewable energy facilities through the Sustainably Priced Energy Enterprise Development (SPEED) Program. This "feed-in tariff" is intended to provide a reasonable return on investment to renewable energy facility developers, thereby spurring deployment of renewable energy. The program establishes a set price that utilities must pay to purchase renewable energy from certain qualifying sources, by means of long-term contracts. The standard offer price will be available to facilities with a plant capacity of 2.2 MW or less, for a total of 50 MW of renewable power state-wide. The applications for 50 MW of SPEED standard-offer contracts are fully subscribed and a lottery was implemented to select final solar and biomass projects. Wood biomass is included as a potential qualifying renewable energy source, but may only receive the standard offer if the plant's system efficiency is 50% or greater. If the program's goals (included in the appendix) are not met, then the RPS will become mandatory and require the state's electric utilities to meet any increase in statewide retail electricity sales between 2005 and 2012 by using renewables with associated attributes, by purchasing RECs, or by making an alternative compliance payment to the Vermont Clean Energy Development Fund.

Oregon is a biomass leader. It has developed a comprehensive wood biomass supply assessment at state and regional levels. The state's active **Forest Biomass Working Group** has produced a comprehensive analysis of forest biomass opportunities map that includes existing wood-based energy facilities and the power transmission grid. The **Oregon Strategy for Greenhouse Gas Reduction** aims to reduce wildfire risk by creating a market for woody biomass from forests. It incorporates use of biomass into discussions linking climate change, wildfire protection plans, and economic development for rural communities. It notes that an additional 100 MW produced from woody biomass plants would result in the thinning of 2.4 million acres over 30 years, and the average annual sequestration from reduced crown fires and improved forest health would be 3.2 million metric tons of CO_2. This CO_2 reduction is in addition to, and does not include, displacing fossil fuels with biomass fuels. It promotes biofuels use and production, and expands research on how climate change could affect expanded production of renewable power including bioenergy. **Oregon House Bill 2200** authorized the State Forester to establish programs to market, register, transfer or sell forestry carbon offsets on behalf of state forestland beneficiaries, the Forest Resource Trust, and other non-federal forest landowners. The bill recognizes a wide range of forest management activities—those designed to protect our environment as well as those designed to provide our wood products—as having the potential to give rise to forestry carbon offsets. Oregon's **Biomass Logging Bill** (SB 1072) promotes the use of biomass from logging projects on federal land as both a restoration tool and electricity generation mechanism. It also directs the Oregon Department of Forestry to participate in federal forest project planning and land management. It spells out that the "Policy of the State" of Oregon is to support efforts to build and place in service biomass fueled electrical power generation plants that utilize biomass collected from forests or derived from other sources such as agriculture or municipal waste. It requires the Oregon Board of Forestry to direct the State Forester to enter into stewardship contract agreements with federal agencies to carry out forest management activities on federal lands. Finally, the **Oregon Renewable Energy Action Plan** (REAP) outlines a plan of action for renewables. Specifically for biomass, it provides that twenty-five megawatts of new biomass-fueled electric generation will be built or under construction, in addition to 5 megawatts of biogas facilities; it allows biomass facilities to qualify for net metering and allows the Oregon Public Utility Commission to adopt rules to increase the 25-kilowatt limit on a net metering facility for customers of Portland General Electric and Pacific Power; it encourage the development and utilization of small energy efficient biomass heating and electrical systems for heating and providing power to institutions, state offices, schools, etc., especially in rural Oregon; and it promotes greater public awareness of the primary and secondary benefits of biomass energy production.

California's State Biofuels Development Plan / Biofuels Production Mandate and Alternative Fuel Use Study is notable for its ambition. California plans to use biomass resources from agriculture, forestry, and urban wastes to provide transportation fuels and electricity to satisfy California's fuel and energy needs. The state will produce its own biofuels at a minimum of 20% by 2010, 40% by 2020, and 75% by 2050. Regarding the use of biomass for electricity, the state shall meet a 20 percent target within the established state goals for renewable generation for 2010 and 2020. The Bioenergy Action Plan includes: research and development of commercially viable biofuels production and advanced biomass conversion technologies; evaluation of the greenhouse gas reductions benefits of bio-fuels and biomass production and use; evaluation of the potential for biofuels to provide a clean, renewable source for hydrogen fuel; and state agencies' purchase of flexible fuel vehicles as 50% of total new vehicles by 2010.

APPENDIX 2-A

18 SELECTED TECHNOLOGY PATHWAYS

Pathway #1: Power Plant—Electricity (green wood)

This technology pathway is fueled by green wood with bark. On average, woodchips have roughly 40 percent moisture content. This means that while one ton of dry woodchips would produce 16.5 million Btus[1] (MMBtu) of heat, one ton of green woodchips would produce only 9.9 MMBtu. The green wood with bark will have some implications on the emissions of this system as bark has high ash content. This technology pathway will use direct combustion using a fluidized bed. This combustion technique suspends the woodchips in midair using jets of upward-blowing air. This increases the contact between carbon and oxygen and hence increases efficiency. A medium (like sand, or lime) is used to make the process uniform and controllable. The resulting hot gases travel up from the furnace to the boiler to heat water and convert it into a high-pressure steam.

The high-pressure steam then travels to a condensing steam turbine, the secondary process in this pathway. When steam enters the turbine, it is hotter per unit weight than when it exits the turbine. Upon leaving, the exhaust steam is condensed below atmospheric pressure which increases the pressure drop between input and exhaust steam. This produces greater power per unit weight of the input steam. The spinning turbine creates electrical energy.

Lastly, when the hot gases travel out of the furnace, they are likely carrying some ash, fines, and other particulates. In order to reduce the particulate emissions from this pathway, an electrostatic precipitator (ESP) removes particles from the air using an electrostatic charge. Gases are not impeded as they move through the ESP, but particulates like dust and smoke remain instead of leaving with the gas. The clean flue gases are discharged to the atmosphere through a high stack.

Pathway #2: Power Plant—Electricity (co-fired, 20% green wood, 80% coal)

In this pathway, green wood with bark is most commonly co-fired with coal. In co-firing, biomass can burn simultaneously with coal, comprising 20 percent of the load that is combusted in a regular coal boiler system. No efficiency is lost in the process. The intent is to reduce the use of fossil fuel and substitute renewable biomass, which is low-carbon if sustainably managed, and sulfur oxide emissions are lowered because biomass has nearly no sulfur content. When the two fuels are burned and release hot gases, they heat water in the boiler which in turn heats the high-pressure steam needed for the condensing steam turbines (as described in Pathway #1). The turbines create electrical energy.

[1] Btu: British thermal unit, a standard unit of energy equal to the heat required to raise the temperature of one pound of water one degree Fahrenheit

In some current applications, co-firing has been found to increase PM emissions. In this pathway, an ESP will be an important component in collecting particulates from the flue gases.

Pathway #3: Power Plant—Electricity (coal)

This technology pathway utilizes bituminous coal, which is the most abundant type of coal in the United States. It is second highest in energy output (after anthracite). The coal is used in a direct combustion furnace. The hot gases created in the furnace travel upward to the boiler to heat water and convert that into a **steam**. The steam then moves into a condensing steam turbine, as used in Pathway #1. The spinning turbine creates electrical energy.

An ESP is used in this pathway to capture particulates.

Pathway #4: Power Plant—Electricity (natural gas)

This pathway utilizes natural gas. Natural gas is composed mostly of methane, has drastically more energy per unit than either oil or propane, and emits lower amounts of nitrogen oxides and carbon dioxide than oil or coal. In this pathway, it is combusted directly to create steam using simple cycle technology representative of most existing gas-fired systems. The steam moves to a gas turbine, also known as a combustion turbine. Three steps are involved in this process. First, incoming air gets compressed to a very high pressure. Then, the combustor burns the fuel, producing a high-pressure, high-velocity gas. As the gas moves through the combustion chamber, it spins the turbine that creates electricity.

No emissions control equipment is associated with this technology pathway.

Pathway #5: Thermal Energy (cordwood)

Green wood with bark is used in this pathway in the form of cordwood. Firewood is commonly measured in units of cords which are a measure of volume, not weight. A standard cord of stacked wood is equal to the amount of wood in a four foot by four foot by eight foot stack (this is equivalent to 128 cubic feet). The energy content of cordwood can vary widely based on species and moisture content. It is very important to note that cords are also used as a volume measure of roundwood and this roundwood cord measure is different (a cord of roundwood is only 85 cubic feet, compared to 128 cubic feet of cordwood). This difference between the two measures is due to less air space between pieces of cordwood that are cut, split, and neatly stacked.

The cordwood is loading by hand and combusted directly in a traditional boiler, such as may be found in a home's basement or possibly even an outdoor boiler. This boiler heats water which is used for domestic water and heating purposes (thermal energy) in a residential setting.

Pathway #6: Thermal Energy (cordwood)

This pathway also utilizes cordwood but is combusted in an EPA-certified boiler in a residential setting. These boilers combine high efficiency combustion with hydronic thermal storage. The hot water storage aids in increasing the system's efficiency because the boiler does not have to operate during times of low-load as long as enough thermal storage is available to meet the demand.

Pathway #7a: Wood Pellets (green wood)

This technology pathway produces pellets and is fueled by green wood with bark. The wood is processed so that it can go through the drying and densification process, in which the air is expelled from the wood at very high pressures and then formed into pellets. Natural plant lignin in the pellet material is melted during the extrusion process and holds the pellets together without glues or additives. Pellets have significantly lower moisture content than the woodchips from which they were created (six percent versus an average of 40 percent, respectively) which means they produce greater Btus per unit. This pathway, combined with 7b, represent the full energy implications of using pellets from forst, through production and combustion of pellets.

Pathway #7b: Thermal Energy (pellets)

After the densification of green wood with bark to create pellets, the process in this pathway is to use direct combustion to burn the pellets to create thermal energy. This combustion occurs in the furnace in which the pellets come in direct contact with the fire. The purpose of biomass burner technologies is to get all of the carbon in pellets to react with oxygen in the air to make carbon dioxide. As this is an exothermic reaction, it will generate a lot of heat. The challenge here is to convert all the carbon and get maximum heat. When the flue gas travels out of the furnace, water captures the heat and is then piped throughout the building or number of buildings for heating and domestic hot water. The water used for heating the air is then piped back to the furnace to be re-heated and looped out again.

The emission control device utilized in this pathway is a cyclone-baghouse combination. With the correct design and choice of fabric, particulate control efficiencies of over 99 percent can be achieved even for very small particles (one micrometer or less) by fabric filters or baghouses. The lowest emission rate for large wood-fired boilers controlled by fabric filters reported is 0.01 lb/MMBtu. For large thermal-only applications (boilers over four to five MMBtu), a baghouse is usually sufficient to handle particulate matter (PM) control (along with a multi-cyclone which is generally included with the boiler by the manufacturer). Considered with Pathway 7a, this represents an application using pellets at the commercial scale, from forest to combustion.

Pathway #8: Thermal Energy (green wood)

This technology pathway is fueled by green woodchips with bark and undergoes direct combustion in a fluidized bed (as described in Pathway #1). The interim product is hot water (and not high pressure steam). The water in the boiler will capture the heat from the combustion chamber and will then be piped through the building for heat and hot water, or thermal energy. The cold water will be piped back to boiler.

A fabric filter or baghouse will collect the particulates to lower the emission rates.

Pathway #9: Thermal Energy (heating oil)

This pathway involves the direct combustion of residual heating oil, which includes number 5 and 6 heating oils. These are often referred to as "heavy oils" because they are what remain after gasoline and distillate oils have been extracted in the distillation process. This oil is laden with high amounts of pollutants, sulfur dioxide being one of the greatest. Residual oil has a high viscosity so before it can be used in a boiler, it must be heated so that it flows more smoothly. Once this has been achieved, the oil gets combusted directly in a furnace where it heats water for thermal applications.

No emissions control equipment is associated with this technology pathway.

Pathway #10: Thermal Energy (natural gas)

This pathway involves the direct combustion of natural gas. The gas is combusted in a furnace where it heats water for thermal applications.

No emissions control equipment is associated with this technology pathway.

Pathway #11: CHP—Electricity (green wood)

In this pathway, the green wood with bark goes through direct combustion in a fluidized bed (as described in Pathway #1). In this pathway, the high-pressure steam moves through to the second part of the process that is in a backpressure steam turbine. The steam enters the turbine where it expands. During expansion, some of its thermal energy is converted into mechanical energy that runs an electrical generator. The low pressure steam that exits the turbine returns to the plant to satisfy thermal applications. As backpressure turbines satisfy both process and heating requirements, they are ideal for combined heat and power (CHP) applications that are far more efficient than electrical energy production alone.

An ESP will serve as the pollution control equipment to remove particulates from the air.

Pathway #12: Gasifier—Electricity (green wood)

In this pathway, the green wood with bark is used to create a producer gas using the process of gasification. Gasification is a thermo-chemical process that converts solid fuel materials into combustible gases that can then be used for heat and power. When biomass is heated with a fraction of what is needed for efficient combustion, it gasifies into the interim product, a mixture of carbon monoxide and hydrogen—synthesis gas or syngas or producer gas. Combustion occurs as a result of mixing oxygen with hydrocarbon fuel. Because gaseous fuels mix with oxygen more easily than liquid fuels, which in turn mix more easily than solid fuels, syngas inherently burns cleaner and more efficiently than the solid biomass from which it was made. One advantage of gasification technology is that it is a decentralized energy conversion system that operates economically even when used in small-scale applications. Although the technology is currently not commercially available in the United States, it has proven to be economical in many locations.

The producer gas is then used in an internal combustion engine to power a generator. The generator spins to create electrical energy while waste heat from both the gasifier and the internal combustion engine can be captured and used as thermal energy, thereby creating a CHP system.

Pathway #13: CHP—Electricity (heating oil)

Residual heating oil is combusted directly, in this pathway, to create steam. This pathway differs from the former because the steam moves through to a backpressure steam turbine. As backpressure turbines create both electrical and thermal energy, they are ideal for CHP applications that are far more efficient than electrical energy production alone.

No emissions control equipment is associated with this technology pathway.

Pathway #14: CHP—Electricity (natural gas)

In this technology pathway, natural gas is combusted directly to create steam. The steam travels to a backpressure steam turbine as described in Pathway #11. The electricity produced by the spinning generator and the over-pressurized steam that satisfies thermal applications at the plant fulfills the CHP component.

No emissions control equipment is associated with this technology pathway.

Pathway #15: Cellulosic Ethanol (green wood)

In order to create ethanol, green wood with bark goes through a primary process of hydrolysis and fermentation (ERRE, 2009). This is a multiple step process. First, sulfuric acid is mixed with the woodchips at which point a hydrolysis reaction occurs. Here, the complex chains of sugars that make up the hemicellulose in the wood get broken and release simple sugars. Later in the process, what cellulose remains gets hydrolyzed into glucose. This glucose then goes through the fermentation process, in which microorganisms convert it to ethanol.

As a by-product of ethanol production, lignin can get combusted directly to produce the electricity required for the production process, or, since more electricity is generally created than is needed, selling the electricity may help the process economics.

An ESP can be applied to the furnace in which the lignin is burned to reduce PM emissions.

Pathway #16: Bio-oil & Bio-Char (green wood)

In this Pathway, green wood with bark undergoes a primary process of pyrolysis at a bio-refinery. Pyrolysis is the rapid chemical decomposition of wood in the absence of oxygen, and occurs spontaneously when the temperature is high enough. This process breaks the wood down into a gas, liquid (bio-oil), and a solid (Biochar). By rapidly decomposing the biomass at high temperatures, it will result in a greater amount of bio-oil whereas slow pyrolysis will produce Bio-Char. Bio-oil can be substituted for conventional liquid fuels, and while it contains roughly 54 percent the heating value of #6 fuel oil (Innovative Natural Resource Solutions, 2004), its benefit is that it is sourced from a renewable resource rather than a non-renewable fossil fuel.

As bio-oil can be substituted for conventional fuels, it can be burned in a furnace to heat water for thermal energy applications.

This pathway utilizes an ESP as its emissions control equipment.

Pathway #17: Bio-products (green wood)

This pathway also utilizes green wood with bark to create a syngas through the process of gasification. Syngas is composed of hydrogen and carbon monoxide. The Fischer–Tropsch process (or Fischer–Tropsch Synthesis) is a set of chemical reactions that convert a mixture of carbon monoxide and hydrogen into liquid hydrocarbons. The process, a key component of gas-to-liquids technology, produces a petroleum substitute, typically from biomass for use as synthetic lubrication and as synthetic liquid fuel, such as ethanol. Electricity is also created by combusting lignin, the by-product of ethanol production.

An ESP is used to remove the particulates from the air exiting the plant.

Pathway #18: Gasification—Cellulosic Ethanol (green wood)

In technology pathway #6, green wood with bark undergoes a primary process of fast pyrolysis at a bio-refinery. The bio-oil produced from fast pyrolysis can be used to produce a variety of bio-products, such as plastics, glues, organic fertilizers, and fuel additives.

This pathway utilizes an ESP as its emissions control equipment.

TECHNOLOGY PATHWAYS SUMMARY

B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T
No.	Technology Pathway	Main Product	Co-products	Typical Capacity	Unit	Hours of operation per year	Output Heat Mmbtu/yr	Gross Effi c.	Net Effi c	Heat Input MMBtu/yr	Btu/lb	Tons (dry) per year	Fuel Requirements lbs (dry) /MMBtu output heat	CO2 Emissions lbs/MMBtu input heat	CO2 Emissions lbs/MMBtu output heat	C Emissions lbs/MMBtu input heat	C Emissions lbs/MMBtu output heat	Maximum Affordable Cost of Biomass with 40% MC ($/ton)
							(F)*(H)*3.412			(I)/(J)		(L)/(M)	(N)/(I)		(P)*(L)/(I)	(P)/3.6667	(Q)/3.6667	
1	Power	Electricity		50	MW	7,200	1,228,320	25%	25%	4,913,280	8,500	289,016	470.59	215.69	862.75	58.82	235.29	$31
	(green wood)																	
2	Power	Electricity		500	MW	7,200	12,283,200	32%	32%	7,677,000	8,500	451,588	73.53	215.69	134.80	58.82	36.76	$31
	(coal &									30,708,000	10,506	1,461,451	237.96	205.30	513.25	55.99	139.98	
	green wood-									38,385,000	10,105	1,913,039	311.49	207.38	648.05	56.56	176.74	
	20% co-fi ring)																	
3	Power	Electricity		500	MW	7,200	12,283,200	32%	32%	38,385,000	10,506	1,826,813	297.45	205.30	641.56	55.99	174.97	
	(coal)																	
4	Power	Electricity		500	MW	7,200	12,283,200	33%	33%	37,221,818	1,028	36,207,994	2.95	117.00	354.55	31.91	96.69	
	(Natural Gas)										cubic ft	dtherms	dtherms					
5	Thermal	Thermal		0.1	MMBTH	1,800	180	60%	60%	300	8,500	18	196.08	215.69	359.48	58.82	98.04	$104
	Energy	energy																
	(cordwood)																	
6	Thermal	Thermal		0.1	MMBTH	1,800	180	68%	68%	265	8,500	16	173.01	215.69	317.19	58.82	86.50	$104
	Energy	energy																
	(cordwood)																	
7a	Wood	Wood pellets		10,000	TPY		159,800	85%	85%	188,000	8,500	11,059	138.41	215.69	253.75	58.82	69.20	$0
	pellets																	
7b	Thermal	Thermal		5	MMBTH	1,800	9,000	80%	80%	11,250	8,500	662	147.06	215.69	269.61	58.82	73.53	$167
	energy	energy																(40% MC)
	(pellets)																	
8	Thermal	Thermal		50	MMBTH	1,800	90,000	75%	75%	120,000	8,500	7,059	156.86	215.69	287.58	58.82	78.43	$104
	(green wood)	energy																
9	Thermal	Thermal		50	MMBTH	1,800	90,000	80%	80%	112,500	152,000	740,132	8.22	173.90	217.38	47.43	59.28	
	(oil)	energy									/gallon	gallons	gallons					
10	Thermal	Thermal		50	MMBTH	1,800	90,000	85%	85%	105,882	1,028	102,998	1.14	117.00	137.65	31.91	37.54	
	(natural gas)	energy									cubic ft	dtherms	dtherms					
11	CHP	Electricity		5	MW	7,200	368,496	75%	25%	491,328	8,500	28,902	156.86	215.69	287.58	58.82	78.43	$104
	(green wood)		Thermal	245,664	MMBtu													
			Electrical	122,832														
12	Gasifi er	Electricity		5	MW	7,200	317,669	75%	29%	423,559	8,500	24,915	156.86	215.69	287.58	58.82	78.43	$104
	(green wood)		Thermal	194,837	MMBtu													
			Electrical	122,832	MMBtu													
13	CHP (oil)	Electricity		5	MW	7,200	341,200	75%	27%	454,933	152,000	2,992,982	8.77	173.91	231.88	47.43	63.24	
			Thermal	218,368	MMBtu						per gallon	gallons	gallons					
			Electrical	122,832	MMBtu													
14	CHP (gas)	Electricity		5	MW	7,200	297,775	80%	33%	372,218	1,028	362,080	1.22	117.00	146.25	31.91	39.89	
			Thermal	174,943	MMBtu						cubic ft	dtherms	dtherms					
			Electrical	122,832	MMBtu													
15	Cellulosic	Cellulosic	Thermal	100	MMGPY		10,256,098	50%	41%	20,512,195	8,500	1,206,600	235.29	127.26	254.51	34.71	69.41	$70
	Ethanol	ethanol	Energy	1,846,098	MMBtu													
16	Bio-oil	Bio-oil	Thermal	15	MMGPY		1,542,667	65%	45%	2,373,333	8,500	139,608	181.00	118.63	182.50	32.35	49.77	$90
	& Bio-char		Energy	474,667	MMBtu													
17	Bio-		Thermal	15	MMGPY		1,542,667	65%	45%	2,373,333	8,500	139,608	181.00	118.63	182.50	32.35	49.77	$90
	products		Energy	474,667	MMBtu													
18	Cellulosic		Thermal	100	MMGPY		10,256,098	50%	41%	20,512,195	8,500	1,206,600	235.29	127.26	254.51	34.71	69.41	$70
	Ethanol		Energy	1,846,098	MMBtu													
	(gasifi cation)																	

APPENDIX 2-B: TECHNOLOGY PATHWAYS SUMMARY

Technology Pathways Summary Table

Orange = Formulas

Yellow = Typical Values assumed by BERC

Green = Calculated Values

Blue = Values taken from References

References (identified by cell)

Published Data by Biomass Power Plant: J5, K5

NREL: J7, K7, J11, K11, J13, K13

Published data by vendors: J15, K15, J18, K18, J21, K21, J23, K23, J26, K26, J28, K28,J30, K30, J32, K32, J35, K35, J38, K38, J41, K41, J46, K46, J48, K48, J50, K50

EERE, DOE: J44, K44

Calculated based on conversion of all carbon to carbon dioxide: P5, P7, P15, P18, P21, P23, P26, P32, P35, P44, P46, P48, P50

EIA: P8, P11, P13, P28, P30, P38, P41

Conversion Factors and Assumptions

1) 1 MWH = 3.412 MMBtu
2) High Heating Value of cellulosic ethanol = 84,100 (DOE)
3) High Heating Value of Bio-oil = 71,200 (DOE)
4) High Heating Value of Wood pellets (dry basis) = 17 MMBtu/ton (BERC)
5) High Heating Value of Wood chips (dry basis) = 17 MMBtu/ton (BERC)
6) High Heating Value of Coal = 10,506 Btu/lb (DOE)
7) High Heating Value of Natural Gas = 1,028 Btu/cubic ft.(DOE)
8) High Heating Value of #6 oil = 152,000 Btu/gallon(DOE)
9) 1 lb. Carbon = 3.6667 lbs CO_2
10) From Cell K12: co-firing with 20% biomass

NREL: Life Cycle Assessment of Coal Fired Power Production by Pamela L Spath & others at http://www.nrel.gov/docs/fy99osti/25119.pdf

EERE, DOE: Theoretical Ethanol Yield Calculator http://www1.eere.energy.gov/biomass/ethanol_yield_calculator.html

EIA: U S Energy Information Administration Independent Statistics and Analysis Voluntary Reporting of Green House Gases program (Fuel & energy Source Codes & emission coefficients) www.eia.doe.gov/oiaf/1605/coefficients.html

APPENDIX 2-C
AFFORDABLE PRICE OF BIOMASS—CALCULATION ASSUMPTIONS

Power		
Sale price of electricity	12.5	cents/kWh
Cost of biomass	33%	
Cost of biomass	0.04125	$/kWh
	3.02242E-06	$/Btu
	3.02	$/MMBtu
HHV of woodchips(dry)	8500	Btu/lb
HHV of woodchips(dry)	17	MMBtu/ton
HHV of woodchips(40 % MC)	10.2	MMBtu/ton
Affordable price of woodchips (40% MC)	31	$/ton
Thermal energy		
Oil price	3	$/gallon
heating value of oil	138,000	Btu/gallon
effciency	80%	
useful energy	110,400	Btu/gallon
cost of energy	27.17	$/MMBtu
Affordable price of woodchips as % of cost of oil	50%	
Affordable price of woodchips	13.59	$/MMBtu
effciency of woodchip boiler	75%	
available heat per ton	7.65	MMBtu/ton
affordable price of woodchips (40% MC)	104	$/ton
wood pellets (6% MC)		
HHV of wood pellets (6% MC)	15.98	MMBtu/ton
effi ciency	80%	
Available heat per ton	12.784	MMBtu/ton
Affordable price of wood pellets (6% MC) as % of cost of oil	75%	
Affordable price of wood pellets (6% MC)	20	$/MMBtu
Affordable price of wood pellets (6% MC)	261	$/ton
Woodchips to wood pellets		
effi ciency	85%	
wood pellets price	261	$/ton
woodchips MC	40%	
Wood pellets MC	6%	
woodchips required per ton of wood pellets	1.575	ton/ton
Affordable price of woodchips as % of cost of wood pellets	60%	
Affordable price of woodchips	85	$/ton

APPENDIX 3-A

REVIEW OF PREVIOUS STUDIES OF MASSACHUSETTS BIOMASS AVAILABILITY

In the past few years, the Massachusetts Sustainable Forest Bioenergy Initiative has funded two studies that address forest biomass availability in Massachusetts: Silvicultural and Ecological Considerations of Forest Biomass Harvesting in Massachusetts (Kelty, D'Amato, and Barten, 2008) and Biomass Availability Analysis—Five Counties of Western Massachusetts (Innovative Natural Resource Solutions (INRS), 2007). Here we review the components of these studies that focus on forest biomass, considering both their methodologies and results.

The general approach to forest biomass fuel used in these two studies is quite similar: both studies estimate net forest growth over an operable land base and equate this volume to biomass availability; thus, they assess how much wood could be harvested on an ongoing basis so that inventories do not decline below current levels. However, there are several important differences in the methods and details of implementing this approach and comparing their results with each other is not straightforward.

As will be seen in the following discussion, the data provided by Kelty et al. (2008) are presented in a manner that is most directly comparable to our own analysis. Kelty et al. (2008) provides two estimates of forest biomass availability on private lands to cover the wide range of potential responses by private landowners. The average of these two estimates is 750,000 green tons per year. When compared with our analysis, this average is consistent with our estimate of biomass supply at high biomass stumpage prices (the High-Price Biomass scenario). Kelty et al. (2008) is focused on forest growth and does not consider harvesting costs, energy prices, or general operational issues. This suggests that the biomass availability estimates provided by Kelty et al. would be reasonable estimates of supply only if bioenergy plants pay substantially higher prices for wood than in current markets.

Our adjustment of the INRS (2007) estimate to a statewide total suggests that biomass availability in Massachusetts would be about 1.4 million green tons per year. However, given the assumptions in this study, it is not clear how to adjust these estimates for sawtimber volumes and the split between private and public lands. Based on our review of this analysis, it would seem that the appropriate range for only biomass fuel on private lands would be about half of this volume, which suggests about 700,000 green tons, similar to the average of Kelty et al. (2008).

Review of "Silvicultural and Ecological Considerations of Forest Biomass Harvesting in Massachusetts"

The portion of this report that is focused on statewide biomass availability states that the question is: "what is the total annual sustainable harvest from Massachusetts forests (that is, the total annual harvest that would not exceed the total annual [net] forest growth)?" The report states that the intention was to assess the biomass levels that "exist in Massachusetts forests" on land that is "likely to be involved in timber harvesting." The report also provides a detailed analysis of biomass availability at the stand level, however, this analysis appears to be independent of the statewide analysis and have no influence on those results.

The methodology consists of three basic steps: 1) calculate average per-acre growth rates for timber stands in Massachusetts, with private and public lands evaluated separately; 2) identify the acreage available for harvesting; 3) adjust this total volume growth to separate sawtimber from other standing wood. These steps are described in more detail below and some key data are shown in Exhibit 3A-1.

Growth rates were developed on the basis of 50-year projections using the Forest Vegetation Simulator for the Northeast. The mean value of this time period was used as a measure of the growth rate in the future.

Two scenarios were established for private land areas because of the difficulty in predicting harvest activity among private landowners: one included all lands in size classes greater than or equal to 10 acres, while the other included only land greater than or equal to 100 acres. These two groups of private forest land areas were then reduced by 7% due to operational constraints such as terrain and wetland areas. Private lands were further reduced by 30% to adjust for landowners that were assumed would not be willing to harvest their lands for timber production.

Public forest land areas were reduced for operational constraints only. The reduction was 7%, the same as for private lands.

Total annual volumes of sustainable wood harvest were then calculated by multiplying growth per-acre growth rates times the number of acres available in each case. These data were then adjusted downward by 36% to account for timber that would likely be removed for sawtimber and not available to bioenergy facilities.

Results are presented i°n Exhibit 3A-1. "Sustainable" biomass availability was estimated to be about 500,000 green tons per year from public lands. For private lands, annual volumes ranged from 400,000-to-1.1 million green tons. Thus, the combined statewide total for biomass availability was estimated to range from 900,000-to-1.6 million green tons per year.

Exhibit 3A-1: Calculations for Biomass Availability Based on Kelty et al. (2008)

	Public	Private	
		≥ 100 acres	≥ 10 acres
Growth (dry tons/acre)	0.94	0.89	0.89
Growth (green tons/acre)	1.71	1.62	1.62
Net Land Area (acres)	465	379	1,073
Total Volume Growth (gt/yr)	795	614	1,736
Biomass Fuel Only (gt/yr)	509	393	1,111

Note: Data for dry tons and land areas taken directly from Exhibit 3-10 in Kelty et al. (2008). Data for green tons have been calculated assuming 45% moisture content.

Review of "Biomass Availability Analysis—Five Counties of Western Massachusetts"

The INRS (2007) report is comprehensive in its coverage of a wide range of sources of woody biomass. It is focused on the five western "core" counties of Massachusetts (Middlesex and Norfolk counties also are included as buffer counties, but are not reported separately from the buffer region). As above, we focus only on the portion of this study that addresses forest biomass.

This study considers forest biomass *growth* and forest *residues* separately. Forest biomass growth is estimated using net growth and removals from FIA data along with an adjustment factor for the growth of tops and branches. Net growth less removals results in estimated annual growth of 1.9 million green tons for western Massachusetts This volume is then reduced by half: "because of landowner constraints, access issues, economic availability, nutrient concerns and the need to harvest less than growth to address landscape-level forest sustainability concerns, INRS suggest that half of this wood be considered actually 'available' to the marketplace" This leaves a total of 960,000 green tons per year of biomass availability. An additional 110,000 green tons of forest residue are estimated to be available in this region (based on TPO data), resulting in an annual total of 1.1 million green tons.

These estimates do not consider the share of wood that might be used for sawtimber. The INRS report indicates that their estimate likely overstates the availability of forest biomass for this reason and others: "In practical terms, it is highly unlikely that this volume of wood could be harvested in an economic or environmentally responsible manner to supply biomass fuel. Further, some of this wood is sawlogs or other high-value material, and as such would be sent to other markets."

We have attempted to put these estimates on a basis that is comparable to the Kelty et al. (2008) analysis by adjusting them to the state level (growth and forest residues are not considered separately because of the small residue share). There are several alternatives for increasing these data from the western region to the state total, but it is not obvious which method would be most appropriate. Relative measures of timberland area, timber inventory, and growth-drain ratios result in expansion factors ranging from 20% to 40%. Thus, the total for biomass availability would be increased to 1.3-to-1.5 million green tons per year.

These estimates are close to the high end of the range (1.6 million green tons) provided by the Kelty et al. (2008). However, it is unclear how to adjust these estimates for potential sawtimber volumes. Kelty et al. (2008) project total net growth and then subtract the sawtimber component, whereas the INRS report projects "net growth less removals" so the growth estimates already partially reflect an adjustment for sawtimber. In addition, for purposes of comparison, it would be useful to separate the INRS volumes by private and public ownerships; however, the analysis reduces net growth on all forest lands by 50% and there appear to be no explicit assumptions regarding the mix of wood available from the two ownerships.

APPENDIX 3-B

LOGGING RESIDUE DATA AND ESTIMATION

Although estimation of this supply would seem to be straightforward, problems with logging residue data make it difficult to estimate both the total volume of residues that are generated as well as the share that is recoverable. Some of these problems are general conceptual issues, while others are specific to the Northeast and/or Massachusetts. An important issue relates simply to the definition of logging residues. Logging residues are not defined by the parts of a tree, but by what is left behind in the forest after a site has been logged. In addition to the obvious candidates for unused material after felling, such as crowns and branches, trees that have been killed or damaged during a logging operation are considered to be part of logging residues.[1] Thus, this becomes a difficult empirical issue because harvesting is dynamic and logging residues will change and evolve with technology, timber demand, and relative costs and prices. Once utilized, the material no longer conforms to the definition of logging residues and this can be a source of confusion.

Another important problem with logging residue data is that the parameters used to derive these estimates are from mill and timber utilization studies that are dated. The primary source of logging residue data in most studies is the Timber Products Output (TPO) reports from the U.S. Forest Service. These reports contain data on both softwood and hardwood residues and are disaggregated to the county level.[2] In the Northeast, these studies were last conducted in 1985, and thus do not reflect current utilization standards, prices, costs, and technologies. In addition, the calculation of logging residues requires a combination of surveys, each with its own problems and sampling errors. These problems are likely to be more serious in small states (where interstate trade is important) because wood flows and sourcing patterns can change substantially over time.

As it turns out, the logging residue data reported by TPO for Massachusetts could not be used because the on-line program generates the data incorrectly.[3] In order to generate logging residues that are consistent with the TPO methodology, we have applied the timber utilization matrices underlying the TPO estimates to their estimates of roundwood harvests.

According to the production data from the TPO reports, industrial roundwood production in Massachusetts is comprised of essentially two "products," sawlogs and pulpwood. ("Other industrial products" is a third category and accounts for 1% of the industrial roundwood total).[4] The volume of logging residues generated per unit of roundwood production is shown in Exhibit 3A-1. Logging residues from softwood harvests are less than for hardwoods because of differences in tree geometry and differences in end-use markets and products. Logging residues for pulpwood are less than for sawlogs because of the ability to utilize a higher proportion of the main stem.

The TPO data for Massachusetts in 2006 indicate that sawlogs accounted for 87% of the industrial roundwood production, and that softwood accounted for 60% of the sawlog production. Applying the coefficients in Exhibit 3B-1 to these data suggest that logging residues totaled 4.27 million cubic feet in 2006, or 50% of roundwood production. This implies that approximately 128,000 green tons of logging residues were generated in 2006 (using a conversion of 30 green tons per thousand cubic feet).

Exhibit 3B-1: Logging Residue Generation in Massachusetts By Product and Species Group
(cubic feet of logging residues per cubic foot of roundwood)

	Softwood	Hardwood
Sawlogs	0.43	0.67
Pulpwood	0.36	0.56

Source: Personal communication with USFS.

Importantly, these data appropriately measure only unutilized residues—wood left behind after a logging operation—and thus would be the correct measure of the total volume of residues that could be available for biomass. However, as noted earlier, a closer look at these data suggests that a significant share of this material can be attributed to breakage or residual stand damage, and thus could not be transported to a landing during a harvesting operation. For this reason, it is often assumed that 50% of "logging residues" are recoverable. Using this assumption, 64,000 green tons of logging residues would have been available for biomass supply in 2006.

Concerns about the TPO data and with implementing those estimates in a manner that is consistent with our projection and harvesting methodology have led us to a second approach: estimation of logging residue generation by calculating the volume of tops and limbs associated with harvesting trees of varying diameter

[1] According to Forest Resources of the United States, 2002 (Smith et al.), logging residues are defined as: "The unused portions of growing stock and non-growing stock trees cut or killed by logging and left in the woods." This includes material that is sound enough to chip (and excludes rotten wood), downed dead trees, and downed cull trees. Material that has been badly damaged during logging but is still standing should be included in logging residues; however, the definitions are confusing in this regard.

[2] The reports are available on-line (www.fia.fs.fed.us/tools-data/other/) and can be accessed on the National Renewable Energy Laboratory website.

[3] The on-line TPO program reports that 8.451 million cubic feet of industrial roundwood products were produced in Massachusetts in 2006. The same number is reported as the total for "Logging Residues" and also for "Mill Residues."

[4] There is also a large volume of fuelwood production; in fact, the volume is substantially higher than industrial roundwood production. However, the TPO methodology assumes that residential fuelwood harvests do not contribute to logging residues.

classes. From a biomass perspective, this approach provides a more useful estimate of "logging residues" since this material has a much better chance of being delivered to a landing at a reasonable cost using whole-tree harvesting methods.[5]

Exhibit 3B-2 shows the average volume of tops and limbs as a share of the merchantable tree volume in the standing inventory of live trees in Massachusetts. These data suggest that for all species combined, reasonable estimates of "logging residues" generated would be about 22%, on average, for sawtimber harvests and 35% for pulpwood. Thus, using the same data on industrial roundwood production as above (from TPO for 2006), logging residues would have been about 2.0 million cubic feet, or 60,000 green tons. Given that this material could be moved to a landing more easily because it consists strictly of tops and limbs, the recovery rate of this material for biomass fuel use could be considerably higher than 50%.

Exhibit 3B-2: Volume of Tops and Limbs as a Share of Merchantable Tree Volume
Based on Massachusetts Inventory Data, 2008

DBH, inches	Share
5.0–6.9	38%
7.0–8.9	31%
9.0–10.9	27%
11.0–12.9	24%
13.0–14.9	22%
15.0–16.9	21%
17.0–18.9	19%
19.0–20.9	18%
21.0–22.9	18%
23.0–24.9	17%

Source: Based on USFS, FIA data. DBH is tree diameter measured at breast height (4.5 feet above ground level).

5 One shortcoming of this approach is that it is not possible to estimate how much of this topwood and limbwood may already be being utilized due to differing utilization standards for products, or for harvests of firewood.

APPENDIX 3-C

FIREWOOD DATA

Fuelwood is by far the largest market for timber cut in Massachusetts, but fuelwood data are poor because the market is unregulated with large numbers of consumers and producers, and there is a large personal use sector where consumers cut their own wood. The FCPs show some data on fuelwood harvests, but these numbers are small and only pertain to volumes that are associated with larger-scale commercial-based harvesting. The large majority of fuelwood cut in Massachusetts is not registered in these plans.

The Timber Product Output reports provide one estimate of fuelwood production in Massachusetts; however, these data are derived from U.S. Census data rather than collected directly from U.S. Forest Service surveys (the source of other TPO data). TPO data indicate that fuelwood production in Massachusetts in 2006 was 41.3 million cubic feet (517,000 cords or 1.3 million green tons), which would suggest that it would have accounted for about 83% of the timber harvest in Massachusetts (see Exhibit 3C-1.) According to this report, virtually all of the fuelwood comes from non-growing stock sources, which includes cull trees (rough and rotten), dead trees, tops and stumps of growing stock trees, and non-forestland sources of trees such as yard trees.

Exhibit 3C-1: Fuelwood Production in Massachusetts, 2006
Million Cubic Feet

	Industrial	Fuelwood	Total	Fuelwood (cords)
Growing Stock	7.0	1.2	8.2	15
Non-Growing Stock	1.5	40.1	41.6	502
Total	8.5	41.3	49.8	517

Source: TPO Reports (USDA, FS).

Unlike the data on industrial roundwood products, the data on fuelwood have not been collected by the USFS since some time prior to 1980. Since then, the data have been collected by Energy Information Administration as part of their Residential Energy Consumption Survey. These data are surveyed at the Census division level and allocated to individual states on the basis of the total number of housing units. In the case of Massachusetts, this methodology clearly overstates fuelwood consumption since Massachusetts accounts for about half of the housing units in New England. For example, in 2007, New England consumption was estimated to be about 927,000 cords, and 439,000 cords were allocated to Massachusetts. Prior to the time when this methodology was adopted, Massachusetts share of New England fuelwood consumption was only 35% in 1975 (and jumped to 49% when housing units were used as the basis of the allocation).

An important question in assessing biomass supplies in Massachusetts is how the residential fuelwood sector might interface with an expanded harvest of forest biomass fuel. Fuelwood is typically harvested in relatively small volumes and on small areas, often by landowners cutting for their own use. We have assumed that forest biomass harvests are unlikely to be integrated with harvests of residential fuelwood due to: 1) the number of acres cut in a typical fuelwood harvest; 2) the volume of logging residue left behind on each acre; and 3) the type of equipment used in these logging operations.

APPENDIX 3-D

A CLOSER LOOK AT BIOMASS POTENTIAL IN SOUTHERN NEW HAMPSHIRE

The analysis of inventories, industry location, and landowner attitudes in this report suggests that the border counties in New Hampshire, Vermont, and New York hold the most potential for increasing supplies of forest biomass. The New Hampshire border zone is the largest of these areas and the one with perhaps the best data. Here we look more closely at recent historical harvests (New Hampshire Report of Cut, 2008) and prices trends (New Hampshire Timberland Owner's Association, Timber Crier) in New Hampshire to see if there are any patterns that suggest than an expansion of timber production looks likely.

Timber Harvest Trends

In New Hampshire, the sawlog harvest declined from 2000 to 2006, with most of the decrease occurring by 2003. This is somewhat surprising given the strength of the housing market during this period. Part of this decline was offset by an increase in pulpwood and fuelwood harvest. Whole-tree chip production was fairly stable over these seven years, averaging about 800,000 green tons per year, equivalent to about 25% of New Hampshire's roundwood harvest.[6]

The harvest in the three counties of southern New Hampshire has been fairly stable as a share of the total cut in the state, fluctuating in the range of 20%–23% during 2000–2006. Similar to overall state trends, sawlog production declined (from 400,000 green tons in 2000 to 300,000 green tons in 2006), while pulpwood rose and whole-tree chip production remained steady at about 230,000 green tons.

Several aspects of these trends have implications for our analysis: 1) in spite of rising timber inventories in New Hampshire, recent harvest levels have been declining; 2) the southern counties share of the harvest has been stable; 3) in the southern counties, whole-tree harvests have been stable as a share of the overall harvest.

Overall trends do not show New Hampshire as a state that is expanding its forest products industries and its harvest levels. In general, this is not a positive trend for a bioenergy industry that is thought to have it biggest advantage when its raw material comes from integrated harvests that depend on other commercial products. Also, the southern share of state harvests has been stable: if the share were rising, one might have some evidence that the region has some competitive advantage, possibly in the area of wood supply.7

[6] Similar to Massachusetts, harvesting of fuelwood does not need to be reported if the volume is considered to be small and for personal use. For New Hampshire, this maximum volume is set at 20 cords.

[7] When sawlog production declines, the production and availability of mill residues will also decline (assuming sawlogs are milled in the "home" market). This is another factor that has negative consequences for biomass fuel supply.

The Relationship of Timber Harvest to Inventory Levels

A key metric that is often used to measure tightness in the timber market is the ratio of timber harvest to timber inventory (FIA data). We have compiled these estimates for the three New Hampshire regions to see if they provide any additional information about harvest potential (see Exhibit 3D-1). The cut-to-inventory ratio statewide is 1.1% (as noted in the table, the harvest data do not include residential fuelwood and logging residues which would likely move this ratio closer to 1.5%). These ratios decline as one moves from north to south: the ratio is 1.3% for the northern counties, 1.1% for the central counties, and 0.9% for the southern counties. As might be expected, timber inventories are growing more slowly in the central and northern areas. In fact, harvesting in the north has outpaced growth and timber inventories on private lands have declined an average of 500,000 green tons per year according to FIA estimates. These higher rates of harvesting in the north are also reflected in stocking levels which we estimate to be only 50 green tons per acre on private lands in the north, compared to 66 tons/acre in the central region, and 75 tons/acre in the south.

These data seem to suggest that if there are opportunities for expansion in New Hampshire, they may lie in the south. However, one cannot draw this conclusion on the basis of cut/inventory ratios or stocking levels alone unless the land in the inventory is similar and managed the same way. For example, it is common to see high cut/inventory ratios for industrial land ownerships (there are forest industry lands in northern New Hampshire) and lower cut/inventory ratios on non-industrial private lands where timber production may not be the most important objective of landowners.

Exhibit 3D-1: Harvest Ratios in New Hampshire
000 Green Tons and Percent

	Harvest	Cut/Inv
New Hampshire	3,238	1.1%
North	1,731	1.3%
Central	809	1.1%
South	698	0.9%

Notes: Harvest data is the average for 2000–2006 and includes sawlogs, pulpwood, fuelwood, and whole-tree chips. "Cut/Inv" is the ratio of harvest to growing stock on private and public timberland. Harvest data exclude residential fuelwood and logging residues and thus understate timber removals.

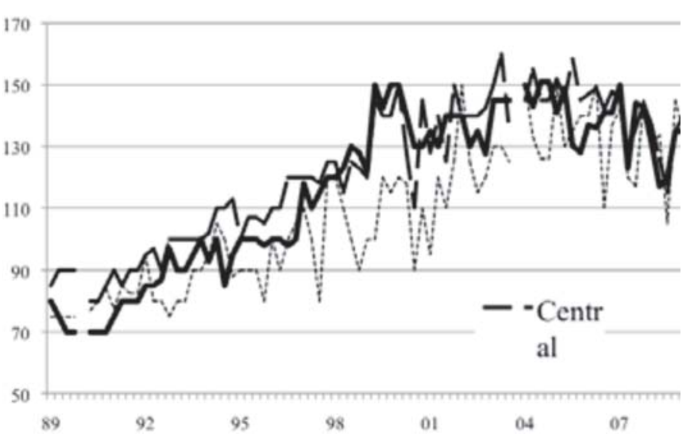
In spite of low cut/inventory ratios and expanding timber inventories in the southern counties of New Hampshire, the harvesting data have shown the south's position as a timber producer has been relatively stable. The southern counties are not growing in an absolute sense, nor have harvest levels increased relative to the central or northern areas. Importantly, we have also seen that whole-tree harvesting is already prevalent in southern New Hampshire. Thus, opportunities for expansion as part of integrated operations might be more limited than in other border zones where whole-tree harvesting is much less common.

Prices and Pockets of Opportunity

The final measure we consider—perhaps the single best indicator—is relative pricing. In a market in equilibrium, prices will track together. If prices deviate from the overall trend, particularly if they are drop lower at times, this may be due to weaker demand and might be an indicator that more timber can be harvested with the region remaining competitive. In Exhibit 3D-2, we have compared white pine sawtimber stumpage prices for the three regions of New Hampshire. We selected white pine because it accounts for about 50% of the sawtimber harvest and is widely distributed through the state (spruce/fir is the next largest species group with 13%, but it is almost entirely produced in the northern zone). We selected sawtimber because: 1) biomass is generally expected to be a follower of higher-valued commercial harvest; and, 2) biomass stumpage prices can easily diverge within regions because they are such a small share of total delivered costs.

Prices for white pine sawtimber stumpage in southern New Hampshire fall right in line with those in the central region suggesting that the buyers of wood can access both areas on an equal footing; hence the south would not appear ripe for greater expansion relative to other New Hampshire regions. The north is a bit more erratic, dropping below the southern price at times and for an extended period in 1997–2000. The data do not suggest any obvious gaps in the south that would be an incentive to build new capacity; in fact, the data suggest that such opportunities may have existed in the north during the 1990s. Although forests in the north have been cut more intensively than elsewhere in the state, prices have not moved higher suggesting that overall pressures on the resource remain similar in the three regions when ownership, attitudes, management objectives and other variables are taken into account.

Exhibit 3D-2: White Pine Sawtimber Stumpage Prices in New Hampshire
Dollars per 1,000 board feet International log rule



Source: New Hampshire Timberland Owner's Association, Timber Crier, various issues: mid-range stumpage prices.

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APPENDIX 4–A

ECOLOGY OF DEAD WOOD IN THE NORTHEAST

Alexander M. Evans and Matthew J. Kelty

TABLE OF CONTENTS

1. INTRODUCTION

Although dead wood and decaying trees have historically had little commercial value, they do have substantial ecological value. This paper reviews the scientific literature to provide the background necessary to craft recommendations about the amount and type of dead wood that should be retained in the forest types of the northeastern U.S. Establishing the ecological requirements for dead wood and other previously low-value material is important because of the increased interest in this material for energy and fuel. More intensive extraction of biomass from the forest may impinge on the forest's ability to support wildlife, provide clean water, sequester carbon, and regenerate a diverse suite of plants.

This background paper covers the topics of dead wood, soil compaction, nutrient conservation, and wildlife habitat in temperate forests generally as well as in specific forest types of the Northeast. Complex issues related to carbon storage in forests and the climate impacts of using forest material for energy and fuel are very important and deserve an in-depth investigation beyond the scope of this paper. Similarly, this paper will not discuss the state of biomass harvesting in the U.S. (Evans 2008, Evans and Finkral 2009) or existing biomass harvesting guidelines (Evans and Perschel 2009) which have been addressed in other recent publications.

The goal of this background paper is to provide a concise summary that can inform discussions about biomass harvesting standards in the Northeast. However, it is important to note that this document makes no suggestions about how a biomass harvest should be conducted or what should be left in the forest after a harvest. Rather we have attempted to lay out the basic science on which recommendations can be built.

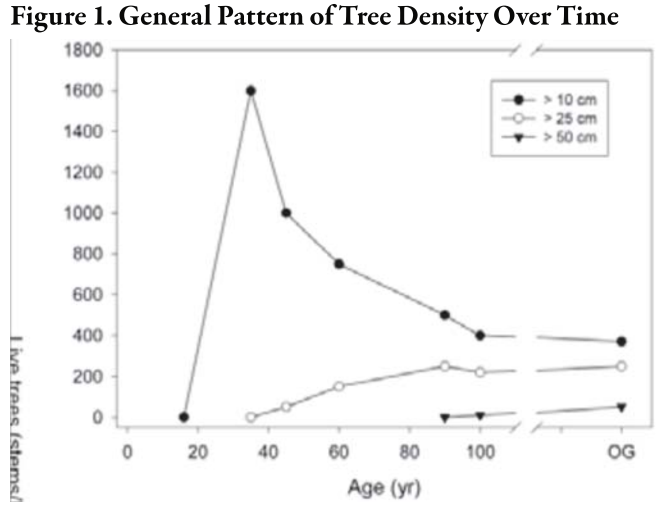
2. ECOLOGY OF DEAD WOOD IN THE NORTHEAST

2a. Dead Wood and Stand Development

Dead wood is important not only in terms of total volume or mass in a stand, but also in terms of piece size—usually measured as diameter at breast height (DBH) for snags (and for live trees) or diameter of the large end for down woody material (DWM). Large-diameter snags or down logs are important habitat for numerous animal species, persist for long periods, store nutrients, and provide substrate for seed germination.

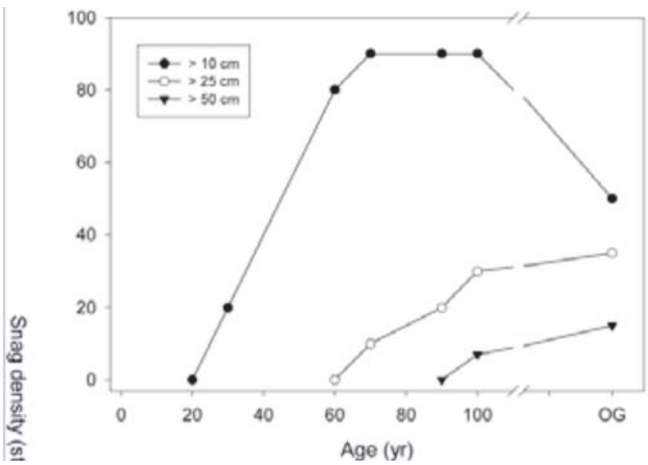
The process of dead wood accumulation in a forest stand consists of the shift from live tree to snag to DWM unless a disturbance has felled live trees, shifting them directly to DWM. The graphs below (Figures 1, 2, and 3) show the general pattern of the production of dead wood in total amount and size. The data in these graphs are taken from research in northern hardwood forests (Gore and Patterson 1986, Goodburn and Lorimer 1998, Hale et al. 1999, McGee et al. 1999, Nyland et al. 2000). The 4 in (10 cm) diameter size is within the range of the minimum size used in most coarse woody material (CWM) inventories. Fine woody material (FWM) refers to smaller-sized dead material. The graphs depict the patterns for a stand that had been harvested as a conventional clearcut, leaving a large amount of small woody material (nearly all <10 in (25 cm) diameter), but no trees >4 in (>10 cm) DBH and no snags. The pattern is shown from just after the clearcut (age 0)–age 100 years, and in the old-growth condition.

Figure 1. General Pattern of Tree Density Over Time



The young stand produces large numbers of trees (~600 stems/ac or ~1500 stems/ha) at age 30, and the intense competition among these trees causes mortality of smaller stems, which creates an increasing number of small snags (Figure 2). Trees begin to grow into 10 in (25 cm) DBH size by age 40, and trees of this size begin to dominate the stand by age 80. Snags of the 10 in (25 cm) DBH size begin to appear at age 60 as mortality of larger trees occur. Large live trees (>20 in or >50 cm) begin to appear at age 90—100, with snags of that size as well.

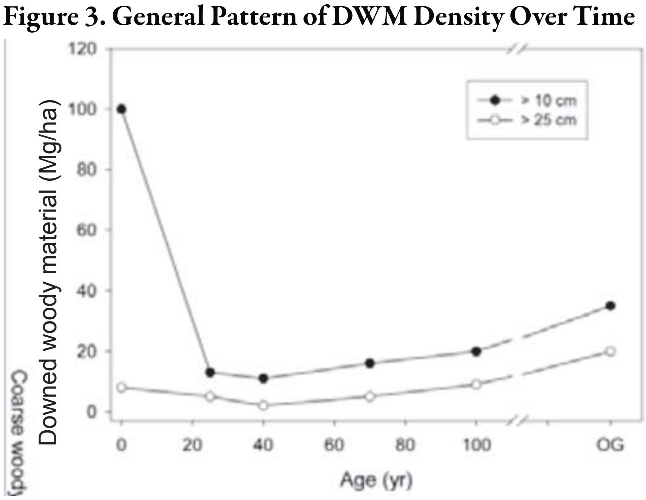
Figure 2. General Pattern of Snag Density Over Time



The large amount of DWM present just after the clearcut (which consists mostly of pieces <10 in (<25 cm) diameter) decomposes rapidly in the first 25 years and continues to decline in mass to age 40. From age 40—100 years, DWM increases as small snags fall, and then larger snags begin to contribute to DWM that include pieces >10 in (>25 cm) diameter. Very few large (> 20 in or >50 cm) pieces of DWM are produced. Large DWM often results from wind or other disturbances that fell large trees in the old-growth stage. Thus, large DWM tends to accumulate periodically from these disturbance pulses; whereas small DWM accumulates in a more predictable pattern in earlier stages of stand development.

This process produces the U-shaped pattern that is often described with a dearth of DWM in the intermediate ages (Figure 3). This pattern shows the importance of retaining large live trees and large snags at the time of harvest; they will contribute large DWM to the forest floor throughout the development of the stand.

Figure 3. General Pattern of DWM Density Over Time



2b. Wildlife and Biodiversity

Dead wood is a central element of wildlife habitat in forests (Freedman et al. 1996). Many forest floor vertebrates have benefited or depended on DWM (Butts and McComb 2000). In the southeastern U.S., more than 55 mammal species, over 20 bird species, and many reptiles and amphibian species were relying on dead wood (Lanham and Guynn 1996, Loeb 1996, Whiles and Grubaugh 1996) with similar numbers for the forests of the Pacific Northwest (Carey and Johnson 1995, McComb 2003). In New England, De Graaf and colleagues (1992) catalogued at least 40 species that rely on DWM.

Some examples of relationships between animals and DWM in the Northeast include a study showing that low densities of highly decayed logs (less than one highly decayed log/ha) had a negative impact on red-back voles (*Clethrionomys gapperi*) in a northern hardwood forest in New Brunswick, Canada (Bowman et al. 2000). DWM retention increased spotted salamander (*Ambystoma maculatum*) populations in a Maine study (Patrick et al. 2006). While DWM is important habitat for red-backed voles in Maine, it did not effect populations at volumes as low as 543 ft^3/ac (38 m^3/ha; McCay and Komoroski 2004). The quantity of DWM had no effect on white-footed mice (*Peromyscus leucopus*) abundance in an Appalachian study, but at the micro-site scale, mice were more often located near DWM (Marcus et al. 2002). Similarly, shrew (*Tupaia* sp.) showed minimal or no response to drastic decreases in the abundance of large logs in managed loblolly pine (*Pinus taeda*) forests of the southeastern coastal plain (McCay and Komoroski 2004) .

In aquatic environments, DWM provided crucial refuge from predation (Angermeier and Karr 1984, Everett and Ruiz 1993). Logs that fell in the water formed a critical component of aquatic habitat by ponding water, aerating streams, and storing sediments (Gurnell et al. 1995, Sass 2009). In fact, removal of large woody material from streams and rivers had an overwhelming and detrimental effect on salmonids (Mellina and Hinch 2009).

DWM is a key element in maintaining habitat for saproxylic insects (Grove 2002). For example, some specialist litter-dwelling fauna that depend on DWM appear to have been extirpated from some managed forests (Kappes et al. 2009). A study from Ontario suggests that overall insect abundance was not correlated with the volume of DWM, though abundance of the fungivorous insect guild was positively related to the volume of DWM (Vanderwel et al. 2006b). Extensive removal of DWM could reduce species richness of ground-active beetles at a local scale (Gunnarsson et al. 2004). More generally, a minimum of 286 ft^3/ac (20 m^3/ha) of DWM has been suggested to protect litter-dwelling fauna in Europe (Kappes et al. 2009).

Dead logs served as a seedbed for tree and plant species (McGee 2001, Weaver et al. 2009). Slash could be beneficial to seedling regeneration after harvest (Grisez, McInnis, and Roberts 1994). Fungi, mosses, and liverworts depended on dead wood for nutrients and moisture, and in turn, many trees were reliant on mutualistic relationships with ectomycorrhizal fungi (Hagan and Grove 1999, Åström et al. 2005). In general, small trees and branches hosted more species of fungus per volume unit than larger trees and logs; however larger dead logs may be necessary to ensure the survival of specialized fungus species such as heart-rot agents (Kruys and Jonsson 1999, Bate et al. 2004).

2C. SOIL PRODUCTIVITY

DWM plays an important physical role in forests and riparian systems. DWM added to the erosion protection by reducing overland flow (McIver and Starr 2001, Jia-bing et al. 2005). DWM also had substantial water-holding capacity (Fraver et al. 2002). DWM in riparian systems provided sites for vegetation colonization, forest island growth and coalescence, and forest floodplain development (Fetherston et al. 1995).

In many ecosystems, CWM decomposed much more slowly than foliage and FWM, making it a long-term source of nutrients (Harmon et al. 1986, Johnson and Curtis 2001, Greenberg 2002, Mahendrappa et al. 2006). DWM decomposed through physical breakdown and biological decomposition (Harmon et al. 1986). The diameter of each piece of DWM, temperature of the site, amount of precipitation, and tree species all influenced the rate of DWM decomposition (Zell et al. 2009). In general, conifers decayed more slowly than deciduous species (Zell et al. 2009). Other factors that encouraged decomposition included warmer temperatures, rainfall between 43 and 51 in/year (1100 and 1300 mm/year), and small-sized pieces (Zell et al. 2009). While there is great variation across ecosystems and individual pieces of DWM, log fragmentation generally appears to occur over 25–85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007).

In some ecosystems, DWM represents a large pool of nutrients and is an important contributor to soil organic material (Graham and Cromack Jr. 1982, Harvey et al. 1987). However, review of DWM in Northern coniferous forests suggested that DWM may play a small role in nutrient cycling in those forests (Laiho and Prescott 2004). The same review showed that DWM contributes less than 10 percent of the nutrients (Nitrogen (N), Phosphorus (P), Potassium (K), Calcium (Ca), and Magnesium (Mg) returned in aboveground litter annually, and approximately five percent of the N and P released from decomposing litter or soil annually (Laiho and Prescott 2004). Although DWM is often low in N itself, N fixation in DWM was an important source of this limiting nutrient in both terrestrial and aquatic ecosystems (Harmon et al. 1986). There was a wide range of non-symbiotic N fixation, but temperate forests received average input of about 1.8–2.7 lb/ac/yr (2–3 kg/ha/year) of N (Roskoski 1980, Yowhan Son 2001).

A review of scientific data suggests that when both sensitive sites (including low-nutrient) and clearcutting with whole-tree removal are avoided, then nutrient capital can be protected (see also Hacker 2005). However, there is no scientific consensus on this point because of the range of treatments and experimental sites (Grigal 2000). It is important to emphasize that the impact on soil nutrients is site dependent. Low-nutrient sites are much more likely to be damaged by intensive biomass removal than sites with great nutrient capital or more rapid nutrient inputs. A report on impacts of biomass harvesting from Massachusetts suggested that with partial removals (i.e., a combination of crown thinning and low thinning that removes all small trees for biomass and generates from 9–25 dry t/ac or 20–56 Mg/ha) stocks of Ca, the nutrient of greatest concern, could be replenished in 71 years (Kelty et al. 2008). The Massachusetts study was based on previous research with similar results from Connecticut (Tritton et al. 1987, Hornbeck et al. 1990). Leaching, particularly of Ca due to acidic precipitation, can reduce the nutrients available to forests even without harvests (Pierce et al. 1993). However, the Ca-P mineral apatite may provide more sustainable supplies of Ca to forests growing in young soils formed in granitoid parent materials (Yanai et al. 2005).

15 years of data from Hubbard Brook Ecosystem Study indicate that a whole-tree clear cut did not result in the depletion of exchangeable Ca pools (Campbell et al. 2007). The Environmental Impact Statement from the White Mountain National Forest (2005 p. 3–19) demonstrated the variation in Ca removed by treatment and forest type, though even whole-tree clear cut was estimated to remove only four percent of the total Ca pool. A study of an aspen/mixed-hardwood forest showed that even with a clearcut system, Ca stocks would be replenished in 54 years (Boyle et al. 1973). Minnesota's biomass guidelines present data that showed soil nutrient capital was replenished in less than 50 years even under a whole-tree harvesting scenario (Grigal 2004, MFRC 2007). Whole-tree clearcutting (or whole-tree thinning, e.g., Nord-Larsen 2002) did not greatly reduced amounts of soil carbon or N in some studies (Hornbeck et al. 1986, Hendrickson 1988, Huntington and Ryan 1990, Lynch and Corbett 1991, Olsson et al. 1996, Johnson and Todd 1998). Lack of significant reduction in carbon and N may be due to soil mixing by harvesting equipment (Huntington and Ryan 1990). However, intensive cutting, such as clearcutting with whole-tree removal, can result in significant nutrient losses (Hendrickson 1988, Federer et al. 1989, Hornbeck et

al. 1990, Martin et al. 2000, Watmough and Dillon 2003)—in one case, 13 percent of Ca site capital (Tritton et al. 1987).

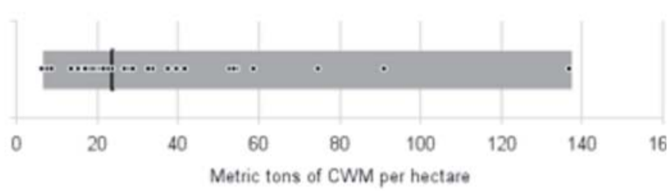
Low-impact logging techniques that reduce soil disturbance can help protect nutrient capital (Hallett and Hornbeck 2000). Harvesting during the winter after leaf fall can reduce nutrient loss from 10–20 percent (Boyle et al. 1973, Hallett and Hornbeck 2000). Alternatively, if logging occurs during spring or summer, leaving tree tops on site would aid in nutrient conservation. Nordic countries have demonstrated that leaving cut trees on the ground in the harvest area until their needles have dropped (one growing season) can also reduce nutrient loss (Nord-Larsen 2002, Richardson et al. 2002).

2D. QUANTITIES OF DEAD WOOD

Site productivity and the rate of decomposition helped determine the amount of dead wood in a given stand (Campbell and Laroque 2007, Brin et al. 2008). As mentioned above, DWM decomposition varies greatly but generally occurs over 25–85 years in the U.S. (Harmon et al. 1986, Ganjegunte et al. 2004, Campbell and Laroque 2007). All mortality agents including wind, ice, fire, drought, disease, insects, competition, and senescence create dead wood (Jia-bing et al. 2005). Of course, these mortality agents often act synergistically.

A review of 21 reports of quantitative measures of DWM in Eastern forest types shows great variability across forest types and stand development stages (Roskoski 1980, Gore and Patterson 1986, Mattson et al. 1987, McCarthy and Bailey 1994, Duvall and Grigal 1999, Idol et al. 2001, Currie and Nadelhoffer 2002). The reports ranged from 3–61 t/ac (7 to 137 Mg/ha) with a median of 11 t/ac (24 Mg/ha) and a mean of 15 t/ac (33 Mg/ha; see Figure 4). Measurements of old forests (>80 years old), had a median of 11 t/ac (24 Mg/ha) and a mean of 13 t/ac (29 Mg/ha) in DWM.

Figure 4. Distribution of DWM Measured in Eastern Forests



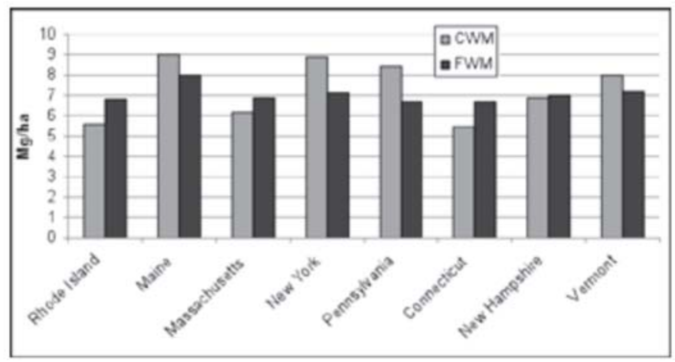
The gray bar shows the range of DWM measurement, the black line shows the median value, and each dot represents one measurement of DWM.

In contrast, a study of U.S. Forest Service inventory plots found a mean of 3.7 t/ac (8.3 Mg/ha) and a median of 2.9 t/ac (6.5 Mg/ha) of DWM across 229 plots in the Northeast (Chojnacky et al. 2004 see Figure 2). This low level of DWM across the landscape may be due to widespread clearcutting in the 1880-1930 period.

Figure 5. U.S. Forest Service Inventory Estimates of Deadwood Data from Chojnacky et al. 2004

3. Research by Forest Type

The following section uses the best available scientific literature to examine the dead wood dynamics of specific forest types in the Northeast. This region encompasses three ecological provinces including Northeastern mixed forest, Adirondack-New England mixed forest-coniferous forest, and Eastern broadleaf forest (McNab et al. 2007). Major forest types in the region are white/red/jack pine (*Pinus* sp.), spruce-fir (*Picea* sp. - *Abies* sp.),



oak-hickory (*Quercus* sp. - *Carya* sp.) or transitional hardwood forests, and northern hardwood forests (Eyre 1980).

The average year round temperature in the Northeast is 46°F (8°C). Winter temperatures average 24°F (-4.3°C) while summer temperatures average 67°F (19.6°C; National Climate Data Center 2008). The prevailing wind direction, from west-to-east, creates a continental climate except for coastal areas moderated by the Atlantic Ocean (Barrett 1980). On average, the region receives 41 in (104 cm) of precipitation which is evenly distributed throughout the year (National Climate Data Center 2008). Elevations range from sea level to mountain tops above 5,300 ft (1,600 m), but much of the region is set on upland plateaus between 500 ft and 1500 ft (150 and 460 m; Barrett 1980). Glaciation created young soils which vary considerably across small spatial scales (Barrett 1980).

Much of the southern portion of Northeastern forests was cleared for agriculture in the early 19th century, leaving less than one percent of the forest cover in an old-growth condition (Cogbill et al. 2002). Currently much of the region is comprised of second- or third-growth forest that has yet to reach late seral stages (Irland 1999). There are about 80 million ac (32 million ha) of timberlands (areas where commercial timber could be produced) and about 4 million ac (1.6 million ha) of reserved forest where harvests are not permitted (Alvarez 2007). Approximately 1,272 million ft3 (36 million m^3) of wood are harvested annually out of 3,157 million ft3 (89 million m^3) of net tree growth (Alvarez 2007).

3A. SPRUCE-FIR FORESTS

Spruce-fir forests dominate the inland areas of Maine as well as the mountain tops northernmost portions of New York, New Hampshire, and Vermont. These forests have cold temperatures and relatively coarse, acidic soils (Barrett 1980). Dead wood is important in spruce-fir ecosystems. For example, in Maine (the state with the greatest area of spruce-fir forests in the Northeast), DWM, snags, and cavity trees are important habitat for 20 percent of bird, 50 percent of mammal, 44 percent of amphibian, and 58 percent of reptile species found there (Flatebo et al. 1999). Animals that rely on DWM in spruce-fir forests include pine marten (Martes americana atrata) (Kyle and Strobeck 2003) and may include some saproxylic vertebrates (Majka and Pollock 2006).

In 2001, researchers found the volume of down dead wood in Maine's spruce-fir forest to be 530 ft^3/ac (37 m^3/ha) or 3.4 t/ac (7.5 Mg/ha) (Heath and Chojnacky 2001, Table 36). While the average was 3.4 t/ac (7.5 Mg/ha) non-industrial private lands only had 3 t/ac, public lands had 3.3 t/ac, while industrial lands had 3.7 t/ac (Heath and Chojnacky 2001, Table 37). The quadratic-mean, large-end diameter of down wood in Maine's spruce fir-forests measured 6.7 in (17 cm; Heath and Chojnacky 2001). The number of dead trees in nine red spruce-balsam fir forests ranged from 85–232/ ac (210–574/ ha) or from 11–43 percent of the basal area (Tritton and Siccama 1990). The nine paper birch-red spruce-balsam fir stands survey ranged from 33–86 dead trees/ac (81–212/ha) or 11–35 percent of basal area (Tritton and Siccama 1990), and overall, 14 percent of the trees in Maine were standing dead (Griffith and Alerich 1996). Dead wood provided an important substrate for spruce and hemlock seedling development (Weaver et al. 2009). While a commercial clearcut reduced the area of dead wood available for seedling growth, 5- and 20-year-selection cutting cycles were not statistically different from the uncut reference stand with 362–501 ft^2/ac (83–115 m^2/ha) of dead wood (Weaver et al. 2009).

As described above, spruce-fir forests tend to have two peaks in DWM over time: one early in stand development and a second peak after the stem exclusion phase (Figure 3). For example, one study showed a change from 63 t/ac (28 Mg/ha) in a stand <20 years, 22 t/ac (10 Mg/ha) in the 41–60-year age class, to 117 t/ac (52 Mg/ha) in the 61–80-year age class, and returning to less than 56 (25 Mg/ha) in the 101–120-year age class (Taylor et al. 2007). Fraver and colleagues (2002) showed that pre-harvest an Acadian forest had 10 t/ac (23 Mg/ha) of DWM. The harvest in this study increased the mass of DWM, but more of the pieces were small diameter (Fraver et al. 2002). While the harvest method (whole tree, tree length, or cut to length) and harvest system affect the amount of DWM left after harvest, many studies do not specify how material was removed.

Snag densities in balsam fir forests of Newfoundland followed a similar pattern over time. Stands contained nearly 16 snags/ac (40/ha) the first year post harvest; then the density declined below the 4 snags/ac (10/ha) required by the regional forest management guidelines at 20 years post harvest; and finally the number of snags returned to initial levels in the 80–100 years post-harvest stands (Smith et al. 2009). Smith and colleagues (2009) recommended retention and recruitment of white birch snags to ensure sufficient snag and DWM density. The Canadian province of Newfoundland and Labrador requires retention of 4 snags/acre while Maine recommends retention of 3 snags greater than 14 inches DBH and one greater than 24 inches DBH (Flatebo et al. 1999, Smith et al. 2009). Other guidelines recommend between 5 and 6 snags/acre greater than 8 inches and an additional 4–6 potential cavity trees (Woodley 2005).

A study of two old-growth balsam and black spruce sites demonstrated a wide range of average DWM piece sizes even in unmanaged lands. In the two study sites, the average diameter of the DWM structures were 54.8 cm and 16.1 cm; average height of snags was 4.73 m and 2.52 m; and length of logs were 5.91 m and 4.81 m (Campbell and Laroque 2007). The differences between the two sites are due, in part, to differences in rates of decomposition, i.e., higher rates of decomposition reduce the average size of DWM pieces.

One study of pre-commercial thinning in spruce-fir forests showed that the mass of DWM was reduced from 29–15 t/ac (64–34 Mg/ha; Briggs et al. 2000). In one study of a spruce-fir whole tree clearcut in Maine, 35 percent of organic matter was in trees and 12 percent was in woody litter and forest floor (Smith Jr et al. 1986). In that study, 23 t/ac (52 Mg/ha) of DWM were left after the harvest, but the whole-tree removal took about 91 percent of N, P, K, and Ca from the site, which was between 2 and 4 times the nutrient removal from a bole-only harvest (Smith Jr et al. 1986). Depletion of Ca is of some concern in Maine, though not as great a concern as in the Central and Southeastern U.S. (Huntington 2005). Spruce-fir forests generally incorporate Ca into merchantable wood at 1.6 kg Ca/ac/yr (1.6 kg $ha^{-1}yr^{-1}$; Huntington 2005).

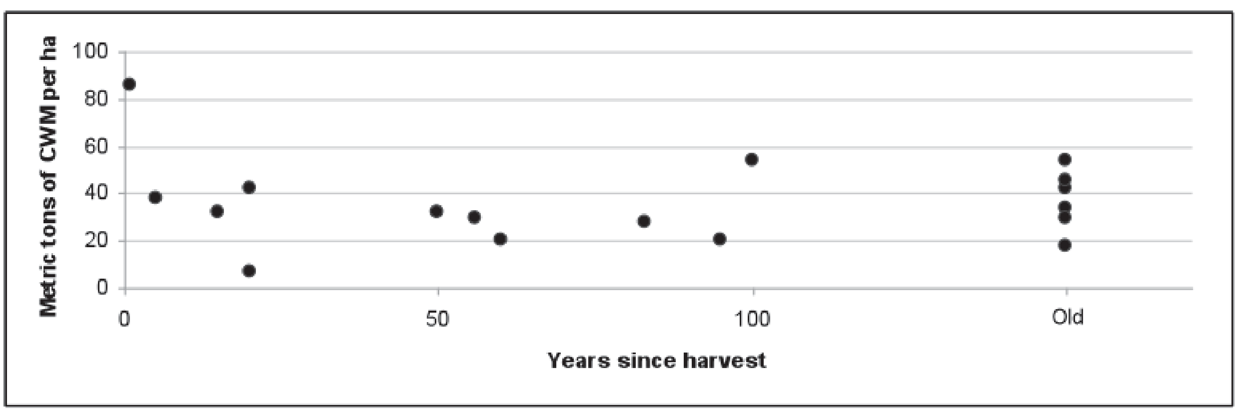
Some sites such as Weymouth Point, Maine, have documented Ca-depletion problems (Smith Jr et al. 1986, Hornbeck et al. 1990, Briggs et al. 2000). The rate of weathering replenishment of Ca in Maine is uncertain, and the Ca-rich mineral apatite may be an important source of Ca (Huntington 2005, Yanai et al. 2005). Climate change and the associated warming and species composition shift may exacerbate Ca depletion in spruce-fir forests (Huntington 2005).

3B. NORTHERN HARDWOOD FORESTS

Northern hardwood forests are dominated by maple (*Acer* sp.), beech (*Fagus grandifolia*), and birch (*Betula* sp.) and cover lower elevations and southern portions of Maine, New York, New Hampshire, Vermont, and the northern portion of Pennsylvania. Northern hardwood forests also include conifers, e.g., hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*), in the mixture (Westveld 1956).

In general, the amount of DWM in northern hardwood forests follows the 'U' pattern mentioned above. Young stands have large quantities of DWM; mature stand have less; and older or uncut stands have more. For example, a study in New Hampshire measured 38 t/ac (86 Mg/ha) of DWM in a young stand, 14 t/ac (32 Mg/ha) in mature stands, 20 t/ac (54 Mg/ha) in old stand, and 19 t/ac (42 Mg/ha) in an uncut stand (Gore and Patterson 1986). Gore and Patterson (1986) also note that stands under a selection system had lower quantities of DWM, i.e., 16 t/ac (35 Mg/ha). A review of other studies identified similar temporal patterns and quantities of DWM (see Figure 6 from data described in Roskoski 1977, Tritton 1980, Gore and Patterson 1986, McCarthy and Bailey 1994, McGee et al. 1999, Bradford et al. 2009).

Figure 6. Quantities of DWM in Northern hardwood forests Forests



Data described in Gore and Patterson 1986, McCarthy and Bailey 1994, McGee et al. 1999, Bradford et al. 2009, and Roskoski 1977

Estimates of the volume of down dead wood in Maine's northern hardwood forests are 598 ft^3/ac (42 m^3/ha) or 9 t/ac (20.5 Mg/ha Heath and Chojnacky 2001). Keeton (2006) estimates a volume of 600 ft^3/ac (42 m^3/ha) of DWM in a multi-aged northern hardwood forest.

The number of dead trees in five hemlock-yellow birch forests range from 16–45/ac (40–112/ha) or from 3–14 percent of the basal area (Tritton and Siccama 1990). The 14 sugar maple-beech-yellow birch stands survey ranged from 14–99 dead trees/ac (35–245/ha) or 5–34 percent of basal area (Tritton and Siccama 1990). Other estimates of snag densities in northern hardwood forests include 5/ac (11/ha) (Kenefic and Nyland 2007), 15/ac (38/ha) (Goodburn and Lorimer 1998), and 17/ac (43/ha) (McGee et al. 1999). Tubbs and colleagues (1987) recommend leaving a between of one and ten live decaying trees/acre of least 18 inches DBH.

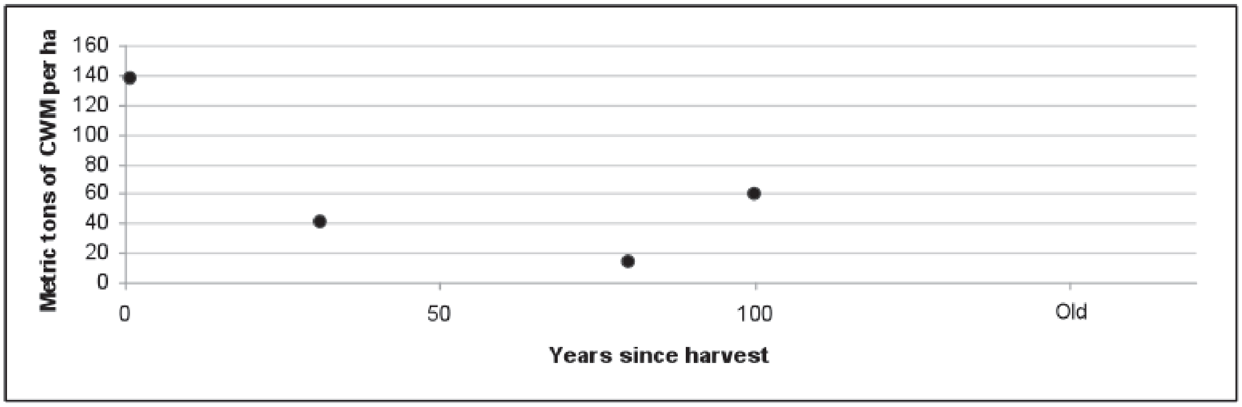
The number of cavity trees is another important habitat element in northern hardwood forests that is reduced by harvest. For example, studies in northern hardwood forests have shown a reduction from 25 cavity trees/ac (62/ha) before harvest and to 11 (27/ha) afterward (Kenefic and Nyland 2007). Another study measured 7 cavity trees/ac (18/ha) in old-growth, 4/ac (11/ha) in even-aged stand, and 5/ac (13/ha) in a stand selection system (Goodburn and Lorimer 1998).

3C. TRANSITION HARDWOOD FORESTS

Oak-hickory forests occupy the southernmost portions of the region. The oak-hickory forests are also considered a transitional forest type between the northern hardwood forests type and the Appalachian hardwoods that dominate further south (Westveld 1956).

As with the other forest types discussed, DWM density tends to follow a 'U' shape in oak-hickory forests. For example, Idol and colleagues (2001) found 61 t/ac (137 Mg/ha) in a one-year post-harvest stand, 18 t/ac (40 Mg/ha) in a 31-year-old stand, and 26 t/ac (59 Mg/ha) in a 100-year-old stand. Tritton and colleagues (1987) measured 5.8 t/ac (13 Mg/ha) in an 80-year-old stand in Connecticut.

Figure 7. DWM in Oak-Hickory Forests



Data described in (Tritton et al. 1987, Idol et al. 2001)

Estimates of the volume of down dead wood in Maine's oak-hickory forests are 244 ft^3/ac (17 m^3/ha) or 0.7 (1.5 Mg/ha; Heath and Chojnacky 2001). Wilson and McComb (2005) estimated the volume of downed logs in a western Massachusetts forest at 143 ft^3/ac (10 m^3/ha).

Out of seven oak stands in Connecticut, the number of dead trees ranged from 19–44/ac (46–109/ha) or 5–15 percent of basal area (Tritton and Siccama 1990). The decadal fall rates of snags in a Massachusetts study varied from 52–82 percent (Wilson and McComb 2005). Snags, particularly large-diameter snags, provide important nesting and foraging sites for birds (Brawn et al. 1982). In general, wildlife habitat requirements for dead wood are poorly documented, but it is clear that some wildlife species rely on dead wood in oak-hickory forests (Kluyver 1961, DeGraaf et al. 1992).

A study in Appalachian oak-hickory forests showed that the decomposing residues left after a sawlog harvest increased concentration of Ca, K, and Mg in foliage and soils after 15 years in comparison to a whole-tree harvest (Johnson and Todd 1998). However, the study found no impacts on soil carbon, vegetation biomass, species composition, vegetation N or P concentration, soil-bulk density, or soil N because of the whole-tree harvest (Johnson and Todd 1998).

3D. White Pine and Red Pine Forests

Pine forests are found in the coastal areas of Maine and New Hampshire and much of central Massachusetts. Pine forests tend to occupy sites with coarse-textured, well-drained soils (Barrett 1980).

Estimates of the volume of down dead wood in Maine's pine forests are 255 ft^3/ac (18 m^3/ha) or 1.6 t/ac (3.5 Mg/ha; Heath and Chojnacky 2001). A review of research on DWM in the red pine forests of the Great Lakes area showed that there were 50 t/ac (113 Mg/ha) of DWM in an unmanaged forest at stand initiation and 4.5 t/ac (10 Mg/ha) in a 90-year-old stand (Duvall and Grigal 1999). In comparison, the managed stand Duvall and Grigal (1999) studied had less DWM at both initiation 8.9 t/ac (20 Mg/ha) and at 90 years 2.9 t/ac (6.6 Mg/ha). The same review showed the unmanaged stand had 30 snags/ac (74/ha) while the managed forest had 6.9/ac (17/ha; Duvall and Grigal 1999). Red and white pine that fall to the ground at time of death will become substantially decayed (decay class IV of V) within 60 years (Vanderwel et al. 2006a).

While not a recognized forest type, stands with a mix of oak, other hardwoods, white pine, and hemlock are common. Many of the red oak and white pine stands on sandy outwash sites are susceptible to nutrient losses because of a combination of low-nutrient capital and past nutrient depletion (Hallett and Hornbeck 2000).

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APPENDIX 4–B

REVISED ASSESSMENT OF BIOMASS HARVESTING AND RETENTION GUIDELINES

Alexander M. Evans, Robert T. Perschel, and Brian Kittler

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1. Introduction

Interest in removing low-grade wood from forests has increased because of rising fossil fuel costs, concerns about carbon emissions from fossil fuels, and the risk of uncharacteristic wildfires.[1, 19] Most existing forest practice rules and recommendations did not anticipate this increased extraction of woody biomass and offer no specific guidance on how much removal is healthy for ecosystems. Intensification of biomass utilization, particularly for energy and fuel needs, presents a range of potential environmental risks.[31, 29] This report provides a review of guidelines put forth by states and other entities to avoid these environmental risks and promote the ecological sustainability of forest biomass utilization, and can inform a similar process to develop guidelines in Massachusetts.

1a. Woody Biomass

While definitions of biomass are usually similar, there can be surprising differences. For instance, the definition of biomass in New Brunswick, Canada's guidelines excludes pulpwood fiber from whole-tree chipping.[42] Technically, the term woody biomass includes all the trees and woody plants in forests, woodlands, or rangelands. This biomass includes limbs, tops, needles, leaves, and other woody parts.[44] In practice, woody biomass usually refers to material that has historically had a low value and cannot be sold as timber or pulp. Biomass harvesting might even remove dead trees, down logs, brush, and stumps.[37] Markets determine which trees are considered sawtimber material and which are relegated to the low-value biomass category. Changing markets and regional variations determine the material considered biomass, but in general it is a very low quality product. In some cases, woody biomass is defined by how the material is used. For example, in Pennsylvania any material burned for energy is defined as biomass.[46]

In this report, the term **biomass** refers to *vegetation removed from the forest, usually logging slash, small-diameter trees, tops, limbs, or trees that cannot be sold as higher-quality products such as sawtimber.* This report does not discuss biomass from agricultural lands and short-rotation woody biomass plantations.


Photo: Zander Evans

Biomass can be removed in a number of ways. Some harvests remove only woody biomass, some combine the harvest of sawtimber or other products with biomass removal, and some remove biomass after other products have been removed. This report focuses on what remains in the forest after harvest and not on the type of harvest. The goal is to ensure the forest can support wildlife, provide clean water, sequester carbon, protect forest soil productivity, and continue to produce income after a biomass harvest or repeated harvests. In some regions, current wood utilization is such that no woody material is available for new markets such as energy. For these high-utilization areas, following biomass guidelines may result in more biomass being left in the forest.

1b. Coarse Woody Material

Woody material is sometimes divided into coarse woody material (CWM), fine woody material (FWM), and large woody material. CWM has been defined as being more than 6 inches in diameter at the large end and FWM as less than 6 inches in diameter at the large end.37 The U.S. Forest Service defines CWM as down dead wood with a small-end diameter of at least 3 inches and a length of at least 3 feet and FWM as having a diameter of less than 3 inches.62 FWM tends to have a higher concentration of nutrients than CWM. Large downed woody material, such as logs greater than 12 inches in diameter, is particularly important for wildlife. In this report, we use the term **downed woody material (DWM)** to encompass all three of these size classes, but in some circumstances we discuss a particular size of material where the piece size is particularly important.

1c. Why "Biomass" Guidelines?

> *Good biomass harvesting practices can enhance and improve forest land; poor practices can damage and devalue it.46*

In the United States, forestry on private and state forests is regulated primarily at the state level. At least 276 state agencies across the country have some oversight of forestry activities, including agencies focused on forestry and other state agencies, such as wildlife or environmental protection.[17] Federal law requires states to address non-point source pollution of waterways. All 50 states have Best Management Practice (BMP) programs that are intended to protect water quality and other values. The programs usually include sections on timber harvesting, site preparation, reforestation, stream crossings, riparian management zones, prescribed burning and fire lines, road construction and maintenance, pesticides and fertilizers, and wetlands. Programs in states vary from laws that prescribe mandatory practices to states that use voluntary BMPs and education and outreach programs. These programs can be categorized in four ways: non-regulatory with enforcement, regulated, and combination of regulatory and not regulatory. In the northeast, Massachusetts and Connecticut are considered regulated, Vermont and New Hampshire are non-regulated with enforcement and Rhode Island, New York, and Maine use a combination of approaches. These programs are routinely monitored and literature suggests that when these BMPs are properly implemented they do protect water quality.[51] With so much existing regulation, why are additional biomass harvesting guidelines necessary? Reasons for biomass harvesting guidelines are likely to mirror the reasons forestry is regulated in general, which include[16]:

- general public anxiety over environmental protection,
- the obligation to correct misapplied forestry practices,
- the need for greater accountability,
- growth of local ordinances,
- landscape-level concerns, and
- following the lead of others.

Photo: Zander Evans

More specifically, biomass harvesting guidelines are designed to fill the gaps where existing BMPs and forest practice regulations may not be sufficient to protect forest resources under new biomass harvesting regimes. In other words, BMPs were developed to address forest management issues at a particular point in time; as new issues emerge, new guidelines may be necessary. Existing guidelines did not anticipate the increased rate or new methods of biomass removal and offer no specific guidance on the amount of extraction that is acceptable for meeting a range of forest management objectives. For example, Pennsylvania's old BMPs encouraged operators "to use as much of the harvested wood as possible to minimize debris," while the new guidelines recommend leaving "15 to 30 percent of harvestable biomass as coarse woody debris."[46] Michigan's guidelines point out that while the state "has a rich history of utilizing woody biomass for bioenergy and biobased products such as lumber, pulp and paper, composites, heat and electrical generation," as "market opportunities expand for woody biomass, it is crucial that harvesting and removal of woody biomass be done using sustainable forest management principles and practices that are ecologically, economically, and socially appropriate."[36] Concerns about long-term site productivity, biodiversity, and wildlife populations drove the Minnesota state legislature to call for biomass harvesting guidelines, and the resulting guidelines are intended to be implemented in close conjunction with the existing Minnesota forestry guidelines, which cover a range of additional management considerations.[37] More generally, biomass guidelines focus on DWM levels, wildlife and biodiversity, water quality and riparian zones, soil productivity, silviculture, and, in some cases, other issues. For example, Maine's guidelines focus "on the amount of biomass that should be left on-site after harvest and the effect on soil productivity, water quality, and biodiversity."[7]

1d. An Examination of Current Guidelines

This report reviews the biomass harvesting or retention guidelines from New York and New England, other states with specific biomass guidance, parts of Canada, Northern European counties, and other organizations, including the U.S. federal government and certification groups. We have grouped New York and the New England states together to offer a snapshot of the current situation in states geographically near Massachusetts. Maryland, Minnesota, Missouri, Michigan, Pennsylvania, Wisconsin, and California are also covered, because of their forest practices guidance on biomass harvest and retention. In some states guidelines are still under review at the time of this writing and subject to change. Readers are encouraged to use the links in Appendix II to check the latest drafts of the guidelines.

The examples in this report detail the status of rules and recommendations for removing biomass from our forests. Entities interested in addressing concerns about biomass removal have taken at least three different approaches. One is to verify that existing forest practice regulations cover the issues raised by biomass harvests, obviating the need for new guidelines. In instances where existing rules or recommendations are found to be insufficient, some entities—including Minnesota, Missouri, Pennsylvania, Wisconsin, and Maine—have taken a second type of approach and chosen to craft separate biomass guidelines that augment existing forest practice guidance. In the third case, entities such as the Forest Stewardship Council (FSC) have chosen to address concerns particular to biomass harvests by revising existing rules or recommendations.

The existing guidelines cover topics such as dead wood, wildlife and biodiversity, water quality and riparian zones, soil productivity, silviculture, and disturbance. Appendix I lists the commonly used subtopics for each and identifies which are covered in a given set of guidelines. In some cases, a subtopic is noted as covered because it appears in another set of forestry practice rules or recommendations instead of that state's biomass guidelines. The list of subtopics was developed from section headings in all the various existing guidelines and is similar to other criteria for sustainable production and harvest of forest biomass for energy.[31] It should be noted that each set of guidelines takes a slightly different approach, addressing topics with a greater or lesser degree of specificity. The precepts of sustainable forest management call for identifiable criteria and indicators, such as those identified through the Montreal Process, for the purpose of benchmarking and measuring forest practices. The critique that follows does not always address why topics are covered with more or less specificity, but presumes that more specificity will increase the likelihood that guidelines will encourage sustainable management.

2. BIOMASS RETENTION GUIDELINES FOR TIMBER HARVESTING IN NEW YORK AND NEW ENGLAND

2a. Maine

In Maine, "guidelines specific to woody biomass retention are missing from existing best management practices and regulations."[40] Therefore, the state undertook a collaborative effort between the Maine Forest Service, the University of Maine, and the Trust to Conserve Northeast Forestlands to develop woody biomass retention guidelines. Participating committee members included Manomet Center for Conservation Sciences, the Forest Guild, the Maine Forest Products Council, and other forestry professional and environmental organizations. After a multi-year process and several drafts, *Consideration and Recommendations for Retaining Woody Biomass on Timber Harvest Sites in Maine* was released in 2010.[7] The project's goal was to address the growing interest in woody biomass and concerns about long-term sustainability of biomass harvesting by developing guidelines for the retention of woody biomass. The Maine guidelines define woody biomass as "logging residues, previously un-merchantable stems, and other such woody material harvested directly from the forest typically for the purposes of energy production."[40] These new guidelines augment the current Water Quality BMPs, which are effectively applied in most harvests (77 percent of stream crossings and 89 percent of approaches to the crossings[39]).

The biomass harvesting recommendations report includes an extensive background section and literature review, including three key documents:

- *Best Management Practices for Forestry,*[38]
- *Site Classification Field Guide,*[9]
- *Biodiversity in the Forests of Maine: Guidelines for Land Management.*[18]

It also includes appendices that summarize regional recommendations pertaining to wildlife trees and biomass harvesting. The background section covers soil productivity, water quality, and forest management, as well as forest biodiversity; at the end of each section are voluntary guidelines. In earlier drafts, the voluntary guidelines offered after each section were more specific and stringent, but the final version lacks specific targets. Earlier drafts referred to the entire effort as "Guidelines," but the reframing of the title indicates the struggle the committee members had in agreeing on specific targets and the vagueness of the final product. For example, the voluntary guidelines for soils indicate forest litter should be left on-site "to the extent possible" and that operators should "minimize removal" of FWM on low-fertility sites.

This lack of specificity is found in other sections as well. The commentary on setting targets for the Forest Biodiversity section helps shed light on the decision-making dynamics that led to the dilution of the final product. The background information for the Forest Biodiversity section draws heavily on *Biodiversity in the Forests of Maine*. This report, a comprehensive manual outlining recommended guidelines for maintaining biodiversity in the forests of Maine, was the culmination of a multi-year process in the 1990s that included a wide range of stakeholders, including industry representatives, forest professionals, and environmental organizations. Originally published by Flatebo and colleagues[22], it was updated by Elliot.[18] Although the final version of the current biomass retention report utilizes the recommendations from the biodiversity report as background information and indicates that woody biomass harvesting practices "will have to comply with established recommendations for biodiversity as defined for non-biomass harvests,"[7] the specific targets listed in the biodiversity report are never incorporated as guidelines. The report indicates that since there was "not widespread acceptance of those guidelines within Maine's forest industry, specific targets for maintenance of site-level biodiversity are not included" in the relevant section.[7]

The result for the Forest Biodiversity section is that the Voluntary Guidelines call for leaving "as much fine woody material as possible" without the specific guidelines for DWM retention found in some other state guidelines. The guidelines also call for leaving "some wildlife trees" without incorporating targets for numbers of trees per acre suggested in *Biodiversity in the Forests of Maine*. The report indicates that this vagueness in the guidelines reflects the challenges of setting specific targets at site levels[18] and that although science can direct selection of biological indicators, it is still weak in selecting specific target levels.[24]

2b. New Hampshire

While New Hampshire currently has no specific biomass harvesting guidelines, existing recommendations and rules address the major biomass harvesting topics. New Hampshire's Slash Law (RSA 227-J:10) focuses on "debris left after a timber harvest" and states that "these branches, leaves, stems, unmerchantable logs, and stumps may take several years to decompose. Slash represents a fire hazard and, often, a messy appearance." The Slash Law sets a limit on the height of slash that can be left on-site, but does not set any minimum to retain on site.

Photo: Christopher Riely

New Hampshire's Basal Area Law (RSA 227-J:9) states that no more than 50 percent of the basal area can be cut near streams, water bodies, and public roads. Intensive biomass removal may decrease this law's ability to prevent erosion, provide wildlife habitat, protect stream temperature and aquatic life, and preserve the aesthetics of the landscape, because removal of DWM is not regulated by a basal area restriction. In New Hampshire, BMPs are voluntary, but the guide *Good Forestry in the Granite State: Recommended Voluntary Forest Management Practices for New Hampshire* includes sections on soil productivity, DWM, and retention of forest structures for wildlife habitat.[13] *Good Forestry* does not provide specific guidance on retention of tops and limbs, though it does recommend leaving "some cull material" in the woods after a biomass harvest. The section on soil productivity provides recommendations that would limit biomass removal on sites with nutrient-poor soils:

- Identify low fertility soils from maps and descriptions.
- Use bole-only harvesting (taking out the main portion of tree only, leaving branches and limbs in the woods) on low-fertility soils, or where fertility is unknown, as a precaution against nutrient loss.
- If whole-tree harvesting hardwoods, try to plan harvests during leaf-off periods to retain leaves and nutrients on site.
- Limit disruption of soil organic layers except when needed to accomplish silvicultural objectives (such as regeneration of species that need a bare mineral soil seedbed).[13]

Similarly, the Habitat section recommends retention of cavity trees and snags:

- In areas under uneven-aged management, retain a minimum of 6 secure cavity and/or snag trees per acre, with one exceeding 18 inches DBH and 3 exceeding 12 inches DBH. In areas lacking such cavity trees, retain trees of these diameters with defects likely to lead to cavity formation.
- In areas under even aged management, leave an uncut patch for every 10 acres harvested, with patches totaling 5 percent of the area. Patch size may vary from a minimum of 0.25 acre. Use cavity trees exceeding 18 inches DBH or active den trees as nuclei for uncut patches. Remember, the larger the tree, the more species that can use it. Riparian and other buffers can help to satisfy this goal.
- Retain live trees with existing cavities.[13]

The *Good Forestry in the Granite State* guide also has recommendations for retention of DWM:

- Avoid damaging existing downed woody debris, especially large (18+ inches) hollow or rotten logs and rotten stumps during harvesting operations (including tree falling, skidding, and road and skid trail layout).
- Leave cull material from harvested trees, especially sound hollow logs, in the woods. Some cull material should be left behind during whole-tree or biomass harvesting operations that may otherwise utilize this material. Large pieces of cull material bucked out on the landing should be returned to the woods.
- Avoid disrupting downed logs in and adjacent to streams, ponds, and wetlands.

- Avoid disrupting upturned tree roots from May to July to protect nesting birds.
- Maintain or create softwood inclusions in hardwood stands to provide a supply of longer-lasting down woody material.[13]

A revision of Good Forestry in the Granite State is currently underway and the recommendations for DWM in the draft are similar to the existing language.

2c. Vermont

Although Vermont's guide to *Acceptable Management Practices for Maintaining Water Quality on Logging Jobs in Vermont* is in its ninth printing, there is very little in the guide that would affect biomass harvesting or retention.[60] The guide's intent is to prevent discharges of mud, petroleum, and wood debris from getting into waterways. These BMPs are not mandatory unless a landowner is participating in Vermont's Use Value Act. The state's two wood-powered power plants in Burlington and Ryegate are required by the Public Service Board to ensure that their wood supply comes from sales with a harvest plan cleared by the Vermont Department of Forests, Parks and Recreation. The main focus of the review of harvest plans is to protect deer wintering areas. Related rules include the Heavy Cutting rules (Act 15), which require clearcuts (a reduction of basal area below the C-level) larger than 40 acres to have a permit (Title 10 V.S.A. Chapter 83, Section 2622). Another regulation that has some relevance to biomass harvesting is the requirement that whole-tree chip harvesters obtain a license (Title 10 V.S.A. Chapter 83, Section 2648).

An act of the Vermont Legislature created a Biomass Energy Development Working Group in 2009. That group is meeting regularly in a two-year initiative to address the major charges of (1) enhancing and developing Vermont's biomass industry while (2) maintaining forest health. As part of its process, subgroups are addressing issues such as economic incentives, supply models, available technology, and workforce availability. A Forest Health subgroup will consider guidelines for retention of woody biomass, forest health indicators, and emerging research on carbon and biomass harvesting issues.

2d. New York

New York's forest practice regulations are based the Environmental Conservation Law (§ 9-0105), though the regulations appear to only cover prescribed fires. *The Best Management Practices for Water Quality* has no recommendation about retention of DWM, snags, or other elements specific to biomass harvesting.[45] These BMPs cover planning, landings, stream crossings, roads and trails, vernal pools, erosion control techniques, and post-harvest considerations. This document is under revision and will include expanded sections on riparian and wetland zone management but nothing on the ecological or silvicultural aspect of biomass harvesting. New York currently has no immediate plans to develop biomass harvesting guidelines. They are monitoring developments in other states and a biomass study now taking place at the Adirondack Research Consortium.

However, when New York initiated its renewable portfolio standard, it established an eligibility procedure for electrical power generators utilizing forest biomass. The resulting requirements are modeled after Vermont's and include procurement plans for each facility to include forest management plans for source forests and harvest plans filed for all harvests. Adherence to these standards is monitored periodically by state foresters. New York varied slightly from Vermont's approach by providing exemptions to properties that are accredited by FSC, Sustainable Forestry Initiative, or Tree Farm.

2e. Rhode Island

Rhode Island's BMP guidance is encapsulated in the document *Rhode Island Conservation Management Practices Guide*.[12] The Guide includes water-quality protections such as filter strips between harvested areas and streams or ponds. Rhode Island does require the registration of "woods operators" with the Division of Forest Environment and notification of intent to harvest timber (RI State Statues, Title 2, Chapter 2-15, Sections 1 and 2). Rhode Island has no current intentions to develop biomass harvesting guidelines, although it is aware of the issue and may address it in the future.

2f. Connecticut

Connecticut's BMP field guide was revised in 2007 and focuses specifically on water-quality issues.[15] This guide, like New York's and Rhode Island's, has little effect on biomass removals or DWM retention.[12, 15, 45] Connecticut is now seeking funding to address biomass harvesting guidelines. Current BMPs recommend keeping slash out of water bodies and vernal pools. Connecticut's BMPs do suggest that "brush and slash may be placed in skid trails and on slopes to slow water flow and retain sediment."[15] One layer of protection is the state's certification program for foresters and loggers. Connecticut is watching the development of the biomass market carefully and would like to have some guidelines in place. It is now looking for funding for developing guidelines, possibly through a joint project between the state forestry department and the Connecticut Forest and Parks Association.

3. Review of State Biomass Harvesting and Retention Guidelines

3a. Michigan Woody Biomass Harvesting Guidance

Since 2008, the Michigan Department of Natural Resources has worked with a stakeholder group drawn from academia, environmental groups, forest industry, and state and federal agencies to develop biomass harvesting guidelines.[36] These guidelines were designed to be used in conjunction with Michigan's *Sustainable Soil and Water Quality Practices on Forest Land* manual.[35] They emphasize that "not every recommendation listed in this guidance can or should apply to every situation." While the Michigan guidelines provide a list of scientific references, there are no specific citations to support the retention or removal of forest biomass.

Topics such as riparian zones and pesticide use are covered by *Sustainable Soil and Water Quality Practices* and not in the biomass harvesting guidelines. Though brief, Michigan's biomass guidelines, in combination with *Sustainable Soil and Water Quality Practices*, cover most of the major biomass harvesting topics (see Appendix I). However, there is little guidance on retention of snags. Michigan's guidelines also lack specificity in some areas. For example, they suggest retention of anywhere from one-sixth to one-third of material less than 4 inches in diameter from harvested trees.

3b. Minnesota: Biomass Harvesting Guidelines for Forestlands

The Minnesota state legislature directed the Minnesota Forest Resources Council (MFRC) and the Minnesota Department of Natural Resources (DNR) to develop guidelines for sustainably managed woody biomass.[37] The goal of the guidelines was to help natural resource managers, loggers, equipment operators, contractors, and landowners make decisions about biomass harvesting. With the support of the DNR's Ecological Services, Fisheries and Wildlife, and Forestry divisions, the MFRC directed the guideline development process. The 12-member interdisciplinary technical committee developed separate guidelines for brushland as well as for forestland. The technical committee reflected a range of expertise deemed pertinent to the development of these guidelines, including soil science, wildlife biology, hydrology, forest management, and silviculture. Meeting summaries were provided online, and the committee's work was peer-reviewed and open to public comment. Minnesota's biomass harvesting guidelines were crafted to be part of the MFRC's 2005 forest management guidebook, *Sustaining Minnesota Forest Resources*, and the existing guidelines were integrated into the new biomass recommendations.

Minnesota's biomass harvesting guidelines are rooted in precepts of ecological forestry. For example, the guidelines recommend emulating natural disturbances with silviculture and maintaining biological legacies after harvest. The guidelines make the case that, in Minnesota, biomass harvesting increases the disparity between managed stands and their natural analogs because it reduces the biological legacies left after harvest, such as slash and fallen logs. The guidelines cover almost all of the topics and subtopics related to biomass harvesting we considered in our analysis (see Appendix I). The only topics not obviously included or referenced were aesthetics, forest diseases, and land conversion.

Photo: Eli Sagor

A recent field test—an experimental biomass harvest—suggests that the harvesting practices utilized for biomass harvest in Minnesota can remove woody biomass without significant negative impacts on snags and DWM. The test harvest had a small effect on the number of snags and on the amount of DWM. Reductions in DWM were small (2 tons per acre or less) and one site showed an increase in DWM.[5] In addition, of the seven test sites where snags were measured, only three had a lower number of snags after harvest.[5]

3c. Missouri: Best Management Practices for Harvesting Woody Biomass

The catalyst for the development of biomass harvesting guidelines in Missouri was state legislation introduced in February 2007 concerning cellulosic ethanol.[34] In response to the lack of BMPs for biomass harvests, the Top of the Ozarks Resource Conservation and Development (RC&D), in partnership with Big Springs RC&D, Bootheel RC&D, the Eastern Ozarks Forestry Council, and the Missouri Department of Conservation, applied for and received a grant from the Northeastern Area State and Private Forestry branch of the U.S. Forest Service to develop BMPs for biomass harvesting. The BMPs development process continued to emphasize participation through a stakeholder meeting for a cross-section of interested parties to discuss issues and possible criteria to be addressed in the BMPs for harvesting woody biomass. A technical committee brought expertise on soil science, wildlife biology, hydrology, forest management, and silviculture to the process. Meeting announcements and notes were provided online to allow for transparency in the development of BMPs.

Photo: Zander Evans

The Missouri guidelines cover the major biomass harvesting topics (see Appendix I). Subtopics not covered in the Missouri guidelines include regeneration, removal of litter and forest floor, and fuel reduction. A section on pesticides was included in an early version of the biomass guidelines, but was later dropped because of its lack of relevance to biomass.

3d. Pennsylvania: Guidance on Harvesting Woody Biomass for Energy

Pennsylvania's guidelines are a direct result of increased interest in woody biomass for energy. The passage of Pennsylvania's Alternative Energy Portfolio Standards Act (Act 213 of 2004) helped drive that interest by requiring "all load-serving energy companies in the state to provide 18 percent of their electricity using alternative sources by the year 2020." In response to the interest in using Pennsylvania's forests to help meet alternative energy goals, the Department of Conservation and Natural Resources (DCNR) created biomass harvesting guidelines, intending to balance the need for alternative energy sources with the need to protect forest resources for all citizens and future generations. Pennsylvania's guidelines include short-term rotational biofuel crops that might not traditionally fall under forest management guidelines.

Harvests on state forests are required to follow Pennsylvania's guidelines. The guidelines also supply recommendations for private lands; these are drawn from *Best Management Practices for Pennsylvania's Forests*, which was published by the Forest Issues Working Group in 1997. However, the new biomass guidelines did not draw on wider stakeholder participation, in part because of the time pressure to produce guidelines before forest-based energy projects were initiated. Pennsylvania's guidelines are also unusual in that they include comments on biomass policy and a supply assessment. For example, the guidelines suggest that facilities requiring 2,000 tons per year are better suited to Pennsylvania than larger facilities. The guidelines also make a case for woody biomass as a carbon-neutral fuel source.

Since Pennsylvania's state forestlands are certified as meeting the standards of FSC, their biomass harvesting guidelines directly reference FSC standards. Pennsylvania's DCNR uses the FSC's Appalachia Regional Standard, but the state biomass harvesting guidelines provide greater specificity on woody biomass removals. For example, the FSC standard requires that "measures to protect streams from degradation of water quality and/or their associated aquatic habitat are used in all operations." The Pennsylvania biomass guidelines extend this idea by adding "biomass harvesting of any materials along stream and river banks or along bodies of water is unacceptable." The Pennsylvania biomass guidelines cover the range of potential biomass harvesting subtopics. Non-point source pollution and pesticides are not dealt with in the biomass harvesting guidelines, but these are covered in general forestry guidelines for Pennsylvania.

3e. Maryland: Development of Forest Biomass Harvesting Guidelines

Maryland is currently in the process of developing biomass harvesting guidelines. The Pinchot Institute for Conservation is facilitating a committee of individuals representing state forestry, environmental and energy agencies, cooperative extension, private landowners, non-profit conservation organizations, and local governments. Specialists in ecology, forest hydrology, forestry, economics, and other disciplines are included on the advisory committee. The guidelines will address the charge of the *Maryland Climate Action Plan*, which states, "All biomass will be sustainably harvested without depriving soils of important organic components for reducing erosion, but will maintain soil nutrient structure, and will not deplete wildlife habitat or jeopardize future feedstocks in quantity or quality." As such, Maryland's biomass guidelines will address the protection of forest soils, water quality and aquatic resources, wildlife habitat and biodiversity, and silviculture and vegetation management. Other topics may also be included in the final version of the guidelines document. This guideline document is also linked to a technical support document that addresses the potential impacts associated with forest biomass harvesting in Maryland and a review of relevant statutes and regulatory and non-regulatory programs that operate within the state.

3f. Wisconsin's Forestland Woody Biomass Harvesting Guidelines

Wisconsin's biomass guidelines were motivated by new price incentives to produce wood-based renewable energy and concerns about the environmental impacts of increased woody biomass removal.[26] The Wisconsin Council on Forestry created an advisory committee with members from tribal, state, non-profit, and private forestry organizations. The guidelines were also reviewed by subject experts.

The guidelines cover much of the same ground as the other state guidelines (Table 1). They take advantage of the existing guidance provided by Wisconsin's *Silviculture and Forest Aesthetics Handbook and Forestry Best Management Practices for Water Quality*. Issues such as regeneration, water quality, and aesthetics are dealt with in the existing manuals rather than the new biomass guidelines. A major focus of the Wisconsin guidelines is the identification of soil types, such as shallow, sandy, or wetland, that are most at risk of nutrient depletion.

Photo: Zander Evans

3g. California Forest Practice Rules

California has some of the most comprehensive forest management regulations in the world. While there are currently no rules designed to specifically address intensive removal of forest biomass, the existing regulations address all of the main topics and most of the subtopics of woody biomass removal (Appendix I). For example, the *California Forest Practice Rules* point out that snags, den trees, and nest trees are a habitat requirement for more than 160 species and play a vital role in maintaining forest health. The importance of snags translates into regulations that require retention of all snags except where specific safety, fire hazard, insect, or disease conditions require they be felled.[11]

California's regulations demonstrate the tradeoffs between the ecological benefits and the potential fire hazards of retaining dead wood on-site in fire-adapted ecosystems.[10] For example, the *California Forest Practice Rules* emphasize the ecological importance of DWM for soil fertility, moisture conservation, and the support of microorganisms, but regulations dictate slash removal rather than retention. However, in riparian areas the Forest Practice Rules require operations to "protect, maintain, and restore trees (especially conifers), snags, or downed large woody debris" that provide stream habitat.[11]

A technical team of the Interagency Forestry Working Group is currently reviewing whether forest practice regulations in the state assure the ecological sustainability of forest biomass production and harvest. This technical team will also examine the carbon sequestration and storage impacts of both forest management and catastrophic fires.

4. BIOMASS GUIDELINES AND POLICY IN CANADA

As with state biomass guidelines in the U.S., woody biomass policy and guidelines in Canada are designed and implemented at the provincial level, not by the central government. Another similarity between the U.S. and Canada is the shift from a greater proportion of private holdings in the East to greater government (i.e., Crown) land ownership in the West. While provincial biomass guidelines would apply to public land and not private land, private landowners in eastern Canada are asking provincial governments for guidance on how best to manage their private land for bioenergy.

An overview of biomass policy and guidelines from east to west in Canada reveals variation similar to that in the United States.[48] Nova Scotia has formed a multi-stakeholder biomass committee of government, industry, and environmental groups that is discussing guidelines. There is currently a two-year moratorium on harvesting logging residue there to allow for input from this committee and then the creation of a government policy. In New Brunswick, the Department of Natural Resources has prepared draft guidelines on forest biomass harvesting. New Brunswick's guidelines take advantage of a decision support tool for sustainable biomass allocation that evolved from a model used to predict impacts of atmospheric deposition. The guidelines exclude harvests on high-risk (low-nutrient) areas, and harvest and silviculture planning remain separate processes guided by the Crown land management framework. The policy calls for biomass harvesting sustainability to be assessed over an 80-year time period, which is "equivalent to the life span of an average forest stand."[42] The New Brunswick guidelines define biomass such that the guidelines do not apply to pulpwood fiber from whole-tree chipping.

Like New Brunswick, Quebec is in the process of developing biomass guidelines based on soil properties. Ontario's policy establishes objectives such as "to improve the utilization of forest resources by encouraging the use of forest biofibre for the production of energy and other value-added bioproducts." However, the management and sustainable use of forest biomass is still guided by existing legislation (e.g., the Crown Forest Sustainability Act and its associated regulated manuals and procedures). In British Columbia, biomass removals during current forest practices (e.g., full-tree with processing at roadside) are already covered under the Forest and Range Practices Act (FRPA). Regulations under the FRPA require the retention of at least 1.6 logs per acre (at least 16 feet in length and 12 inches in diameter on the coast and 6.5 feet in length and 3 inches in diameter in the interior; FRPA §68). In addition, a strategic plan for increased biomass removals is being developed, and scientists have begun to collate data that will be used to formulate guidelines for increased slash harvesting.

A 2008 conference entitled "The Scientific Foundation for Sustainable Forest Biomass Harvesting Guidelines and Policies," hosted by Canada's Sustainable Forest Management Network, helped set the stage for future policy development by providing an overview of existing research on biodiversity,[33] site productivity considerations for biomass harvests,[55] and existing knowledge gaps.[56]

5. BIOMASS GUIDELINES AND POLICY IN NORTHERN EUROPE

Woody biomass provides a large contribution to the heat of Northern Europe and is also utilized for co-firing with coal and for straight biopower facilities in some countries such as the Netherlands and in the UK. Though management guidelines are similar across Northern Europe, their integration under the broader forest management policy is more varied. For example, the UK and Finland have determined that biomass harvesting guidelines work best as independent reference documents to help guide practitioners, whereas Austria and Sweden have integrated biomass harvesting protocols directly into their broader forest management protocols and regulations. The following section will review the approach that countries in Northern Europe have taken to biomass harvesting standards.

5a. Sweden

The use of forest-based bioenergy in Sweden increased in the 1980s as a result of growing concern over a reliance on imported oil and nuclear power. In 1991, the Swedish government introduced a carbon tax on fossil fuels used for heat and transportation. Since this time, the use of forest-based biomass for energy generation has more than doubled and forest-based bioenergy now accounts for more than 27 percent of total Swedish energy consumption (Swedish Energy Agency, 2008). Harvest regimes have responded to this growing demand for biomass by becoming increasingly mechanized, with preference for whole-tree harvesting (WTH) systems for both thinnings and final clearcut harvests.[4, 8, 50, 32] From 50 to 80 percent of slash is typically removed, depending on site conditions and economic constraints.[32] By some estimates, the share of bioenergy in Sweden could feasibly double before environmental and economic considerations fully constrain this supply.[43]

Sweden is 67 percent forested, and the vast majority of these forests are held by private owners with high willingness to manage their forest and harvest timber. The responsibility for ensuring that energy wood harvests are done in a sustainable manner is largely left to individual landowners, and the greatest area of concern that landowners have about the sustainability of biomass harvesting centers on nutrient cycling and site productivity.[52] WTH clearcutting systems can increase soil nutrient losses by up to 7 percent, lead a reduction in site productivity of up to a 10 percent, and have been linked to an increased rate of loss of biodiversity in managed forests in Sweden.[54, 8, 49] In an attempt to mitigate these risks, the Swedish Forest Agency developed a set of recommendations and good-practice guidelines for WTH in 1986; these were updated in the 1990s and codified in the Swedish Forest Act of 2002. This legislation seeks to control WTH practices in order to limit impacts to forest soils, water resources, and long-term site nutrient balances.

The general approach of Sweden's guidelines and regulations is to classify different sites according to the risks associated with biomass removal at these sites. Different recommendations are then applied based on these classifications. In Sweden these specifications are to ensure that

- all forest residues are dried and needles are left on-site before biomass removal,
- sites in northern Sweden with abundant lichens should be avoided, and
- sites with acidified soils, peat lands, or sites with a high risk of nitrogen depletion should be compensated with ash and nitrogen application.

Like other Nordic countries, Sweden prohibits in-stand drying of forest residues in late spring and early summer to manage risks associated with bark beetle infestations. The guidelines and regulations also specify appropriate forest residue removal rates for different regions of Sweden, based on the risk of soil nutrient loss associated with historic and current patterns of acid deposition in these different regions. WTH clearcut operations are prohibited where they may negatively impact endangered species. The guidelines also stipulate that at least 20 percent of all slash must be left on-site. In addition to these site-specific guidelines, Swedish guidelines and regulations include criteria and indicators for sustainable forest management, forest certification, legislation, soil fertility, soil organic matter, wood production, biodiversity and wildlife, insects and fungi, hydrology and water quality, archaeological resources, cultural resources, recreational resources, nature conservation, silviculture, retention of tree species that are less commonly left in the stand, and stump harvesting.[53]

To hedge against the risk of soil nutrient depletion, the Swedish Forest Agency introduced additional wood ash recycling requirements in 2008; these supplement existing guidance on fertilization. The updated guidelines and regulations require that ash be applied to sites if the amount of harvest residues removed over the course of a rotation exceeds a half ton per hectare (0.2 tons per acre). For areas where biomass removals do not exceed this limit, ash recycling is deemed unnecessary; however, the regulation stipulates that ash be recycled in areas of high acid deposition, such as the southwest portion of the country. In Sweden, typical biomass removals are 0.5–1 ton per hectare, so recycling is de facto required on most sites. The prescription is to apply 2–3 tons per hectare every ten years and not to exceed two applications (i.e., 6 tons of ash per hectare). Ash is also supposed to meet certain chemical composition standards and be hardened when applied to facilitate infiltration of nutrients into soils.[32] Sweden's guidelines also suggest that it is acceptable to apply ash in stands that have not yet been harvested, as a means to mitigate potential loss of site productivity if whole-tree removals are planned. Sweden is a strong proponent of forest certification, and the Swedish FSC standards specify that the recommendations of the Swedish forest agency are to be followed where biomass is used for energy.

5b. Finland

Finland is 74 percent forested with boreal and sub-boreal mixed softwood forests largely dominated by pine, spruce, and birch species. Upwards of 80 percent of the domestic roundwood supply comes from the three-quarters of the land base that is in private ownership.[27] This land base supports a robust bioenergy sector. A full 20 percent of Finland's total energy consumption comes in the form of bioenergy, with 11 percent of the nation's electricity production coming from wood.[25, 27, 50] Approximately 47 percent of the annual Finnish roundwood supply is consumed in the production of energy.[25] Finland also imports an estimated 21 percent of the total wood it consumes for energy.[30] Finnish forest policy has made a goal of increasing the annual use of wood for energy by 5 million cubic meters, or nearly 5 million green tons.[52]

As in Sweden, harvests in Finland are highly mechanized, and WTH clearcuts are common practice. It is estimated that typical harvests of this nature remove between 60 and 80 percent of the total site biomass.[54, 28, 47, 50, 61] Finnish biomass harvesting guidelines suggest that 30 percent of residue should remain and be distributed evenly over the site following clearcuts. In addition to final harvests, biomass is also produced though early and mid-rotation thinning of small-diameter trees. This activity is not widespread across Northern Europe, due to operational and economic constraints, with the exceptions being Denmark, some Baltic states, and Finland.[2, 50] Finland subsidizes both early rotation thinnings and the subsequent production of energy in order to support the production of commercial timber products.[53]

The Finnish approach to ensuring forest sustainability is to classify different sites according to the risks associated with biomass removals from these sites and to then apply different management recommendations based on these classifications. Site classifications include: mesic uplands and sites with fertile soils, sub-xeric and xeric sites, barren upland sites with lichens, peatland forest sites, stands with rocky soils, stands with low levels of available nutrients, water conservation areas, managed stands with more than 75 percent spruce, and stands where biomass removals have previously been performed through WTH clearcutting systems.[53]

Finnish guidelines contain operational protocols for site preparation, stump harvests, storing energy wood at roadside, and management of rotten wood.[3] Additional issues addressed include wood production, biodiversity, wildlife habitat, insects and fungi, recreational resources, silviculture, stump harvesting, and biomass production costs (Stupak et al., 2008). Specific recommendations include that large dead trees either standing or on the ground should not to be collected or damaged. Exceptions can be made for certain salvage harvests in the wake of a significant disturbance event, and protocols for this are explicit. Riparian areas must be left unharvested, and the requisite width of riparian management zones depends on site characteristics (e.g., slope of harvesting sites and other watershed characteristics).

In Finland, it is also common and recommended practice to remove stumps and roots in certain circumstances. This is done mainly in spruce stands as a part of preparing the site for the next planting and as a risk-management practice used to avoid root rot.[27, 52] Stump wood cannot be removed from riparian areas or steep slopes unless "preventative measures" are taken. Stumps are also not to be removed from wetlands, sites with rocky soils, dry soils, or thin soils, or if stumps are less than 6 inches in diameter. Stump removal protocols also recommend leaving a certain target number of stumps per acre for different soil types.[21] Finland prohibits in-stand drying of forest residues in late spring and early summer to manage risks associated with bark beetle infestations.

While Finland does not require ash recycling through regulations, it is estimated that more than 10 percent of wood ash produced is typically returned to forests, usually in peat soils where it acts as a fertilizer. Finnish guidelines recommend that wood ash be spread on peat land after thinnings to act as a fertilizer, or if logging residues or stumps are extracted from nutrient-poor sites.[53] Ash is commonly spread with forwarders at a rate of about 3–5 tons per acre every ten years, i.e, slightly more than is recommended in Sweden.[47, 53]

5c. Denmark

Denmark has less forestland than Finland or Sweden but woody biomass is still an important energy source. The Danish Biomass Agreement of 1993 called for increasing the rate of biomass produced for energy (primarily heating) by 1.4 million tons annually, with woody biomass to contribute 0.2–0.4 million tons annually.[52]

In Denmark, whole-tree chipping of small diameter trees from mid-rotation thinning is common; guidelines for public forestry lands recommend that these materials dry for at least two months before they are chipped, to avoid nutrient losses.[47] It is not common practice to harvest slash associated with final clearcut harvests because of the logistical constraints in removing this biomass and/or because of concerns about soil nutrient depletion and impacts to plant and animal communities.[50] Issues addressed in Danish guidance documents include soil fertility, soil organic matter, management of insects and fungi, silviculture, stump harvesting, and production costs.[52, 53] Danish guidance documents classify sites according to the dominance of hardwoods or softwoods and recommend that "stand-wise evaluations" be completed prior to harvests and that forest residues are dried for at least two months during the spring or summer. Other recommendations focus on stands of special conservation value for flora and fauna, and others for which wood production is not a primary objective. Guidance recommends avoiding exposed forest edges, nature conservation areas, and rare forest types.

Danish forest policy generally suggests that nutrients lost in logging may be compensated for through fertilization, and that stumps are not to be removed.[52, 53] Forest policy also suggests that the maximum allowable amount of wood ash that should be applied over ten years ranges from 0.5 to 7.5 tons per hectare, although this depends on the specific chemical composition of the ash.

5d. The United Kingdom

With the UK's biomass-based energy sector growing, the UK Forestry Commission has released a series of technical reference documents designed to help forest managers assess risks associated with biomass harvests.[41, 59, 57, 58] These documents cover slash removal and stump removal as well as the associated risks to soil fertility, soil organic matter, biodiversity and wildlife, hydrology and water quality, archaeological resources, cultural resources, recreational resources, and nature conservation.

The UK biomass harvesting guidance encourages managers to first classify sites according to their susceptibility to risks associated with biomass removal. In 2009, the UK Forestry Commission reevaluated the existing system of site classification used to assess the acceptability of biomass harvests. The previous classification had restricted the overall biomass supply by classifying large portions of the UK as sensitive forestland. The new classification was implemented to facilitate a more reliable biomass supply without adversely impacting natural resources.[58] The guidance classifies sites according to soil types as being of low, medium, or high risk and lists associated slash and stump removal management actions for each of these soil classifications. The assessment of site suitability for biomass harvests is to be based on the most sensitive soil type that covers greater than 20 percent of the site area. The guidelines suggest that site-specific risk assessments should be carried out before each harvest and should include a soil test. The guidance documents also recognize that there are significant uncertainties about the long-term sustainability of removing these materials and suggests that additional research is required to assess the full range of impacts, including net carbon balance.

Photo: UK Forestry Commission

In the UK, biomass harvests typically occur in conifer plantations where slash is windrowed and left for 3–9 months following final timber harvests. This material is subsequently bailed and collected.[58] Thinnings also supply biomass, but this volume is currently not significant. The guidelines suggest that thinnings pose less of an immediate risk to soil nutrient and base cation balance than do final clearcut harvests. In addition to removing timber harvest residues, there is increased interest in harvesting stumps. The UK Forestry Commission recently released interim guidance on stump removal, which states that in some instances the benefits of stump harvesting will outweigh the potential disadvantages, but that the removal of stumps very much requires a site-by-site evaluation. The report acknowledges that stump removal "poses a number of risks to the forest environment that can threaten both sustainable forest management and the wider environment," including soil compaction, rutting, sedimentation, soil carbon loss, removal of macro- and/or micronutrients, and loss of soil buffer capacity due to loss of base cations.[59]

It is important to note that the slash removal guidance states that residue removals are acceptable on all high risk soil types as long as compensatory applications of fertilizer or wood ash are used. The guidelines in turn warn that application of wood ash may induce either nitrogen deficiency on nutrient-poor soils, or leaching of nitrates and/or soil acidification on nitrogen-saturated sites. The guidelines also point out that the application of fertilizers and wood ash may not be acceptable under forest certification programs that have stringent standards for the application of chemicals.

6. OTHER ORGANIZATIONS AND CERTIFICATION SYSTEMS

6a. International Organizations

A number of international organizations have take up the issue of biomass harvest and retention. The International Energy Agency (IEA) conducts research through several programs. For example, Task 43 (feedstocks to energy markets) considers environmental issues, establishment of sustainability standards, exploration of supply chain logistics, and appropriate connections between harvesting standards and international trade and energy markets (www.ieabioenergy.com). The Global Bioenergy Partnership (GBEP) seeks to develop a common methodological framework to measure greenhouse gas emissions from biofuels and to developing science-based benchmarks and indicators for sustainable biofuel production. Throughout 2009, a GBEP task force was focused on the development of a set of relevant, practical, science-based, voluntary criteria and indicators as well as examples of best practices for biomass production. The criteria and indicators are intended to guide nations as they develop sustainability standards and to facilitate the sustainable development of bioenergy in a manner consistent with multilateral trade obligations (www.globalbioenergy.org). The Ministerial Conference on the Protection (MCPC) of Forests is a pan-European process to identify criteria and indicators for sustainability and adaptive management. In 2007, the MCPC initiated a special project to assess the need for sustainability criteria given the increased demand for biomass. The implications of carbon balances on biomass energy are also being explored and may impact the EU's 2009 Renewable Energy Directive (www.foresteurope.org).

6b. Federal Biomass Policy

U.S. federal policy on the use of woody biomass from forests has focused on how to define biomass and how or if sustainable should be legislated. Key areas of legislative focus are the type of wood that qualifies as renewable biomass, what kinds of ownerships can provide woody biomass, and the types of forest from which woody biomass can be procured. The following summary highlights aspects of federal law and proposed legislation that most directly influence the use of woody biomass from forests for energy.

- **Section 45 of the U.S. Internal Revenue Code** The tax code defines what kinds of biomass are eligible for producing energy that qualifies for federal tax incentives such as the federal renewable energy production tax credit and investment tax credit. "Closed-loop biomass" is defined as "any organic material from a plant which is planted exclusively for purposes of being used at a qualified facility to produce electricity," whereas "Open-loop biomass" includes a number of opportunity fuels, such as "any agricultural livestock waste nutrients," "any solid, nonhazardous, cellulosic waste material or any lignin material which is derived from...mill and harvesting residues, pre-commercial thinnings, slash, and brush," a variety of "solid wood waste materials," and agricultural biomass sources.

- **Farm Security and Rural Investment Act of 2002 Public Law 107–171—May 13, 2002** This law included both "trees grown for energy production" and "wood waste and wood residues" in its definition of biomass.

- **Energy Policy Act of 2005 Public Law 109–58—Aug. 8, 2005** The Energy Policy Act defined biomass to include "any of the following forest-related resources: mill residues, pre-commercial thinnings, slash, and brush, or non-merchantable material," as well as "a plant that is grown exclusively as a fuel for the production of electricity." This definition was more detailed than the previous 2002 Farm Bill and excluded material that would traditionally sell as timber.

- **The Energy Independence and Security Act of 2007 Public Law 110–140—Dec. 19, 2007** The Energy Independence and Security Act included the Renewable Fuels Standard (RFS) and provided the most detailed definition of biomass to date. One of the most important distinctions it made was to separate woody biomass from private and federal lands. Biomass from federal lands was excluded and could not be used to produce renewable fuels. However, an exception was provided for woody biomass removed from the "immediate vicinity of buildings" for fire protection. The RFS also excluded biomass from certain types of forests seen as rare: "ecological communities with a global or state ranking of critically imperiled, imperiled, or rare pursuant to a State Natural Heritage Program, old growth forest, or late successional forest." The RFS made an effort to discourage conversion of native forests to plantations by excluding woody biomass from plantations created after the enactment of the law. The RFS also established a subsidy of up to $20 per green ton of biomass delivered for facilities producing electric energy, heat, or transportation fuels.

- **Food, Conservation, and Energy Act of 2008 Public Law 110–246—June 18, 2008** The 2008 Farm Bill continued the trend toward great specification in the definition of renewable biomass. This time woody biomass from federal lands was included where it was the byproduct of preventive treatments to reduce hazardous fuels, contain disease or insect infestation; or restore ecosystem health. On private lands, the definition included essentially all trees and harvest residues. The exclusion for rare forests in the 2007 RFS was not included. The 2008 Farm Bill also initiated the **Biomass Crop Assistance Program (BCAP)** to improve the economics of establishing and transporting energy crops and collecting and transporting forest biomass. Regarding eligibility requirements for this program, forest lands producing biomass must be covered by a "forest management plan." The determination of what constitutes an "acceptable plan" is at the discretion of the State Forester.

Other legislation has been proposed that includes more specific provisions designed to ensure the sustainability of biomass production. For example, HR 2454 would require that biomass from federal land be "harvested in environmentally sustainable quantities, as determined by the appropriate Federal land manager." S 1733, introduced September 9, 2009, stipulates that biomass be produced while ensuring "the maintenance and enhancement of the quality and productivity of the soil" and promoting the "well-being of animals." The future fate of the federal biomass definition is likely to be part of the large climate-change legislation being debated in Washington. Climate-change legislation may include a national Renewable Energy Standard (i.e., a renewable portfolio standard) that would dictate what kind of woody biomass can be included to meet renewable electricity generation goals. Some proposals would shift the burden of sustainability to the states and require biomass harvesting guidelines or regulations that meet some federal oversight.

6c. Forest Stewardship Council: U.S. National Forest Management Standard

Photo: Zander Evans

The FSC standards for the U.S. do not specifically address biomass or whole tree harvests. In other words, "biomass and whole tree harvests are addressed along with other types of removals."[23] The FSC U.S. National Standard covers biomass harvesting at a more general level than most state guidelines, since they are nationwide. The main sections that affect biomass harvest are Criterion 6.2 (habitat for rare species), 6.3 (ecological functions), and 6.5 (soils and water quality). For example, Indicator 6.3.f of the guidelines requires that "management maintains, enhances, or restores habitat components and associated stand structures, in abundance and distribution that could be expected from naturally occurring processes"; these habitat components include "live trees with decay or declining health, snags, and well-distributed coarse down and dead woody material." This proposed requirement would place some limits on biomass removal, but it is not specific about the amount of DWM that should be retained on-site. Indicator 6.5.c limits multiple rotations of whole tree harvesting to sites where soil productivity will not be harmed.

Since FSC guidelines are not focused solely on biomass harvests, they go beyond other biomass guidelines in areas such as habitat connectivity. By the same token, because FSC guidelines cover many different kinds of harvests in many different forest types with diverse forest management objectives, the standards do not contain many subtopics that are specific to biomass harvest (Appendix I).

The FSC standards are considered "outcome focused." Rather than prescribing how to achieve desired outcomes, they allow a variety of practices to be used, so long as the management objectives and the FSC standards are not compromised. For example, one element that shows up in some biomass guidelines is re-entry, but FSC does not include this. Missouri's guidelines advise, "Do not re-enter a harvested area [for the purposes of biomass harvesting] once the new forest has begun to grow," in order to reduce the risk of compaction, which is a recommendation echoed in the Minnesota and Pennsylvania guidelines. The FSC standards, however, do not specifically advise against re-entering a stand for the purpose of biomass harvesting. Instead, issues of compaction and the impacts of other soil disturbing activities are addressed in relation to all management activities under both 6.5 and 6.3.

6d. Other Voluntary Certification Systems

Other voluntary certification systems have standards which may influence forest biomass harvest and retention. For example, the Council for Sustainable Biomass Production (CSBP) released draft standards in 2009 and plans to release a preliminary standard in 2010.[14] The draft standards were open for stakeholder and expert review and comment. The CSBP standards address soil, biological diversity, water, and climate change. As with FSC standards, CSBP makes general recommendations such as "retain biomass materials required for erosion control and soil fertility" (1.1.S3), but do not provide specific guidance on retention of DWM or snags.

7. COMMON ELEMENTS OF BIOMASS HARVESTING GUIDELINES

Though the existing biomass guidelines cover different ecosystems, they share a number of important elements. The following sections assess the similarities and differences between the guidelines' recommendations on dead wood, wildlife and biodiversity, water quality and riparian zones, soil productivity, and silviculture. In addition, we compare the process used to develop each set of guidelines.

7a. Dead Wood

One of the central concerns in biomass removal is the reduction of the quantity of dead wood on-site. Maine's guidelines recommend leaving tops and branches scattered across the harvest area "where possible and practical." To ensure sufficient DWM debris is left on-site, Michigan's draft guidelines recommend retention of one-sixth to one-third of the residue less than four inches in diameter. Minnesota guidelines recommend leaving all preexisting DWM and to "retain and scatter tops and limbs from 20 percent of trees harvested." Wisconsin's guidelines recommend retaining all pre-harvest DWM and tops and limbs from 10 percent of the trees in the general harvest area, with a goal of at least 5 tons of FWM per acre. Wisconsin's guidelines also point out that "some forests lack woody debris because of past management," and that extra DWM should be left in those areas. Pennsylvania's guidelines suggest leaving 15 to 30 percent of "harvestable biomass" as DWM, while Missouri's suggest 33 percent of harvest residue (with variations for special locations such as stream sides).

Maine, Minnesota, Pennsylvania, and Wisconsin suggest leaving all snags possible. Except for some hazard exceptions, California requires retention of all snags. Missouri provides an example of clear and specific recommendations by suggesting 6 per acre in upland forests and 12 per acre in riparian corridors. Michigan does not have a specific recommendation for snag retention.

7b. Wildlife and Biodiversity

Many of the potential wildlife and biodiversity impacts stem from leaving too little dead wood on-site. The biomass guidelines reviewed here agree on the importance of avoiding sensitive sites for wildlife. These include areas of high biodiversity or high conservation value such as wetlands, caves, and breeding areas. Obviously, areas inhabited by threatened or endangered animals and plants receive special consideration. However, as the Minnesota guidelines point out, biomass harvesting may still be appropriate if management plans include specific strategies for maintaining habitat for rare species and/or to restore degraded ecosystems. Pennsylvania's guidelines suggest that biomass removal may be an opportunity to "develop missing special habitats, such as herbaceous openings for grouse and other species, through planting, cutting, or other manipulations." Additional suggestions from state guidelines include inventorying habitat features on the property, promoting individual trees and species that provide mast, and retaining slash piles that show evidence of use by wildlife. Missouri's guidelines make the case against forest conversion in terms of wildlife: "Do not convert natural forests into tree plantations or pasture; natural forests provide more wildlife food and habitat."

7c. Water Quality and Riparian Zones

Photo: Zander Evans

In general, water quality and riparian concerns do not change with the addition of biomass removals to a harvest plan. Streams and wetlands tend to be protected by existing regulation. For example, Maine's guidelines cite the existing laws governing water quality protection as well as the publication *Protecting Maine's Water Quality*. Where restrictions in wetlands and riparian zones are defined in terms of basal area, more specific guidance may be needed for biomass harvests, which can have a large ecological impact with a small change in basal area. An example of riparian recommendations from Minnesota's guidelines is to "avoid harvest of additional biomass from within riparian management zones over and above the tops and limbs of trees normally removed in a roundwood harvest under existing timber harvesting guidelines." Though the Missouri Watershed Protection Practice already includes requirements for stream and river management zones, the Missouri biomass guidelines reiterate how to protect streams and rivers during a harvest.

7D. Soil Productivity

As with water quality, some aspects of soil productivity are usually included in standard forestry BMPs. For instance, Minnesota's biomass guidelines point readers to the state's timber harvesting guidelines, which contain sections titled "Design Outcomes to Maintain Soil Productivity" and "Minimizing Rutting." However, Minnesota's biomass guidelines do add warnings about harvesting biomass on bog soils and shallow soils (less than 8 inches) over bedrock. An appendix to Wisconsin's guidelines lists over 700 specific soil map units which are nutrient poor and unlikely to be able to support sustainable biomass removal. Maine's guidelines use the Briggs classification of soil drainage classes to identify sites that are more sensitive to biomass removals.9 Missouri's guidelines contain a specific section on sustaining soil productivity, especially on steep slopes and shallow soils. Michigan recommends leaving more than one-third of harvested tops on shallow, nutrient-poor or semi-organic soils. However, Michigan's guidelines suggest that the amount of retention can be reduced on jack pine stands on nutrient poor sites.

Another concern that arises with biomass harvest is removal of the litter layer or forest floor. Maine, Michigan, Minnesota, Pennsylvania, and Wisconsin's guidelines state that forest floor, litter layer, stumps, and root systems should all be left.

7E. Silviculture

Many silvicultural prescriptions call for the removal of small, unhealthy, or poorly formed trees to open up more growing space for crop trees or regeneration, but these types of removals often cost money rather than generate income. By providing income from the removal of this material, biomass markets can help support good silviculture. At the same time, biomass removals raise some silvicultural concerns. The Minnesota guidelines point out that an increase in the amount of live vegetation removed may cause swamping, i.e., a decrease in transpiration and an increase in soil moisture. Swamping can kill seedlings and negatively impact regeneration. Removal of tree tops and branches may also remove seeds or cones, which may reduce the amount of natural regeneration. Biomass removals can help deal with forest insect problems, but removing the biomass material from the site must be timed to avoid contributing to pest problems such as bark beetles.

Some states have used biomass guidelines to make silvicultural recommendations that may improve stands but are not directly related to biomass harvesting. The Missouri biomass guidelines provide silvicultural suggestions for the number of crop trees per acre for stands in different stages of development. Pennsylvania's guidelines suggest that forest stewards "provide for regeneration each time harvests are made under the uneven-aged system," focus on the residual stand more than the trees being removed, and avoid high grading. Wisconsin's guidelines suggest retaining "reserve trees and patches at 5–15 percent crown cover or stand area" in even-aged regeneration cuts and three or more large-cavity trees, large mast trees, and trees that can become large trees in the future. Maine's guidelines recommend retention of cavity and mast trees while Wisconsin's guidelines recommend retaining five percent of the area unharvested in salvage operations following severe disturbances.

Another operational recommendation that Minnesota, Missouri, and Pennsylvania all make is to avoid re-entering a stand to remove biomass. Re-entering a site where timber was recently harvested can increase site impacts such as soil compaction and harm post-harvest regeneration. For this reason, the Missouri guidelines advise that "woody biomass should be harvested at the same time as sawlog timber to avoid re-entry." Maine's guidelines recommend that woody biomass removal be integrated with traditional forest operations where possible.

7F. Biomass Guidelines Development

The process of developing guidelines can be as important as the specific recommendations. Most guidelines try to draw from the most recent forest science. Developing new biomass guidelines allows states to incorporate new research and ideas. Minnesota used funding from the University of Minnesota Initiative for Renewable Energy and the Environment to conduct a review of the scientific literature on biomass harvests. Other guidelines borrow from existing guidelines. For example, Pennsylvania's guidelines borrow extensively from Minnesota's guidelines and summarize the FSC's standards for the region.

The amount of stakeholder participation varies across the guidelines. While Pennsylvania's guidelines were created from within the DCNR, Minnesota, Missouri, and Wisconsin included public participation and a technical committee from the wider forestry community. Public participation can be unwieldy, but often generates greater public support for forestry projects.[20]

Some of the biomass guidelines, such as those from New Brunswick, Canada, focus on the identification of geographies where biomass harvesting is most appropriate. Wisconsin takes a complementary approach, identifying soil types where biomass removal is inappropriate. By mapping soil types, guidelines can highlight those areas where concerns about nutrient depletion are lowest. Suitability mapping also permits the consideration of the landscape-scale impacts of biomass harvesting. Pennsylvania's guidelines are notable because they consider the supply of biomass from forests as well as the appropriate scale of utilization. As mentioned previously, Pennsylvania's guidelines make a case for small-scale (less than 2,000 tons of biomass per year) biomass utilization facilities.

8. CONCLUSION

This revised assessment of biomass guidelines reviews a wide range of approaches to the sustainable use of biomass that can inform the development of guidelines in Massachusetts. The section on New York and New England may be the most helpful, because these states are dealing with similar timber types and land ownership patterns. However, there are number of other state-based approaches, such as those of Minnesota and Michigan, that can be readily transferable. Northern Europe has a long history of intensive biomass use, and while their harvesting systems and approach to forest management are currently very different, their approaches to ecological issues can be translated to concerns in Massachusetts. The sections on other organizations and federal policy provide insight into how Massachusetts guidelines might be influenced or integrated with other approaches.

The final section, which explores the common elements of biomass harvesting guidelines, offers a structure to develop guidelines tailored to Massachusetts. The Forest Guild has used that structure to develop a set of guidelines, *Biomass Retention and Harvesting Guidelines for the Northeast*, which is readily applicable to Massachusetts. These guidelines are included as a separate document.

The following recommendations for the development of future biomass guidelines in Massachusetts are based on the existing guidelines and available science, and will change as more is learned about biomass removals:

- Develop guidelines that are based on sound science and include wide stakeholder engagement. As the Minnesota guidelines describe it, "Provide the best scientific judgment, tempered by the consensus process among a broad group of forest management interests, related to practices that will sustain a high level of biodiversity."
- Define "woody biomass" and other important terms clearly.
- Base biomass harvesting recommendations on local ecology. They should recognize state or local natural communities, disturbance regimes, and other ecological traits. Technical committees and scientific literature provide a firm base for harvest recommendations.
- Consider developing guidelines for each of the subtopics listed in Appendix I—though not all subtopics will be appropriate for every location.
- Make clear and specific recommendations for the retention of standing dead trees, existing CWM, CWM generated by the harvest, FWM, and forest floor and litter layer. Because reduction of dead wood is one of the key differences between biomass removal and traditional harvest, it should be a focus of future guidelines. Nutrients removed from the site should be replenished. For even-aged systems, nutrients should be replenished to adequate levels by the end of the rotation. Uneven-aged systems should maintain nutrient levels close to the optimum. Nutrient levels may be temporarily reduced after each entry, but should return to adequate levels by the next cutting cycle.
- Make biomass guidelines practical and easy to follow. Where biomass guidelines supplement existing forestry rules and guidelines, the new guidelines should provide clear references to the relevant sections of the existing rules and guidelines both for convenience and to increase the likelihood of implementation.
- Take advantage of the opportunity to create new forestry recommendations that encourage excellent forestry: forestry that goes beyond minimum BMPs and enhances the full suite of ecological values. For example, biomass guidelines may be an opportunity to suggest alternatives to high grading and other practices that damage the long-term health of the forest. Similarly, biomass guidelines can present the chance to advocate for appropriately scaled biomass utilization, as Pennsylvania guidelines already do.

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11. APPENDIX I

SUMMARY TABLE OF BIOMASS GUIDELINES

	ME	MN	MO	PA	WI	FSC
Dead Wood						
Coarse woody material	√	√	√	√	√	√
Fine woody material	√	√	√	√	√	√
Snags	√	√	√	√	√	√
Wildlife and Biodiversity				√		
Wildlife	√	√	√	√	√	√
Sensitive wildlife species	√	√	√	√	√	√
Biodiversity	√	√	√	√	√	√
Plants of special concern	√	√	√	√	√	√
Sensitive areas	√	√	√	√	√	√
Water Quality and Riparian Zones						
Water quality	√	√	√	√	√	√
Riparian zones	√	√	√	√	√	√
Non-point source pollution	√	√	√	√	√	√
Erosion	√	√	√	√	√	√
Wetlands	√	√	√	√	√	√
Soil Productivity						
Chemical (Nutrients)	√	√	√	√	√	√
Physical (Compaction)	√	√	√	√	√	√
Biological (Removal of litter)	√	√		√	√	
Silviculture						
Planning	√	√	√	√		√
Regeneration		√		√	√	√
Residual stands	√	√	√	√	√	√
Aesthetics			√	√	√	√
Post operations	√	√	√	√	√	
Re-entry		√	√	√		
Roads and skid trail layout	√	√	√	√	√	√
Disturbance						
Insects		√	√	√	√	√
Disease			√	√	√	√
Fire		√	√	√		√
Fuel reduction		√		√		√
Pesticides		√		√		
Invasives		√	√	√		
Conversion from forest			√	√		√

12. APPENDIX II

LINKS TO BIOMASS HARVESTING GUIDELINES

- Considerations and Recommendations for Retaining Woody Biomass on Timber Harvest Sites in Maine http://www.maine.gov/doc/mfs/pubs/biomass_retention_guidelines.html

- Minnesota: Biomass Harvesting Guidelines for Forestlands http://www.frc.state.mn.us/FMgdline/BHGC.html

- Missouri: Best Management Practices for Harvesting Woody Biomass http://mdc4.mdc.mo.gov/applications/MDCLibrary/MDCLibrary2.aspx?NodeID=2055

- Pennsylvania: Guidance on Harvesting Woody Biomass for Energy http://www.dcnr.state.pa.us/PA_Biomass_guidance_final.pdf

- Wisconsin Council on Forestry: Use of Woody Biomass http://council.wisconsinforestry.org/biomass/

- Forest Stewardship Council http://www.fscus.org/standards_criteria/

- Canada: The Scientific Foundation for Sustainable Forest Biomass Harvesting Guidelines and Policies http://www.sfmnetwork.ca/html/biomass_workshop_e.html

- New Brunswick: Forest Biomass Harvesting Policy http://www.gnb.ca/0078/Policies/FMB0192008E.pdf

APPENDIX 4–C

FOREST BIOMASS RETENTION AND HARVESTING GUIDELINES FOR THE NORTHEAST

TABLE OF CONTENTS

1. INTRODUCTION AND BACKGROUND

Interest in removing wood with a historically low economic value from forests has increased because of rising fossil fuel costs, concerns about carbon emissions from fossil fuels, and the risk of catastrophic wildfires. Even as federal, state and regional programs encourage the utilization of forest biomass, there are concerns about its potential adverse effects on biodiversity, soil productivity, wildlife habitat, water quality, and carbon storage. At the same time, biomass removal and utilization have the potential to provide a renewable energy source, promote the growth of higher-value trees and forest products, reduce forest fire risk, support the removal of invasive species, and help to meet the economic development goals of rural communities. These guidelines are designed to encourage protection of soils, wildlife habitat, water, and other forest attributes when biomass or other forest products are harvested in the Northeastern United States.

Our Principles

1. The well-being of human society is dependent on responsible forest management that places the highest priority on the maintenance and enhancement of the entire forest ecosystem.

2. The natural forest provides a model for sustainable resource management; therefore, responsible forest management imitates nature's dynamic processes and minimizes impacts when harvesting trees and other products.

3. The forest has value in its own right, independent of human intentions and needs.

4. Human knowledge of forest ecosystems is limited. Responsible management that sustains the forest requires a humble approach and continuous learning.

5. The practice of forestry must be grounded in field observation and experience as well as in the biological sciences. This practical knowledge should be developed and shared with both traditional and non-traditional educational institutions and programs.

6. A forester's or natural resource professional's first duty is to the forest and its future. When the management directives of clients or supervisors conflict with the Mission and Principles of the Guild, and cannot be modified through dialogue and education, a forester or natural resource professional should disassociate

The Forest Guild Guidelines

The Forest Guild guidelines are designed to augment and enhance existing Best Management Practices (BMPs) or new state-based biomass guidelines that may, in some cases, leave managers and policy makers looking for more detailed recommendations. While these guidelines were developed to address biomass harvesting, they also are intended to inform all harvests in northeastern forests. We developed these guidelines to assist several audiences: field foresters, loggers, state-based policy makers charged with developing biomass guidelines and standards, biomass facilities wishing to assure sustainability, third party certifiers, and members of the public interested sustainable forest management.

These guidelines are based on the Forest Guild's principles (see text box). Forest Guild members are concerned with reconciling biomass removals with the principles of excellent forestry—forestry that is ecologically, economically, and socially responsible. Excellent forestry exceeds minimum best management practices and places the long-term viability of the forest above all other considerations. It uses nature as a model and embraces the forest's many values and dynamic processes. Excellent forestry maintains the functions, structures, and composition that support the health of the entire forest ecosystem. Excellent forestry is different in each ecoregion, but is guided by science, place-based experience, and continuous learning.

Forest Guild members acknowledge their social responsibilities as forest stewards to address climate change and mitigate the buildup of atmospheric carbon. In addition, we understand how renewable fuels derived from well-managed forests can provide energy security and enhance rural communities. At the same time, we have an ecological imperative to ensure that all our harvests—including biomass harvests—maintain or enhance the ecological values of the forest.

Creating the Guidelines

Our working group consisted of 23 Forest Guild members representing public and private field foresters and resource managers, academic researchers and members of major regional and national environmental organizations. The process was led by Forest Guild staff and was supported by two Forest Guild reports: *Ecology of Dead Wood in the Northeast 4 and An Assessment of Biomass Harvesting Guidelines*.5 Wherever possible we base our recommendations on peer-reviewed science. However, in many cases research is inadequate to connect practices, stand level outcomes, and ecological goals. Where the science remains inconclusive, we rely on field observation and professional experience. The guidelines provide both general guidance and specific targets that can be measured and monitored. These guidelines should be revisited frequently, perhaps on a three-year cycle, and altered as new scientific information and results of field implementation of the guidelines become available.

"Sustainability" and Biomass Harvesting

Using a common definition, sustainable biomass harvests would "meet the needs of the present without compromising the ability of future generations to meet their needs" (Brundtland Commission 1987). Crafting a more precise definition of sustainable forest management is inherently complex because forest ecosystems are simultaneously intricate, dynamic, and variable. Sustainable forest management must integrate elements of ecology, economics, and societal well being. These guidelines primarily pertain to issues of sustaining ecological function and productivity; they are not meant to replace a comprehensive assessment of forest sustainability.

In general, the sustainability of managed forests must be judged on timelines that span generations. Individual trees can persist for centuries and management decisions made today will have important implications well beyond the tenure of any one manger. The indigenous focus on the impact of decisions seven generations into the future is more appropriate. Similarly, sustainability must be judged on scales larger than that of the individual forest stand. For example, large mammal home ranges, water quality, and a viable forestry industry all depend on landscapes that encompass multiple stands. Due to the difficulties of defining appropriate time frames and spatial scales, the concept of forest sustainability is best thought of as an adaptive process that requires regular monitoring and recalibration. Consequently, these guidelines are presented not as static targets to be maintained at all times in all places, but rather as guideposts on a path to sustainability.

Definitions

Biomass

In a scientific context, the term "biomass" includes all living or dead organic matter. In common parlance, biomass usually refers to woody material that has historically had a low value and was not considered merchantable in traditional markets. Biomass harvesting can also involve the removal of dead trees, downed logs, brush, and stumps, in addition to tops and limbs. Changing markets and regional variations determine which trees are considered sawtimber or pulpwood material and which are relegated to the biomass category. This report does not discuss biomass from agricultural lands and short-rotation woody biomass plantations.

In this report, the term biomass refers to *vegetation removed from the forest, usually logging slash, small-diameter trees, tops, limbs, or trees not considered merchantable in traditional markets*. Similarly we use the phrase **biomass harvesting** to refer to the *removal of logging slash, small-diameter trees, tops, or limbs*.

Biomass can be removed in a number of ways. Some harvests remove only woody biomass, some combine the harvest of sawtimber or other products with biomass removal, and some remove biomass after other products have been removed. This report focuses on post-harvest forest conditions and not on the type of harvest. The goal is to ensure the forest can support wildlife, maintain biodiversity, provide clean water, sequester carbon, protect forest soil productivity, and continue to produce income after a biomass harvest or repeated harvests. In some regions, current wood utilization is such that very little woody material is available for new markets such as energy. For these high-utilization areas, application of these guidelines may result in more biomass being left in the forest.

Downed Woody Material

Woody material is sometimes divided into coarse woody material (CWM) and fine woody material (FWM). CWM has been defined as more than 6 inches in diameter at the large end and FWM that is less than 6 inches in diameter at the large end.17 The USDA Forest Service defines CWM as downed dead wood with a small-end diameter of at least 3 inches and a length of at least 3 feet, and FWM as having a diameter of less than 3 inches.25 FWM has a higher concentration of nutrients than CWM. Large downed woody material, such as logs greater than 12 inches in diameter, is particularly important for wildlife. In this report, we use the term **downed woody material (DWM)** to encompass all three of these size classes, but in some circumstances we discuss a specific size of material where the piece size is particularly important.

2. GUIDELINES FOR BIOMASS RETENTION AND HARVESTING FOR ALL FOREST TYPES

The following recommendations are applicable across a range of forest types in the Northeast. However, different forest types naturally develop different densities of snags, DWM, and large downed logs. Unfortunately, even after an exhaustive review of the current science there is too much uncertainty to provide specific targets for each forest type. The recommendations in this section set minimum retention targets necessary for adequate wildlife habitat and to maintain the integrity of ecological process such as soil nutrient cycling. Wherever possible, exceed the targets as a buffer against the limitations of current research. Section 3 presents research that may help landowners and foresters interested in additional tree, snag, and DWM retention tailored to specific forest types.

Site Considerations to Protect Rare Forests and Species

- Biomass harvests in critically imperiled or imperiled forest types (i.e., globally recognized or listed as S1 or S2 in a State National Heritage Program) should be avoided unless necessary to perpetuate the type. Management of these and other rare forest types (for example, those ranked S3 by state Natural Heritage Programs) should be based on guidance from the local Natural Heritage Program and/or other local ecological experts.

- Biomass harvesting may be appropriate in sensitive sites to control invasive species, enhance critical habitat, or reduce wildfire risk. However, restoration activity should be guided by ecological goals and not designed solely to supply biomass. It is unlikely that restored sites will contribute to the long-term wood supply, because biomass removals for restoration may not be repeated at regular intervals.

- Old growth forest stands with little or no evidence of harvesting are so rare in the Northeast that they should be protected from harvesting, unless necessary to maintain their structure or ecological function. Areas with scattered old growth trees or late-successional forest characteristics should be carefully managed to ensure retention of their ecological functions. Biomass generally should not be removed from these areas.

Retention of Downed Woody Material

Though CWM represents a large pool of nutrients in some ecosystems, it likely plays a relatively small role in nutrient cycling for managed Northeastern forests. A review of scientific literature suggests that biomass harvesting is unlikely to cause nutrient problems when both sensitive sites (including low-nutrient sites) and clearcutting with whole-tree removal are avoided (see Evans and Kelty 2010 for a more detailed discussion of the relevant scientific literature). However, there is no scientific consensus on this point because of the limited range of treatments and experimental sites.

Maintenance of Soil Fertility

Biomass harvesting on low-nutrient sites is a particular concern. For example, Hallett and Hornbeck note that "red oak and white pine forests growing on sandy outwash sites are susceptible to nutrient losses due to inherently low-nutrient capitals and/or nutrient depletion by past activities such as farming, fire, and intensive harvesting."9 Maine's *Woody Biomass Retention Guidelines*1 list shallow-to-bedrock soils, coarse sandy soils, poorly drained soils, steep slopes, and other erosion-prone sites as sensitive to biomass removals. We encourage states to identify low-nutrient soil series where biomass harvesting should not occur and those soil series where biomass harvests require particular caution. Wisconsin's *Forestland Woody Biomass Harvesting Guidelines* is an excellent example.11

In areas that do not qualify as low-nutrient sites, where 1/3 of the basal area is being removed on a 15- to 20-year cutting cycle, it is our professional judgment that retaining 1/4 to 1/3 of tops and limbs will limit the risk of nutrient depletion and other negative impacts in most forest and soil types. Additional retention of tops and limbs may be necessary when harvests remove more trees or harvests are more frequent. Similarly where the nutrient capital is deficient or the nutrient status is unknown, increased retention of tops, branches, needles, and leaves is recommended. Conversely, if harvests remove a lower percentage of basal area, entries are less frequent, or the site is nutrient-rich, then fewer tops and limbs need to be retained on-site.

Guidelines for DWM Retention

- In general, when 1/3 of the basal area is being removed on a 15 to 20 year cycle, retain 1/4 to 1/3 of the slash, tops, and limbs from harvest (i.e., DWM).
- Three main factors influence the percentage of tops and limbs that should be left onsite:
  - number of live trees left on-site,
  - time between harvests, and
  - available soil nutrients.
- As harvesting intensity increases (and the three preceding factors decrease) more slash, tops, and limbs from harvests should be left on-site
- As harvesting intensity decreases (and the three factors increase) less slash, tops, and limbs from harvests are required to protect site productivity.
- Avoid harvesting on low-nutrient sites or adjust retention of tops, branches, needles, and leaves.
- Retain DWM of all sizes on-site including FWM, CWM and large downed logs.
- In general, leave DWM distributed across the harvest site. However, there may be cases where piles of DWM provide habitat, or redistribution of DWM collected at the landing would cause excessive damage to soil or regeneration.
- Minimize the removal of needles and/or leaves by harvesting in winter, retaining FWM on-site, or leaving felled trees on-site to allow for needle dro

Retention of Forest Structures for Wildlife and Biodiversity

- Leave and protect litter, forest floor, roots, stumps, and large downed woody material.
- Leave and protect live cavity trees, den trees, other live decaying trees, and snags (i.e., dead standing trees >10"). Individual

snags that must be felled for safety requirements should not be removed from the forest.

Table 1. General Guidelines for Retaining Forest Structures

Structure	Minimum Target (per acre)		Considerations
	Number	Basal area (ft²)	
Live decaying Trees 12 –18 inches DBH	4	3–7	Where suitable trees for retention in these size classes are not present or may not reach these targets due to species or site conditions, leave the largest trees possible that will contribute toward these targets.
Live decaying trees >18 inches DBH	1	2	
Snags >10 inches DBH	5	3	Worker safety is top priority. Retain as many standing snags as possible, but if individual snags must be felled for safety reasons, leave them in the forest.

Table 1 is based on the scientific literature review in *The Ecology of Dead Wood in the Northeast*4 as well as other biomass harvesting and retention guidelines[5]. These guidelines are not meant to be attained on every acre, at all times. Rather, they are average targets to be applied across a stand, harvest block, or potentially an ownership.

- If these forest structures do not currently exist, select and identify live trees to become these structures in the future. Retaining live decaying trees helps ensure sufficient snags in the future. Similarly, both decaying trees and snags can eventually become large downed logs.
- If forest disturbances such as hurricanes, ice storms, and insect infestations create large areas of dead trees, leaving all snags or decaying trees may be impractical. If an area is salvage logged, leaving un-salvaged patches totaling 5% to 15% of the area will provide biological legacies important to wildlife. However, the potential for insect populations to build up in dead trees may prohibit retention of unsalvaged patches in some situations.
- Since there are differences in decay rates and wildlife utilization, retain a variety of tree species as snags, DWM, and large downed logs.
- In areas under even-aged management, leave an uncut patch within or adjacent to every 10 acres of regeneration harvest. Uncut patches, including riparian buffers or other set-asides within the management unit, should total 5% to 15% of the harvest area.
- Build retention patches around large legacy trees, den or cavity trees, large snags, and large downed logs, to maximize structural and habitat diversity.
- Marking retention trees will help ensure that sufficient numbers are retained during the current harvest, and that and they will not be removed in subsequent harvests.
- Management that maintains multiple vegetation layers, from the overstory canopy to the midstory, shrub, and ground layers will benefit wildlife and plant species diversity.

Water Quality and Riparian Zones

In general, water quality and riparian concerns do not change with the addition of biomass removals to a harvest plan. Refer to state water quality best management practices (BMPs) and habitat management guidelines for additional measures to protect streams, vernal pools, and other water bodies (see Appendix I for a list of these BMPs and habitat management guidelines).

- DWM retention described above is also important for water quality, because DWM reduces overland flow and holds water.
- Leave and protect existing woody material in streams, ponds, and lakes. DWM in riparian systems provides sites for vegetation colonization, forest island growth and coalescence, and forest floodplain development.
- Leave and protect live decaying trees (e.g., cavity/den trees), snags, and large downed logs in riparian or stream management zones.
- Keep vernal pools free of slash, tops, branches, and sediment from forestry operations. If slash falls into the pool during the breeding season, it is best to leave it in place to avoid disturbing egg masses or other breeding activity that may already be occurring.
- Within 100 feet of the edge of a vernal pool, maintain a shaded forest floor to provide deep litter and woody debris around the pool. Also avoid ruts, bare soil, or sources of sediment near vernal pools.
- Extra care should be taken working in or around forested wetlands because of their importance for wildlife and ecosystem function. Wetlands are often low-fertility sites and may support rare natural communities, so removal of DWM may be inappropriate.

Harvesting and Operations

Most concerns about the operational aspects of biomass harvesting are very similar to all forestry operations. However, some key points are worth emphasizing:

- Protect forest land from conversion to non-forest use and native forest from conversion to plantations.
- Involve a professional forester (or a licensed forester in states where available) in development of a long-term management plan and supervision of harvests.
- Engage a certified logger from the Master Logger Certification Program or other similar program when harvesting.
- Follow all best management practices (BMPs) for the state or region.
- Plan and construct roads and skid trails based on professional advice and BMPs.
- Integrate biomass harvesting with other forest operations. Re-entering a site where timber was recently harvested to remove

biomass can increase site impacts such as soil compaction and may harm post-harvest regeneration.

- Use low impact logging techniques such as directional felling or use of slash to protect soil from rutting and compaction from harvest machines.
- Use appropriate equipment matched to site and operations.

3. RELEVANT RESEARCH FOR NORTHEASTERN FOREST TYPES

Although there is too much scientific uncertainty to provide specific targets for each forest type, the research described below may help landowners and foresters interested in additional tree, snag, and DWM retention tailored to specific forest types. We hope the need to better quantify decaying tree, snag, and DWM retention requirements will catalyze new research efforts and the retention target can be updated based on new science.

Measurements of Downed Woody Material

Most of the scientific research measures DWM in terms of dry tons per acre rather than percentage of DWM retained after harvest. Tons per acre may not currently be a useful measurement unit for forester and loggers, but we present data in those units here because of their prevalence in scientific literature. This measurement unit may become more prevalent as biomass harvesting increases. Field practitioners typically have not paid a great deal attention to volumes of DWM. Measurement techniques are available to integrate DWM sampling into forest inventories; over time, field practitioners will develop an awareness of volumes-per-acre of DWM, similar to standing timber volumes. The Natural Fuels Photo Series illustrates various levels of DWM and can be used to assist this process (http://depts.washington.edu/nwfire/dps/).

In general, stands have the most DWM when they are young (and trees are rapidly dying from competition) or when they are old (and trees are in various states of decline). Healthy, intermediate-aged stands tend to have less DWM. The following table represents a target range for the mass of DWM left on-site after harvest (including both existing and harvest-generated DWM). The table is based on a number of studies that documented the ranges of observed DWM in managed and unmanaged stands in the Northeast (see Evans and Kelty 2010 for more details). The selected target ranges reflect measurements from unmanaged stands more than those from managed stands and take into account patterns of DWM accumulation during stand development.

Table 2. DWM Ranges by Forest Type

	Northern HW	Spruce-Fir	Oak-Hickory	White and Red Pine
Tons of DWM per acre*	8–16	5–20	6–18	2–50

** Includes existing DWM and additional material left during harvesting to meet this target measured in dry tons per acre.*

Spruce–Fir Forests

Research data on DWM in Maine's spruce-fir forest include 3.4 tons per acre10 and a range from 22 to 117 tons per acre.[20] The low estimate of 3.4 tons per acre is from a survey that includes intensively-managed lands that may not have enough DWM to maintain ecosystem processes and retain soil nutrients,10 while the higher estimates come from unmanaged lands.[20]

The basal area of dead trees from a survey of paper birch-red spruce-balsam fir and red spruce-balsam fir stands ranged from 11 to 43 percent of stand basal area.[23] The Canadian province of Newfoundland and Labrador requires retention of 4 snags per acre, while Maine recommends retaining 3 snags and/or cavity trees greater than 14 inches DBH and one greater than 24 inches DBH.[6, 19] Smith and colleagues recommend retention and recruitment of white birch snags to ensure sufficient snag and DWM density.19 Other guidelines recommend between 5 and 6 snags per acre greater than 8 inches DBH and an additional 4 to 6 potential cavity trees at least 10 inches DBH.[26]

Northern Hardwood Forests

Measures of the DWM in northern hardwood forests are as low as 3.1 tons per acre (Roskoski 1977), but 16 other measurements from 6 scientific articles average 17 tons per acre, with a low of 8 tons per acre.[18, 21, 8, 14, 16, 2] Dead trees made up 3 to 14 percent of the basal area in five hemlock-yellow birch stands and 5 to 34 percent of basal area in sugar maple-beech-yellow birch stands.[23] Other research suggests retention of between 5 and 17 snags per acre.[7, 15, 13] Tubbs and colleagues recommend leaving between one and ten live decaying trees per acre at least 18 inches DBH.[24] Research has documented a range of 7 to 25 to cavity trees per acre in unmanaged stands.[7, 13]

Transitional Hardwood /Oak-Hickory Forests

Measures of the DWM in transitional hardwood forests, i.e., oak-hickory forests of southern New England, range from 5.8 to 18 tons per acre.22, 12 Out of seven oak stands in Connecticut, the number of dead trees ranged from 19 to 44 per ac or 5 to15 percent of basal area.[23]

White and Red Pine Forests

Estimates of the volume of downed dead wood in white and red pine forests range from 1.6 to 50 tons per acre of DWM.3, 10 Unmanaged red pine stands in the Great Lakes area had 30 snags per acre while a managed forest had 6.9 per acre.[3] Many of the red oak and white pine stands on sandy outwash sites are susceptible to nutrient losses because of a combination of low-nutrient capital and past nutrient depletion.[9]

4. CARBON CONSIDERATIONS AND GUIDELINES

To date, forestry or biomass harvesting BMPs have not included guidelines for the management of carbon. However, climate change has the potential to fundamentally change both forests and forestry over the next century. Moreover, climate change has added carbon management to the responsibilities of forest managers and landowners (Forest Guild Carbon Policy Statement 2010). Protecting forests from conversion to other land uses is the most important forest management measure to store carbon and mitigate climate change. Biomass harvests may reduce the incentive to convert forests to other uses by providing additional income to forest landowners, and maintaining the forest industry and availability of markets.

The extent to which forest biomass can serve as a low-carbon alternative to fossil fuels is currently the subject of intense debate. In 2010, the Forest Guild is engaged in a comprehensive study commissioned by the Massachusetts Department of Energy Resources and led by Manomet Center for Conservation Sciences. Together with Manomet and other partners, we are investigating the impact of various forest practices on atmospheric carbon between managed and unmanaged forests. The results of this study will be available by June 2010 and will be used to expand this section on the carbon considerations for biomass harvesting. The Manomet study will model different biomass harvest scenarios to help determine which forest practices have less of an impact on the accumulation of atmospheric carbon.

In the interim, the following sections offer suggestions based on research that is currently available. It is important to recognize that in some cases a practice that contributes to a significant reduction in atmospheric carbon may be, or may appear to be, in conflict with considerations regarding biodiversity or long-term site productivity, as outlined in previous sections of this document. For example, while utilizing logging slash for energy may prove important in a scenario designed to reduce atmospheric carbon, the retention of some logging slash post harvest may also be important for the maintenance of forest productivity. In such cases, as in many areas of forestry, divergent goals must be balanced for the specific operating unit or ownership. As discussed in previous sections, the guidelines in this report are primarily intended to support decision making about the maintenance of ecological function and value in a forest management context.

Strategies that Improve the Carbon Budget on Managed Forests

Some forest management strategies can increase carbon sequestration rates and store more carbon over time than others. Silviculture that encourages the development of structural complexity stores more carbon than silvicultural methods that create homogenous conditions. Uneven-aged management is often used to promote a structurally complex forest and can sequester more carbon than less structurally complex forests managed with even-age methods. Even-aged management systems periodically remove most of the forest carbon. When used in existing mature forests they may have a greater negative carbon impact, particularly since near-term carbon emission reductions are most important. Where even-aged management systems are appropriate, encouraging advance regeneration, or retaining residual components of the original stand, may be the fastest way to build up or maintain forest carbon. Extending rotation length will also result in an increased mean carbon stocking volume and a potential increase in carbon in harvested wood products stored offsite.

The use of logging slash for energy production has a lower carbon impact than the use of live trees for energy because logging slash will decay and emit carbon and other greenhouse gases, while live trees will continue to sequester carbon. Similarly, since trees naturally die, decay, and emit carbon, harvests that focus on suppressed trees likely to die in the near future produce fewer carbon emissions overall than the harvest of trees that are healthier, sequester carbon faster, and have long life expectancies. By using biomass harvests to remove suppressed trees with shorter life expectancies, the remaining healthier trees, "crop trees," can grow faster and larger and produce higher-value products. These more valuable products have the potential to store carbon off-site longer than products with a shorter life cycle, such as paper or shipping pallets. These products also will meet human needs while emitting less carbon than alternatives such as steel or concrete. However, the harvest of future crop trees for energy is the worst case scenario: such a harvest reduces on-site carbon, probably limits the economic productivity of the stand, and reduces the opportunity to produce higher-value products that provide long-term carbon storage and displace more carbon-intensive products.

Determining the Carbon Impact of Biomass Harvesting

While the use of forest biomass for energy production can be helpful in mitigating climate change, accounting procedures for carbon mitigation programs must accurately account for all of the impacts of the proposed biomass use. The accounting should be based on a life cycle analysis that evaluates the effects of forest management and biomass removals on forest carbon . In order to determine the carbon impact of a biomass harvest, the analysis must include the following elements:

1. The amount of carbon removed from the site.

2. The amount of carbon used to grow, remove and transport the material to utilization.

3. The efficiency and carbon emissions of the use of forest biomass for energy, compared to business-as-usual (i.e., no biomass harvest) alternatives.

4. Future carbon sequestration rate for the site.

5. The impact of biomass removals on the site's capacity to grow forest products that store carbon or replace other carbon-intensive products.

6. The time required to re-sequester the carbon removed from the site and the time required to re-sequester the carbon that would have been sequestered in the business-as-usual scenario.

7. The business-as-usual scenario which includes

   a. Predicted harvest rates for the forest type and site in question

   b. Carbon emissions factors for the production, transportation, and use of the business-as-usual fuel, most likely a fossil fuel.

A full accounting that includes these elements can help answer complex questions regarding forest management and carbon impacts. For example, logging slash plays a number of functions. It is a valuable source of nutrients, provides biodiversity habitat, stores carbon on-site and is a potential source of renewable energy. Biomass retention guidelines provide targets for how much to retain for ecological reasons. But how much to remove as a renewable fuel versus how much to leave for on-site carbon storage can only be answered by comprehensive modeling of carbon flows over time.

Guidelines for Carbon Storage

- When managing for shade-tolerant and mid-tolerant species, a shift from even-aged to uneven-aged management will increase the retention of carbon on-site.
- When appropriate to the tree species, a shift to regeneration methods that encourage advanced regeneration, such as from clearcut to shelterwood, will retain carbon on-site for longer periods.
- Retain reserve trees or standards or delay their removal.
- Delay regeneration harvests or lengthen harvest cycles to grow trees for longer times and to larger sizes.
- Encourage rapid regeneration.
- Capture natural mortality as efficiently as possible while retaining adequate numbers of snags, decaying trees, and DWM.
- Use biomass harvests to concentrate growth on healthy crop trees that can be used to manufacture products that hold carbon for long periods or replace carbon-intensive products.

5. RESOURCES AND REFERENCES

BMPs and Other State Guides

- Maine's Woody Biomass Retention Guidelines http://www.maine.gov/doc/mfs/pubs/biomass_retention_guidelines.html
- Biodiversity in the Forests of Maine: Guidelines for Land Management http://www.maine.gov/doc/mfs/pubs/pdf/biodiversity_forests_me.pdf
- Vernal Pool Habitat Management Guidelines (Maine) http://www.maine.gov/doc/mfs/pubs/pdf/vernal_pool_hmg.pdf
- Good Forestry in the Granite State: Recommended Voluntary Forest Management Practices for New Hampshire http://extension.unh.edu/resources/files/Resource000294_Rep316.pdf
- Acceptable Management Practices for Maintaining Water Quality on Logging Jobs in Vermont http://www.vtfpr.org/watershed/documents/Amp2006.pdf
- Massachusetts Forestry Best Management Practices Manual http://www.mass.gov/dep/water/drinking/forstbmp.pdf
- Connecticut Best Management Practices for Water Quality while Harvesting Forest Products http://www.ct.gov/dep/cwp/view.asp?A=2697&Q=379248
- Northeast Master Logger Certification Program http://www.masterloggercertification.com/
- Natural Fuels Photo Series http://depts.washington.edu/nwfire/dps/

Forest Guild Reports

- Ecology of Deadwood in the Northeast
- www.forestguild.org/publications/research/2010/ecology_of_deadwood.pdf
- An Assessment of Biomass Harvesting Guidelines www.forestguild.org/publications/research/2009/biomass_guidelines.pdf
- Synthesis of Knowledge from Biomass Removal Case Studies www.forestguild.org/publications/research/2008/Biomass_Case_Studies_Report.pdf
- A Market-Based Approach to Community Wood Energy: An Opportunity for Consulting Foresters www.forestguild.org/publications/research/2008/Market_Based_CWEP_Approach.pdf

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APPENDIX 5

SUMMARY OF PUBLIC INPUT TO STUDY

The intent of the public meeting held on December 17, 2009 in Holyoke, Massachusetts was:

1) to share information about the study and the questions it will address; and

2) solicit public input about additional questions the research team should consider (within the scope of the DOER RFP).

Nearly 200 people attended the public meeting. Following an overview presentation, those that were interested in providing input were broken into to small groups where the questions and comments were recorded and reported out. Those questions and comments are contained in the table below. The team reviewed these inputs and addressed those that were relevant to the study and within the scope of what DOER asked the team to assess. Additional input was solicited via the internet. The internet site was meant to be a venue for the submission of additional comments and not a forum for discussion with the study team. Maintaining an ongoing public dialogue during the study was outside the scope and budget of the study commissioned by DOER.

Outside of the public meeting, many additional submissions of comments, opinion, technical resources, and relevant articles were also submitted to the team and distributed to the appropriate subject matter expert. Submissions were made by a range of concerned citizens, organizations, and technical experts.

Comments/Questions developed during small group breakout sessions at December 17, 2009 input meeting in Holyoke, MA (note: several submissions were illegible)	
Comment	**Category**
Why weren't researchers working on this issue in west included on panel?	Comments/Questions to DOER
Will each of these questions be explicitly dealt with in a public way?	Comments/Questions to DOER
Why aren't they looking at emissions/pollution?	Comments/Questions to DOER
How is study being coordinated with adjacent states?	Comments/Questions to DOER
If we gave this level of scrutiny to every other power producer, would anything get built?	Comments/Questions to DOER
Are new technologies (such as combined heat and power) being encouraged for existing power plants?	Comments/Questions to DOER
Can (we) guarantee exactly what emissions are emitted?	Comments/Questions to DOER
Sustainable communities - where is power going? (local or distant)	Comments/Questions to DOER
What happens when the wood runs out, will you turn to waste? Trash? And are there adequate standards in place to govern trash?	Comments/Questions to DOER
What if your assumptions and study results are wrong and the biomass plants are built?	Comments/Questions to DOER
What if your assumptions are based on sustainable harvesting and there is no enforcement after the plants are built, and illegal clearcuts are rampant?	Comments/Questions to DOER
Why isn't this being run as a MEPA Study?	Comments/Questions to DOER
Will you also consider water resources needed for biomass electric?	Comments/Questions to DOER
Are they delaying biomass plants until these studies are done? If not, what is the purpose of these studies? Can't this be studied in lab or research? What if state is [?] without proper data?	Comments/Questions to DOER
What is states statutory authority to ban issuance of new qualifications for REC and effect on ongoing biomass projects? Need explanation of RPS in MA and neighbors. Address electricity market fundamentals as it drives biomass.	Comments/Questions to DOER
Adequacy of DCR to oversee forest cutting on private lands and state & capacity to expand question to other states.	Comments/Questions to DOER
What can be done to prevent invasive species transfer with increasing wood transport of other tree parts?	Comments/Questions to DOER
Why won't the state halt existing permitting process for biomass while study in progress instead of issuing permits in environment of uncertainty?	Comments/Questions to DOER

How can the state prevent clustering of incinerators?	Comments/Questions to DOER
When are sociological impacts of biomass to be studied?	Comments/Questions to DOER
Why are there four proposals at this time for biomass plants?	Comments/Questions to DOER
What are the impacts of biomass plants on river ecology and water resources?	Comments/Questions to DOER
How can you be permitting the plants before the sustainability has been determined?	Comments/Questions to DOER
Is there a regional solution to biomass plants?	Comments/Questions to DOER
This is all second growth forest, why cut and destroy the best carbon sequesters we have (which don't charge)?	Comments/Questions to DOER
The wind blows for free, how much do you charge?	Comments/Questions to DOER
If 1/3 of biomass in MA is proposed to use construction and demolition debris, then why are we only studying woody forest biomass?	Comments/Questions to DOER
Will you examine the impact of increased biomass harvesting on the economics of tourism and recreation that exists in western MA?	Comments/Questions to DOER
Please consider the possibility of a statewide referendum in 2012 to stop all logging on public lands.	Comments/Questions to DOER
Why do we need biomass?	Comments/Questions to DOER
Carbon accounting of corporate energy consumption vs. future energy consumption.	Comments/Questions to DOER
What will harvesting of forests do to tourism industry?	Comments/Questions to DOER
What are the consequences of continued over-reliance on fossil fuels vs. various biomass scenarios?	Comments/Questions to DOER
With overall electric consumption projected to go down, why do we need biomass plants?	Comments/Questions to DOER
Why not put subsidies to conservation or non-emission technologies?	Comments/Questions to DOER
Will Governor be able to take wood from private lands by eminent domain?	Comments/Questions to DOER
How can we allow biomass combustion when we cannot remove particulate matter < 2.5?	Comments/Questions to DOER
Concern if RECs for sustainable forestry for biomass, then we'll lose control of forest.	Comments/Questions to DOER
Who will answer the question about human health?	Comments/Questions to DOER
90% of the energy used in MA is from fossil fuels, 4.5% from hydro. Wind and solar are minimal. If we can't use biomass, then how will we get to the 10% RPS? What's the solution for getting off fossil fuels?	Comments/Questions to DOER
When and how, if at all, will the state address it's August, 2009 decision to only include waste sources in the renewable fuel standard? What about non-food energy crops? Cellulosic ethanol? Algae and direct-to-fuel microbes and processes? Is this study going to be the main input to the state's stance on biofuel feedstocks? If so, then why is the focus only on forests and wood? What about fallow lands? Non-thermal transformation of feedstocks and other advanced technologies?	Comments/Questions to DOER
Have they considered the ballot initiative where sufficient signatures were just collected fort the 2010 ballot and the fact that if it passes, incinerators will not be eligible for renewable energy credits and how this will impact the economics of the biomass effort? Related: citizen consideration of a similar ballot in 2012 for prohibition of all logging on public lands?	Comments/Questions to DOER
Will the research address the advisability of any biomass harvesting or removal first? All other questions follow.	Comments/Questions to Team
What is the definition of clearcutting (is it prohibited, is it proceeding?)?	Comments/Questions to Team
Are you aware state not FSC cert and has not been since April 10th? And there are serious conditions open on their forestry practices?	Comments/Questions to Team

Water quality and hydrology issue?	Comments/Questions to Team
How much non-renewable energy is used to produce renewable energy?	Comments/Questions to Team
Clean wood vs. construction/demo wood	Comments/Questions to Team
Alternative transportation of wood opportunities.	Comments/Questions to Team
Nitrogen cycles/methane cycles. How are they affected by biomass harvesting?	Comments/Questions to Team
How will biomass harvesting (removal of organic matter) affect acid rain impacts on forest soil?	Comments/Questions to Team
Where will you get your information on the technological aspects of burning biomass?	Comments/Questions to Team
How will biomass harvesting contribute to the spread of invasive species?	Comments/Questions to Team
Silvicultural perspective - what markets other than biomass are there for low grade wood?	Comments/Questions to Team
Is there a realistic time frame for the scope of study? Is there a way to address the time issue?	Comments/Questions to Team
How are they defining "forest health" and "forest sustainability"?	Comments/Questions to Team
Where will the displaced animals go?	Comments/Questions to Team
Incentives to landowners?	Comments/Questions to Team
Shifting balance of renewable?	Comments/Questions to Team
Will you consider energy security of local fuel?	Comments/Questions to Team
What are the positions of the Audubon Society and other environmental groups on biomass energy?	Comments/Questions to Team
Need to consider project finance implications in order to avoid considering unfeasible options or recommendations.	Comments/Questions to Team
Will DOER-funded SFBI studies be considered/utilized?	Comments/Questions to Team
Look at long experience with biomass energy in New England (especially southern NH).	Comments/Questions to Team
Look at other uses of biomass (ethanol etc.).	Comments/Questions to Team
Are BMPs required to be followed on public land? Concern they have not been followed in the past consistently.	Comments/Questions to Team
Where are you drawing the circle for supply of biomass per plant? Is it limited to 50 mile radius for each plant? Are you looking at a limit on plants with regard to supply (e.g., when several new plants are proposed and there are existing plants)?	Comments/Questions to Team
Are they considering pyrolysis as an alternative technology?	Comments/Questions to Team
Are you comparing biomass to other renewables or only to carbon based fuels?	Comments/Questions to Team
Are they starting with an hypothesis or asking questions without an hypothesis? What method are they using - published sources - for answering questions? Are they bringing a bias that they are trying to prove as true?	Comments/Questions to Team
Whate about the impact on wood prices? Are the changes in prices being considered in the economic impact analysis? The mix of biomass sources could change in price and so could carbon.	Comments/Questions to Team
Is there representation on the team from agricultural interests? Look at impacts on farmland.	Comments/Questions to Team
What about non-forest biomass resources? Are they being considered?	Comments/Questions to Team
What about infrastructure limits? (e.g., we have XX tons/day - but no way to get it to where [facilities are]).	Comments/Questions to Team
Are the total scope of impacts being considered? Co-firing issue needs to be taken into account more fully.	Comments/Questions to Team

NY study - How will their results affect our study? Or be taken into account as we embark on this?	Comments/Questions to Team
What is the geography being studied - just within Massachusetts?	Comments/Questions to Team
Are other pollutants being considered besides carbon (e.g., black carbon)?	Comments/Questions to Team
Are you factoring in the impacts of climate change over the next 50 years when evaluating the resource?	Comments/Questions to Team
BMPs are based on historical records.	Comments/Questions to Team
Are you considering energy to dry biomass?	Comments/Questions to Team
Why wasn't the study done prior to permitting plants?	Comments/Questions to Team
Are you looking at all scale technologies (e.g., home wood stoves) or only on larger-scale institutional level?	Comments/Questions to Team
Are you considering that biomass may not be sustainable or a good idea for harvesting for energy at all?	Comments/Questions to Team
After you establish the baseline, could you then create a model that would examine the impact of a biomass plant within 50-75 miles radius of the plant and compare the environmental impact of biomass to the other fuel sources used within that region, like wind, hydro, coal, oil, etc., and not include areas with no proposed biomass plants?	Comments/Questions to Team
Will this report dive right in or preface with layperson friendly terms and fundamental terms? Providing something accessible to public including life cycle of a tree and forest as it relates to carbon sequestration.	Comments/Questions to Team
Will they share report on progress or black box final issue?	Comments/Questions to Team
Existing Pine Tree Biomass already burning biomass. Are they addressing the draw of biomass plants to pull in new wood products? Do we need additional constraints on any plant? Need to address impossibility of ensuring fuel specifications.	Comments/Questions to Team
Will baseline study - look at each energy source, compare sustainability, renewability and carbon consequences including conservation, solar, efficiency, wind.	Comments/Questions to Team
See how more advanced country (Japan, Scandinavia, etc.) have dealt with biomass reducing fossil fuel.	Comments/Questions to Team
Climate models see MA as warmer - more erratic weather. Potential of drought to kill forest if too dense. Will model consider drought effect on unmanaged forest?	Comments/Questions to Team
Can the team openly address skepticism toward state and skepticism about panel members' past activities as a delay tactic. Biomass developers have applauded this study.	Comments/Questions to Team
Address biochar benefits/feasibility.	Comments/Questions to Team
When studying levels of carbon sequestration in between managed and unmanaged forest, distinguish "poorly managed forest" from "well managed forest".	Comments/Questions to Team
Will you study different biomass harvesting systems (i.e., cut-to-length vs. whole tree) in terms of stand damage, soil nutrient levels, and democratizing access to biomass markets (i.e., allowing all loggers to participate in the market, not just those with expensive logging/chipping systems) - This would require new biomass plants to accept round wood.	Comments/Questions to Team
Assessing amount of clean wood waste generated (i.e., tree trimming; ice storm wood; sawmill remains; waste pallets; secondary manufacturing waste; roadside trimming).	Comments/Questions to Team
Full transparency of funding sources of the members of the study group.	Comments/Questions to Team
Define "biomass". Is it woody biomass?	Comments/Questions to Team
Consider pyrolysis as technology.	Comments/Questions to Team
Consider methane production from natural forest decomposition.	Comments/Questions to Team
Assess the impact of residential use of biomass vs. commercial use of biomass.	Comments/Questions to Team

Will MA DFW goals of early successional habitat creation be considered?	Comments/Questions to Team
Regulations by basal area. Is this the best way to regulate whole tree harvesting?	Comments/Questions to Team
Are you considering that management on stand land may change?	Comments/Questions to Team
What capacity of mechanized operators will be required?	Comments/Questions to Team
It is not just a question of "sustainability". Is it a good idea to burn forests when we have too much pollution, too much carbon in the atmosphere, and already stressed forests.	Comments/Questions to Team
What is the impact of biomass market on incentives for private forest landowners? Will this help keep forest land in forests?	Comments/Questions to Team
Add other indicators of forest health.	Comments/Questions to Team
What were the positions of the consultants on biomass prior to being commissioned for this study?	Comments/Questions to Team
Research Question 2 may want to factor in diesel and gasoline truck transportation of forest fuels to the biomass plants as that relates to sustainability.	Comments/Questions to Team
How many invasive species will come to visit when we truck in wood from the whole northeast? Worcester has had to euthanize a whole bunch of its trees.	Comments/Questions to Team
Will you look at the impact of increased wood harvesting for biomass on the market for firewood? A concern in Franklin County is that the wood market will drive up the price of firewood for people who rely on it to heat.	Comments/Questions to Team
How is waste biomass byproduct factored into biomass equation?	Comments/Questions to Team
More clarification on assumptions in study.	Comments/Questions to Team
Why so many men on the study team?	Comments/Questions to Team
Will efficiency of different biomass energy technologies be taken into consideration?	Comments/Questions to Team
What are environmental and economic impacts of inefficient combustion of biomass?	Comments/Questions to Team
Will building/construction of power plants be factored into LCA?	Comments/Questions to Team
Will biomass harvesting be like strip mining and how do we prevent it?	Comments/Questions to Team
Consider indirect impacts in addition to land impacts.	Comments/Questions to Team
Balance effect of development and managed forests.	Comments/Questions to Team
Is construction and demolition material included in the study?	Comments/Questions to Team
Will the policy address the need for innovation in bioenergy and recognize new technologies such as gas pyrolysis and alternative feedstockes such as wastewood, construction debris, etc.	Comments/Questions to Team
Is construction and demolition material included in the consideration for the study?	Comments/Questions to Team
How much trucking will there be and how will that affect local traffic patterns and the quality of life? What is the energy impact of the trucking and will that be considered as part of the life-cycle analysis? Why are four plants so close together all being proposed at the same time and where will the wood come from?	Comments/Questions to Team
Indirect impacts – in addition to the land impacts, what is the environmental cost of the "growth induced impacts"? (such as the growth of the local economy?	Comments/Questions to Team
How can we balance the effect of development versus managed forests. What will be the land ownership incentive? The incentive to hold land in private hands? If we become too restrictive, then people will not be able to earn income from their land and have to sell off to developers. Concern about incentives for land ownership. Also, concern if REC's for sustainable forestry for biomass are impacted, then we will lose control of our forests.	Comments/Questions to Team

Request to include long-term anthropological perspective of human forest use in the area and how social and economic situations, values, etc. affect the use of forest. Going all the way back to native American Indians; through colonial times, to industrialization to the present. (editor comment: are we so vain as to think we will leave no heritage)?	Comments/Questions to Team
What is the H2O content of the wood being considered?	Comments/Questions to Team
Are we going to include extreme scenarios in the baseline such as a complete cut-off of foreign oil (i.e. middle east nuclear scenario) and the ability of the state (and the country) to continue to function? Will an extreme case be included in the baseline?	Comments/Questions to Team
How will more smaller plants with more lax air quality regulations and controls affect health?	Public Health Concerns
Look at health issues.	Public Health Concerns
Will you be looking at the broadest range possible of forest health indicators? Should make sure to also overlay analysis with the other detailed biodiversity planning in state, including Woodlands and Wildlands and TNC Ecoregional Plans.	Public Health Concerns
Call on state to address the medical society's statement that biomass incinerators pose unacceptable health risks.	Public Health Concerns
Why propose biomass within city limits or in a valley with a high percentage of respiratory illness? Are you mad?	Public Health Concerns
Air quality changes from biomass.	Public Health Concerns
Fine particulate given off by large trucks and impact on air quality.	Public Health Concerns
Other emissions from biomass combustion (other health impacts).	Public Health Concerns
What will happen to remnants from burning – the ash? Will there be environmental problems from it?	Public Health Concerns
Who will answer the question about human health and local environments? These plants are in low-lying valleys with poor air circulation and bad air quality already. What about the local climate and weather and current health issues (such as already high cancer rates)?	Public Health Concerns

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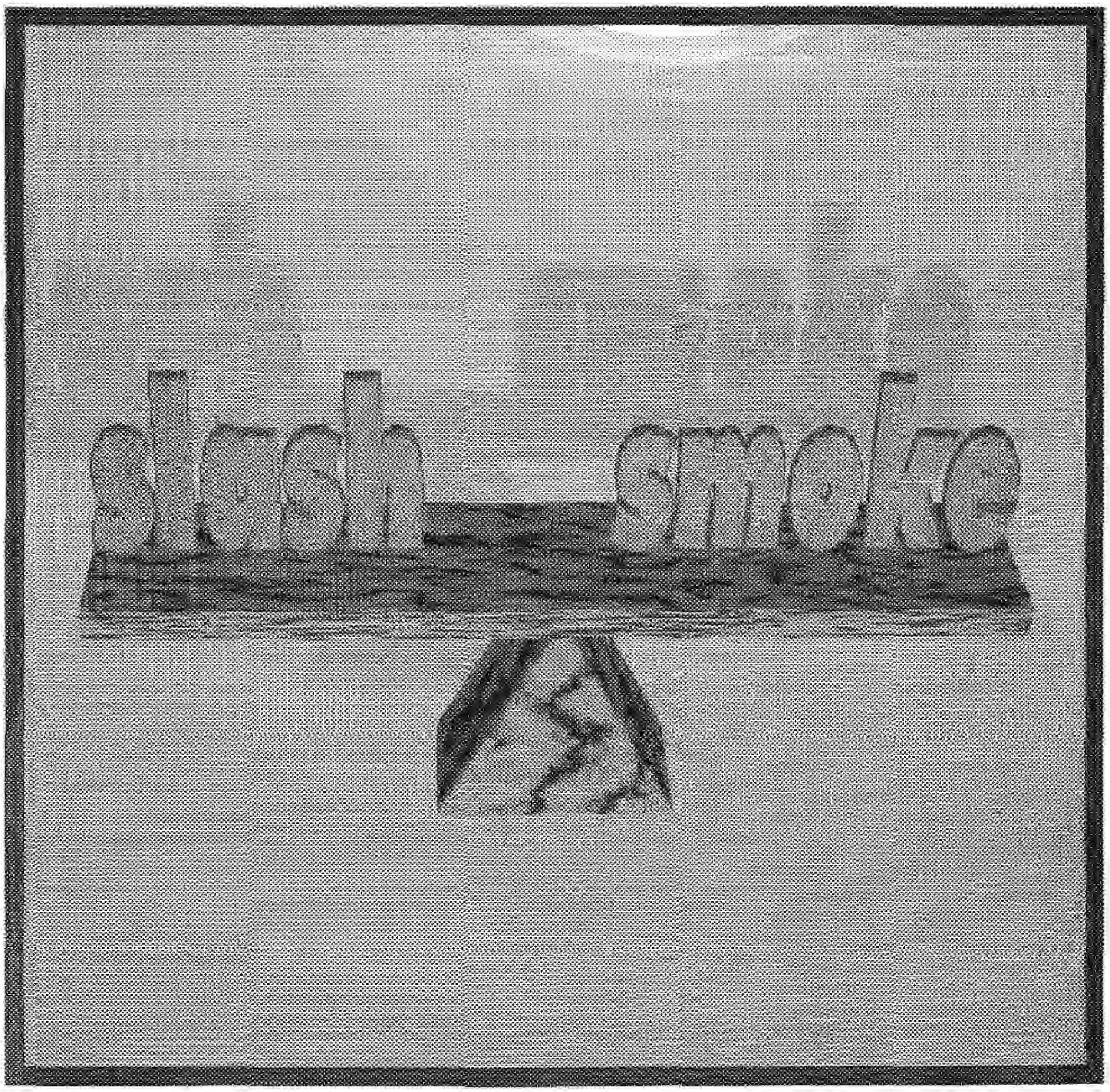
Forest Service
Pacific Northwest
Research Station
General Technical
Report
PNW-GTR-364
February 1996

Guidelines for Estimating Volume, Biomass, and Smoke Production for Piled Slash

Colin C. Hardy



Abstract

Hardy, Colin C. 1996. Guidelines for estimating volume, biomass, and smoke production for piled slash. Gen. Tech. Rep. PNW-GTR-364. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 17 p.

Guidelines in the form of a six-step approach are provided for estimating volumes, oven-dry mass, consumption, and particulate matter emissions for piled logging debris. Seven stylized pile shapes and their associated geometric volume formulae are used to estimate gross pile volumes. The gross volumes are then reduced to net wood volume by applying an appropriate wood-to-pile volume packing ratio. Next, the oven-dry mass of the pile is determined by using the wood density, or a weighted-average of two wood densities, for any of 14 tree species commonly piled and burned in the Western United States. Finally, the percentage of biomass consumed is multiplied by an appropriate emission factor to determine the mass of PM, PM10, and PM2.5 produced from the burned pile. These estimates can be extended to represent multiple piles, or multiple groups of similar piles, to estimate the particulate emissions from an entire burn project.

Keywords: Fuel, emissions, piled slash, smoke management.

Contents

Introduction

The guidelines in this document address the critical need to quantify both biomass consumption and the air quality impacts from the burning of piled woody debris. Piling and burning of woody debris from activities such as timber harvesting, road building, and residential or commercial development has been a common practice for as long as these activities have occurred. It is an especially common forestry practice, where logging residue (slash) is piled on the site by bulldozers, and on the terminus (landing) of yarding and skidding trails by bulldozers, cable-yarding equipment, and log loaders. Numerous objectives are met by piling and burning: reduction of on-site woody fuel loading and the resultant reduction in harvest-created fire hazard; scarification of the surface layer and exposure of mineral soil to enhance regeneration of trees; removal of woody and organic material from roadbeds and structure sites to improve the integrity of the construction substrate; and sanitation disposal of stumps and roots infected by disease or pathogens. In some cases, logging slash is piled in anticipation of subsequent use in nonlumber markets such as combustion for energy (hog fuel), pulp chips, and firewood.

On-site burning of piled slash has both negative and positive implications for smoke management. Burning of woody biomass, regardless of its condition and distribution, creates products of incomplete combustion such as carbon monoxide, methane, and particulate matter. A variety of research methods have produced much new knowledge about the quantity and characteristics of smoke emissions from vegetation fires (Ward and Radke 1993) and from piled slash (Ward and others 1989). Piling and burning of slash has positive smoke management implications as well. In contrast with broadcast burning of the same material, piled slash burns more efficiently, with notably less smoke produced per unit mass of fuel consumed (Ward and others 1989). Further, piled slash can be burned under a broad range of weather conditions. This enables the burning of piles to be scheduled for periods of optimal dispersion and also during periods when the conflicts with impacts from other sources are minimized.

Smoke management programs in several Western States now actively encourage the piling and burning of slash, where possible, instead of the more typical practice of broadcast burning. Permitting and fee structures have created incentives for piling and burning. The increased emphasis on the practice demands significant improvements in our ability to quantify preburn fuel loadings, fuel consumption, and emissions production from burning piles.

Several previous efforts have led the way toward the methods and guidelines presented here. Techniques for estimating weights of piles and stumps were developed from a land clearing project in Washington (Mohler 1977). Relations between easy-to-measure dimensions and woody fuel volumes were developed by McNab (1980) for inventorying windrowed forest residues in the Southern United States. Results from these two methods were verified by Johnson (1981) when he compared them to results from destructive sampling. Little developed a method for estimating pile volumes using four stylized shapes and respective volumetric formulae.[1] These shapes, when combined with a ratio estimator for reducing gross pile volume to net wood volume, provide an efficient, simple method for field estimation of woody fuel volumes in piles.

The initial steps in these guidelines use the methodology and generalized shapes presented by Little (see footnote 1). Three additional shapes are included, as are species-specific wood densities, a range of wood-to-pile volume ratios,[2] and several nomagrams intended to reduce the number of manual calculations required to estimate volumes, mass, and smoke emissions.

The Six-Step Process

Six steps are required to estimate particulate emissions from a pile. The product from each step is both relevant in itself and a prerequisite parameter for completion of the next step.

Determine:

1. Total gross volume of the pile.
2. Net volume of the woody biomass.
3. Density or weighted-average density of the wood.
4. Consumable (oven-dry) mass of wood.
5. Proportion of mass consumed.
6. Mass of particulate matter produced (PM, PM10, PM2.5).

[1] Little, S.N. 1980. Estimating the volume of wood in large piles of logging residue. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 7 p. Administrative Report PNW-1. On file with: Forestry Sciences Laboratory, P.O. Box 3890, Portland, OR 97208-3890.

[2] Hardy, Colin C., Vihnanek, R. Packing ratios for piled woody debris. Manuscript in preparation. USDA Forest Service, Pacific Northwest Research Station, Seattle, WA 98105.

Step One—Total Gross Volume of the Pile

A. Select a representative pile shape—A pile can be categorized into one of the seven generalized shapes shown in figure 1. The number for each shape is its "shape code." The dimensions required to compute volumes are shown in figure 1 for each shape. Choose the shape most similar to your pile(s) from the following descriptions and figures and record the appropriate dimensions.

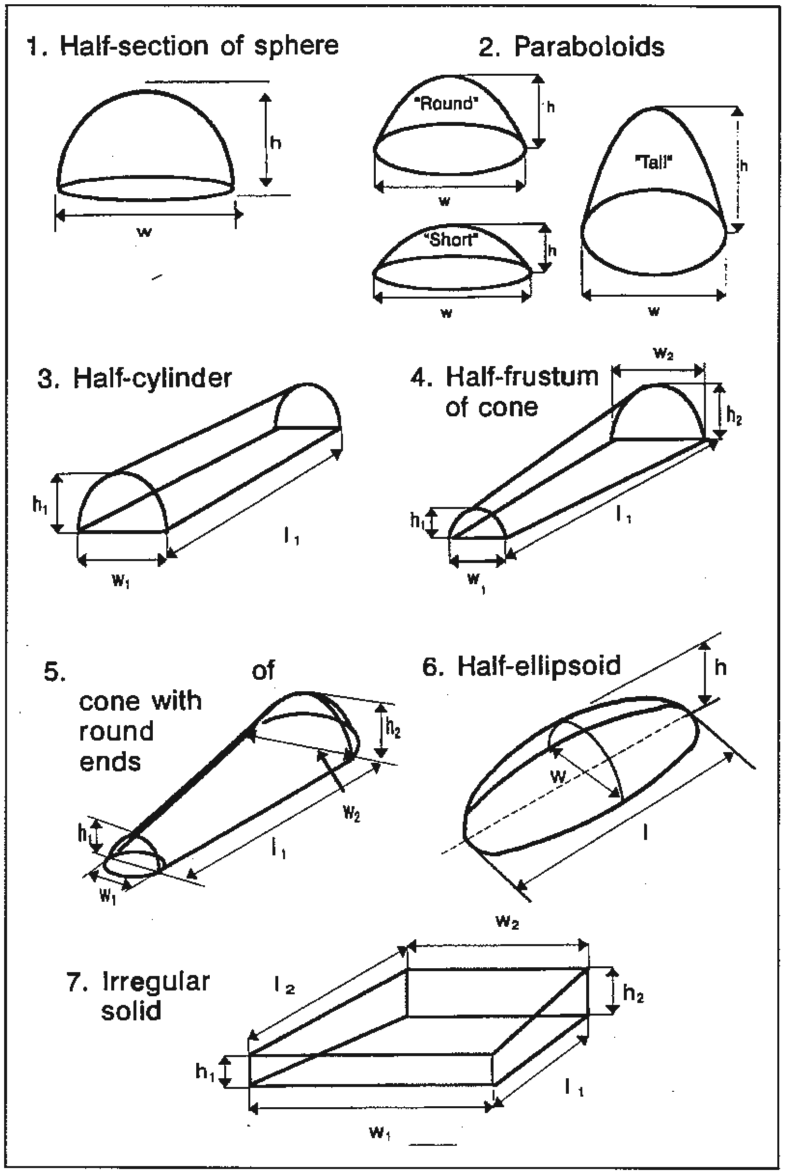


Figure 1—Five generalized pile shapes are used to represent the possible configurations of piled woody debris. Each illustration is numbered by its "shape code" (see text).

Shape code 1: Half-section of a sphere—Truly half of a ball or sphere (fig. 2). The base is round, the width is twice the height, and the sides are well- and evenly-rounded. (Observe and record either height [*h*] or width [*w*].)

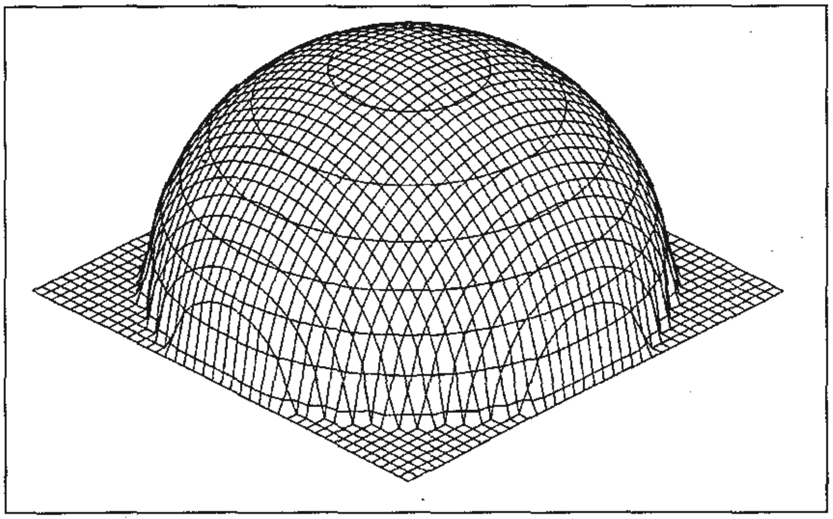


Figure 2—The half-section of a sphere is truly half a ball, with well-rounded sides.

Shape code 2: Half-paraboloid—The base is round, but the sides are parabolic, not round. Three variations of the half-paraboloid shape are shown in figure 3: half-round paraboloid, half-"tall" paraboloid, and half-"short" paraboloid. (Observe and record height [*h*] and width [*w*].)

Shape code 3: Half-cylinder—The pile is generally rounded side-to-side, with both ends of the pile approximately the same height and straight (fig. 1). Logs stacked parallel by a loader or crane can form this shape. (Observe and record width [*w*], height [*h*], and length [*l*].)

Shape code 4: Half-frustum of a cone—This shape is similar to a half-cylinder, but the cylinder tapers lengthwise, so the heights of the ends are different (fig. 1). This shape is seen when logs are stacked parallel, with the tapers oriented in the same direction. (Observe and record length [*l*] and heights or widths of the small and large ends [h_1 or w_1, and h_2 or w_2].)

Shape code 5: Half-frustum of a cone with rounded ends—Similar to shape code 4, but the ends are rounded (fig. 1). In this case, the rounded ends caused by uneven stacking and mixed piece sizes can add considerable volume to the pile. (Observe and record length of straight section of the side [*l*] and width of the small and large end [w_1, w_2].)

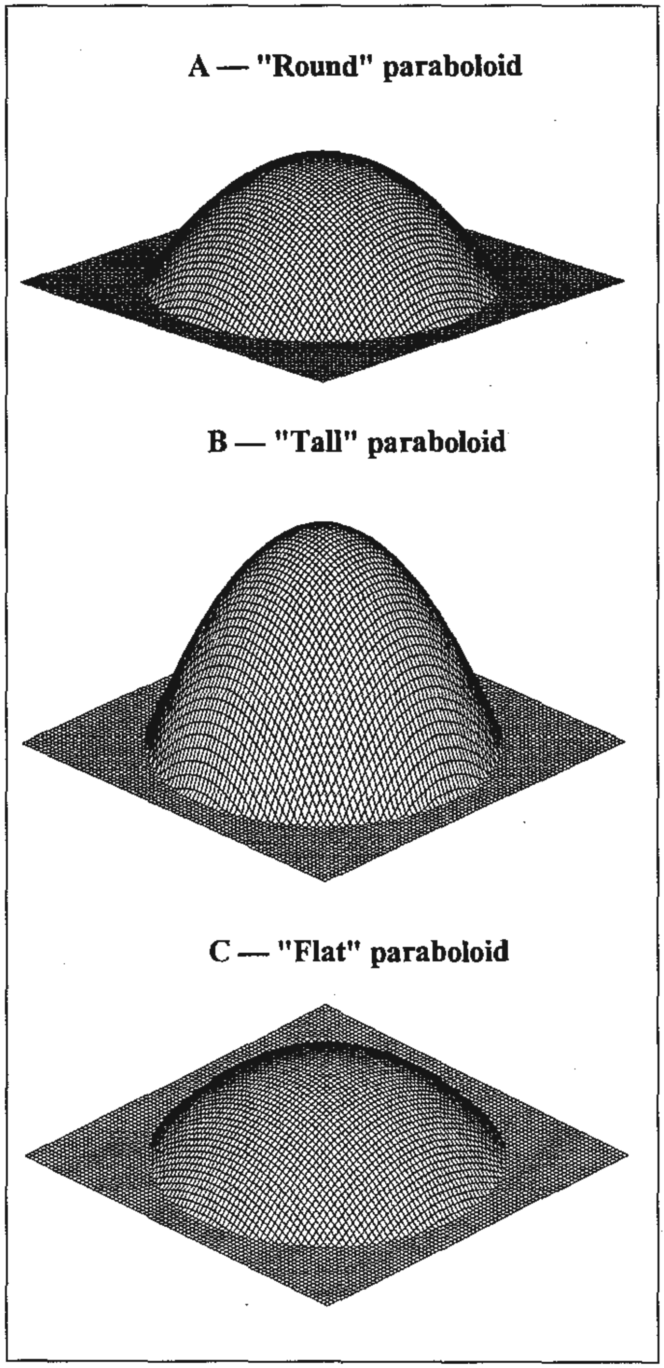


Figure 3—The base of the half-paraboloid is round but the sides are parabolic. Three variations of this shape use the same volumetric formula: (A) "round," (B) "tall," and (C) "flat."

Shape code 6: Half-ellipsoid—This pile is elongated, rounded side-to-side, with well-rounded ends (fig. 4). This shape is typical of windrowed slash. (Observe and record height [*h*], total length [*l*], and width at the widest section [*w*].)

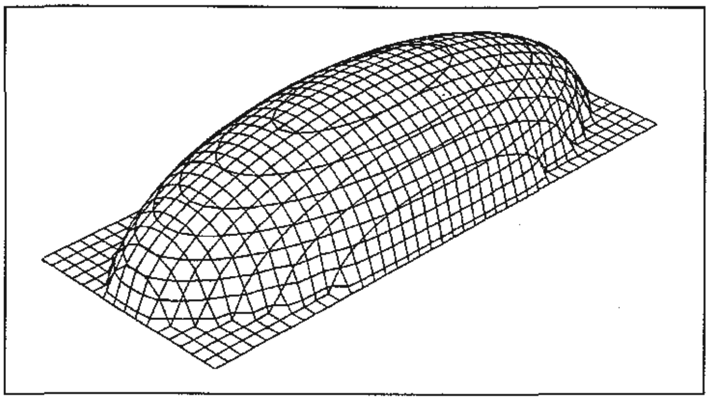


Figure 4—The half-ellipsoid shape represents a long, symmetric, tapering pile with well-rounded ends.

Shape code 7: Irregular solid—This pile is irregularly shaped with straight but uneven sides (fig. 1). The dimensions for opposing sides are not necessarily equal. (Observe and record lengths [l_1, l_2], widths [w_1, w_2], and heights [h_1, h_2].)

B. Calculate the gross volume—The gross volume for a pile represented by any of the seven shape codes can be calculated from the following volumetric formulae,

where: V = gross pile volume (cubic feet),

l, l_1, l_2 = length(s) in feet,

h, h_1, h_2 = height(s) in feet, and

w, w_1, w_1, w_2 = width(s) in feet.

Look-up tables or nomagrams are provided for some of the shapes and are referenced below with the respective formula.

Shape code 1—

$$V = \frac{2\pi h^3}{3} \text{ or } V = \frac{\pi h w^2}{6} \quad .$$

Columns 1-3 of table 1 contain look-up data for this volume. Use either pile height (column 2) or piile height and width (column 1) to determine gross volume (column 3).

Table 1—Gross pile volumes for half-section of a sphere (spheroid) and half-paraboloid pile shapes (shape codes 1 and 2, respectively)[a]

	Spheroids only		Volume by paraboloid height (in feet)								
Pile width	Height	Volume	4	6	8	10	12	14	16	18	20
– – Feet – –		*– – – – – – – – – – – – – – – – Cubic feet – – – – – – – – – – – – – – – – – –*									
4	2.0	17	25	38	50	63	75	88	101	113	126
5	2.5	33	39	59	79	98	118	137	157	177	196
6	3.0	57	57	85	113	141	170	198	226	254	283
7	3.5	90	77	115	154	192	231	269	308	346	385
8	4.0	134	101	151	201	251	302	352	402	452	503
9	4.5	191	127	191	254	318	382	445	509	573	636
10	5.0	262	157	236	314	393	471	550	628	707	785
11	5.5	348	190	285	380	475	570	665	760	855	950
12	6.0	452	226	339	452	565	679	792	905	1018	1131
13	6.5	575	265	398	531	664	796	929	1062	1195	1327
14	7.0	718	308	462	616	770	924	1078	1232	1385	1539
15	7.5	884	353	530	707	884	1060	1237	1414	1590	1767
16	8.0	1072	402	603	804	1005	1206	1407	1608	1810	2011
17	8.5	1286	454	681	908	1135	1362	1589	1816	2043	2270
18	9.0	1527	509	763	1018	1272	1527	1781	2036	2290	2545
19	9.5	1796	567	851	1134	1418	1701	1985	2268	2552	2835
20	10.0	2094	628	942	1257	1571	1885	2199	2513	2827	3142
21	10.5	2425	693	1039	1385	1732	2078	2425	2771	3117	3464
22	11.0	2788	760	1140	1521	1901	2281	2661	3041	3421	3801
23	11.5	3185	831	1246	1662	2077	2493	2908	3324	3739	4155
24	12.0	3619	905	1357	1810	2262	2714	3167	3619	4072	4524

[a] The volume for a spheroid (column 3) is determined from either the width (column 1) or height (column 2). For a half-paraboloid, find the intersection of width (column 1) and height (columns 4-12).

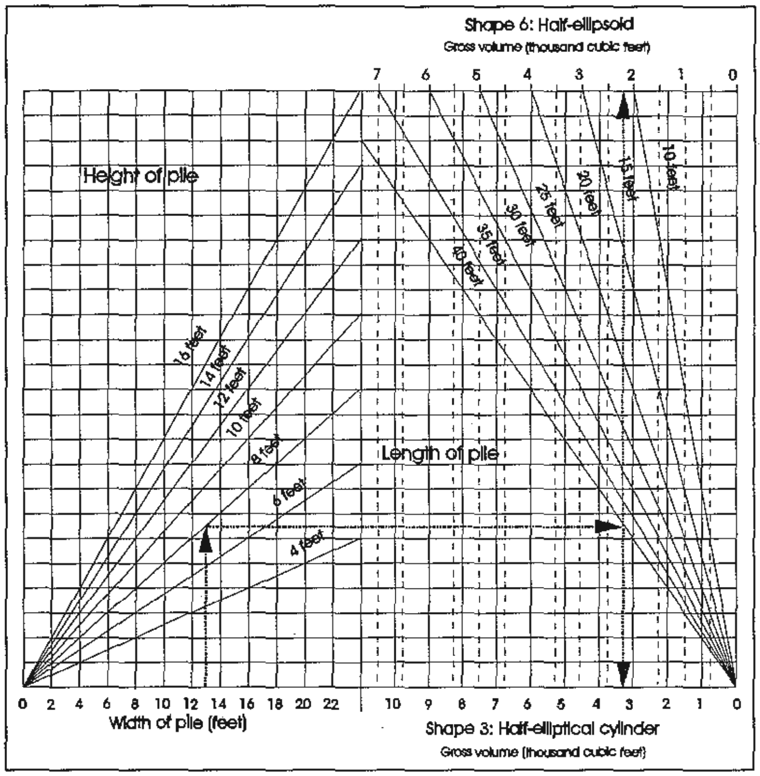


Figure 5—The nomagram assists in determining gross pile volumes (X-axes at upper and lower right) for shape codes 6 and 3, respectively, by width (X-axis at lower left), height (diagonal lines at left), and length (diagonal lines at right).

Shape code 2—

$$V = \frac{\pi h w^2}{8} \ .$$

The volume for any of the three variations of half-paraboloid is derived from the same equation. Columns 1 and 4-12 of table 1 contain look-up data for this volume, where the intersection of width (column 1) and height (columns 4-12) contains the gross pile volume.

Shape code 3—

$$V = \frac{\pi w l h}{4} \ .$$

Figure 5 is a nomagram for estimating gross volumes for shape codes 3 and 6 by using width, height, and length.[3]

Begin at the X-axis (horizontal axis) labeled "width of pile"; go up from the correct width to the diagonal line for the correct height; go horizontally to the diagonal line at the right for the correct length; go down to the right-hand X-axis (labeled "gross volume") for shape code 3, half-elliptical cylinder, to determine the gross pile volume.

[3] Full-page versions of all nomagrams are given in appendix 2.

Shape code 4—

$$V = \frac{\pi l_1 \left[h_1^{\,2} + h_2^{\,2} + (h_1 h_2) \right]}{6} \text{ if using heights,}$$

or

$$V = \frac{\pi l_1 \left[w_1^{\,2} + w_2^{\,2} + (w_1 w_2) \right]}{24} \text{ if using widths.}$$

Shape code 5—

$$V = \frac{\pi \left\{ l_1 \left[w_1^{\,2} + w_2^{\,2} + (w_1 w_2) \right] + w_1^{\,3} + w_2^{\,3} \right\}}{24}$$

Shape code 6—

$$V = \frac{\pi w l h}{6} \; .$$

Figure 5 is a nomagram for estimating gross volumes for shape codes 3 and 6 by using width, height, and length.

Begin at the X-axis (horizontal axis) labeled "width of pile"; go up from the correct width to the diagonal line for the correct height; go horizontally to the diagonal line at the right for the correct length; go up to the top right-hand X-axis (labeled "gross volume") for shape code 6, half-ellipsoid, to determine the gross pile volume.

Shape code 7—

$$V = \frac{(l_1 + l_2)(w_1 + w_2)(h_1 + h_2)}{8} \; .$$

Some piles contain a significant amount of soil, whether entrained among the wood pieces or mounded beneath the pile. The net wood volume must be reduced by an estimate of the percent of the volume occupied by soil.

Step Two—Net Volume of the Woody Biomass

Much of the gross volume of a pile is occupied by air. The ratio of wood volume to total pile volume is called the "packing ratio." The gross pile volume must be multiplied by an appropriate packing ratio to determine the net volume of woody material in a pile. Research on the packing ratio of piled slash has determined that the net wood volume can range from as low as 6 percent to as high as 26 percent (see footnote 2). These values represent extremes from 17 piles studied. The variation in packing ratio is due to numerous factors, including piling specifications, operator and machine performance, species content, and size class distribution. Only expert judgment can be used to ultimately determine a packing ratio for a particular pile or group of similar piles. For the purpose of these guidelines, data from research can be used to suggest the following packing ratios:

Tractor
Hand
Grapple
Landing

- Piles with species content dominated by ponderosa pine, with mean diameters of the large woody fuel of less than 10 inches were found to have a mean packing ratio of 10 percent (0.10).[4]
- Piles dominated by short-needled conifers had packing ratios ranging from 15 percent (0.15) to 20 percent (0.20).
- Highly compacted, clean piles with larger logs (diameters greater than 10 inches), especially those built with a crane or loader, can have packing ratios as high as 25 percent (0.25).

Multiply the gross pile volume determined in step one by an appropriate packing ratio to calculate net wood volume. The nomagram shown on the left side of figure 6 can be used to make this calculation. Begin at the X-axis labeled "gross pile volume"; go up to a diagonal line representing an appropriate packing ratio for the pile(s); go left to the Y-axis to determine the respective net wood volume. This step can be combined with step four if the nomagram is used.

Step Three—Density or Weighted-Average Density of the Wood

The oven-dry density of wood is used to calculate mass of wood for fuel loading, fuel consumption, and smoke production. Table 2 contains oven-dry densities for 14 tree species commonly piled and burned in the Western United States. Use these values if the wood in a pile is predominately one species. If two species are identified, refer to the nomagram in figure 7 to derive a weighted-average density for the pile.

First, find a line in figure 7 connecting the two species. Move from the end of the line representing the dominant species towards the other species until the line interesects the correct percentage content (vertical lines labeled on the X-axis) for the dominant species; then move horizontally either left or right to the Y-axis to determine a weighted-average density for the two species.

[4] Scientific names for tree species are included in table 2.

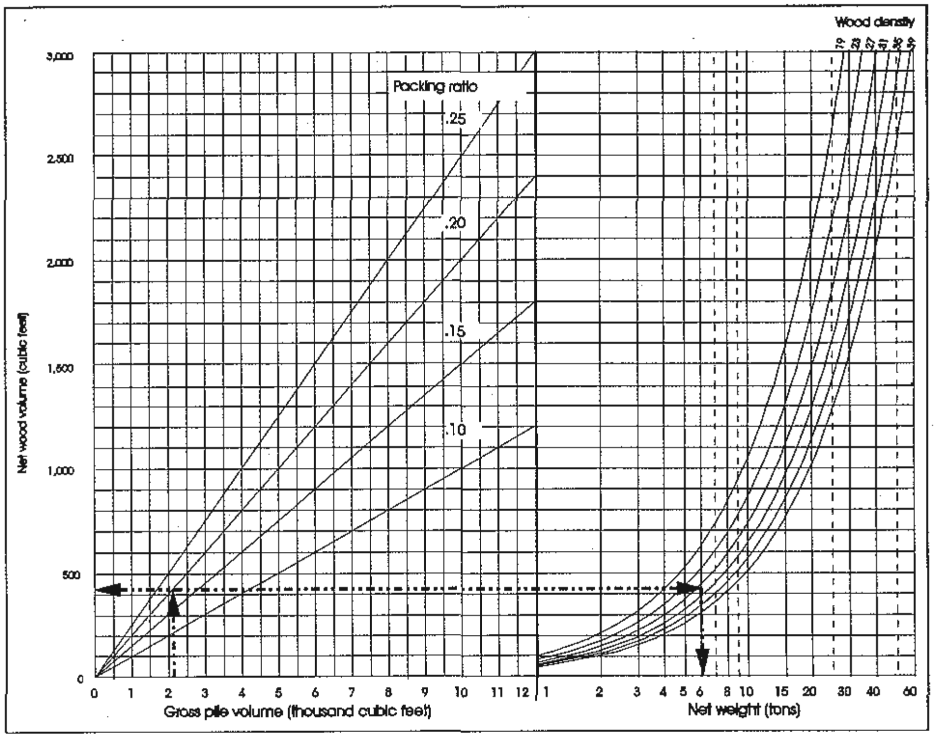


Figure 6—The nomagram assists in determining net mass of the wood (X-axis at right) by using gross pile volume (X-axis at left), packing ratio (diagonal lines at left), net wood volume (Y-axis), and wood density (curves at right).

Table 2—Green specific gravity and oven-dry density for 14 tree species commonly piled and burned in the Western United States

Species	Specific gravity (green)	Density (oven dry)
	Dimensionless	*Lb/ft^3*
Rotten wood (not species-specific)	0.30	18.7
Western redcedar (*Thuja plicata* Donn ex D.Don)	.31	19.4
Black cottonwood (*Populus trichocarpa* Torr. & Gray)	.31	19.4
Quaking aspen (*P. tremuloides* Michx.)	.35	21.9
True fir (noble) (*Abies procera* Rehd.)	.37	23.1
Red alder (*Alnus rubra* Bong.)	.37	23.1
Sitka spruce (*Picea sitchensis* (Bong.) Carr.)	.37	23.1
Ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.)	.38	23.7
Lodgepole pine (*P. contorta* Dougl. ex Loud.)	.38	23.7
Western hemlock (*Tsuga heterophylla* (Raf.) Sarg.)	.42	26.2
Bigleaf maple (*Acer macrophyllum* Pursh)	.44	27.5
Vine maple (*Acer circinatum* Pursh)	.44	27.5
Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco)	.45	28.1
Western larch (*Larix occidentalis* Nutt.)	.48	30.0
Tanoak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehd.)	.58	36.2

Sources: Panshin and others 1964, USDA Forest Service 1974.

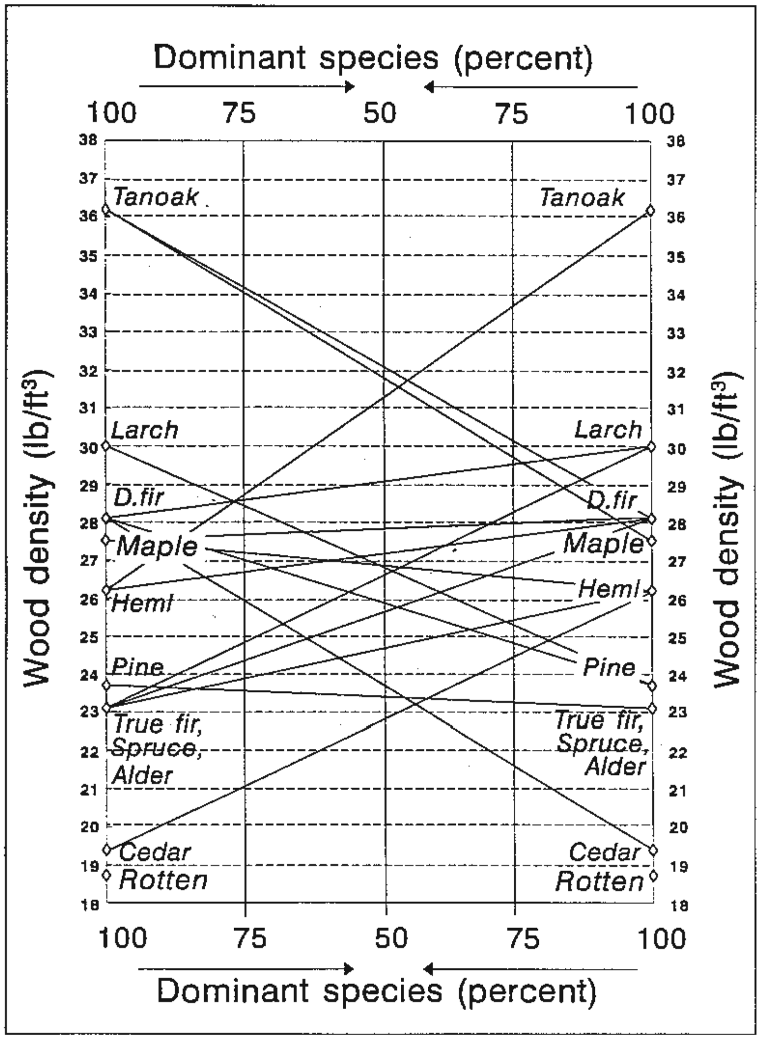


Figure 7—The nomagram can be used to calculate a weighted-average wood density (Y-axes) for two species by finding a point along a diagonal line representing two species that intersects with a vertical line indicating the correct proportion of the two species.

Step Four—Consumable (Oven-Dry) Mass of Wood

Multiply the wood density or weighted-average wood density for the pile by the net wood volume to calculate the oven-dry (consumable) mass of the pile. Divide the result by 2,000 to convert to tons. The nomagram shown on the right side of figure 6 can be used for this step.

Begin with the correct net wood volume shown on the Y-axis at the left side of figure 6; move right, horizontally, to the appropriate wood density curve on the right side of the nomagram; proceed downward (vertically) to determine the net weight of wood in the pile. Note that the X-axis at the right (net weight) is logarithmic, so interpolations must be made only between adjacent numbers on the X-axis.

Step Five—Proportion of Mass Consumed

The percentage of wood mass consumed when piles are burned typically ranges between 75 and 95 percent. Smoke management-reporting programs in several Western States recommend either 85 percent (0.85) or 90 percent (0.90). Experience and expert knowledge must be used to determine the most appropriate value for percentage of consumption. Multiply the percentage by the consumable mass of wood from step four to calculate the total mass of material consumed.

Step Six—Mass of Particulate Matter Produced

The mass of an emission produced by a fire is calculated by multiplying the mass of fuel consumed by an appropriate emission factor for the emission of interest. These guidelines provide emission factors for three size classes of particulate matter: PM (total particulate matter), PM10 (particulate matter smaller than 10 micrometers mean-mass diameter), and PM2.5 (particles smaller than 2.5 micrometers mean-mass diameter). The emission factors for these particle sizes differ with the combustion efficiency of the fire. Cleaner piles burn more efficiently than dirty piles. Consequently, cleaner piles produce less of the products of incomplete combustion, of which particulate matter is a major emission species. Figure 8 provides emission factor curves for PM, PM10, and PM2.5; the relations between the emission factors and combustion efficiency illustrate the impacts of different amounts of soil mixed into a pile. Expert judgment as well as agency policy must be considered when using the curves in figure 8.

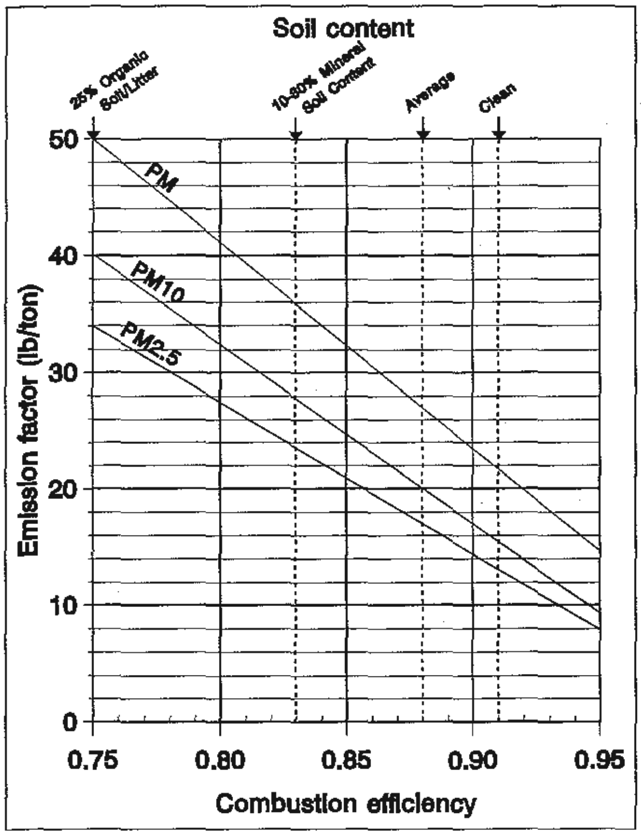


Figure 8—An appropriate emission factor for PM, PM2.5, and PM10 can be determined from knowledge of the relative amount of soil in the pile.

Start in figure 8 from an appropriate combustion efficiency determined from the relative cleanliness of the pile(s); combustion efficiency and soil content are found on the lower and upper X-axes, respectively. Follow the vertical line up from combustion efficiency, or down from soil content, to the intersection of the line for PM, PM10, or PM2.5; from that intersection move horizontally left to determine the emission factor.

Multiply the emission factor by the oven-dry mass of material consumed (from step five) to calculate the total mass of the particulate matter emission produced by the pile(s).

Recommendations and Guidance

The largest errors expected from using these guidelines will occur during the process of determining the gross pile volume(s). The seven stylized pile shapes do not provide an exhaustive choice of geometric shapes for piled slash. These seven are presented because they reflect general shapes observed by the author and other experts, and also because their volumes can be calculated relatively easily from either the formulae or the nomagrams. When the dimensions for a pile are observed, care must be taken to account for irregularities in the pile's surfaces. Try to mentally "smooth" the lobes, ridges, and valleys into an average, smooth surface. Long logs and poles extending from the pile's nominal surface can be accounted for by increasing the dimension(s) of the pile appropriately. If a significant amount of soil is either entrained within the pile or mounded beneath it, the volume of the soil must be estimated and subtracted from the gross pile volume.

The packing ratios presented in these guidelines represent empirical field data from destructive sampling of 17 piles. Even though guidelines are provided to determine an appropriate packing ratio for specific piles, an agency or administrative unit may choose to specify packing ratios for applications under their jurisdiction.

A continuous range of emission factors for PM, PM10, and PM2.5 are presented in these guidelines. The values given for "average" piles are weighted means from eight in situ field tests of emissions from burning of piles of woody debris. Results from many other related tests were used to develop the regression lines (fig. 8), which predict emission factors by using combustion efficiency. The values for PM10 were not derived from actual field observations—only PM2.5 and PM were measured in the field tests from which these data were prepared. PM10 emission factors were estimated by using limited knowledge of the size distribution of particles.

These guidelines provide procedures for estimating volume, biomass, and particulate matter emissions from a single pile. Most applications of these procedures typically will be made for multiple-pile projects. Some or all steps in these guidelines can be extended to represent a group of similar piles. For example, average dimensions can be used for all piles of a similar shape. If possible, it is helpful to map the location, shape, and dimension of each pile on a unit or project. At a minimum, piles of similar shape or size should be tallied; the formulae and nomagrams can then be applied at another time. Each agency or administrative unit may prescribe a specific method for obtaining and aggregating the data for a project.

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Appendix 1: A Hypothetical Example

- **Shape–**The pile was built with a bulldozer and can be considered a windrow. It is elongated, with an elliptically shaped base, and is rounded side-to-side with well-rounded ends.
- **Dimensions–**Length is 40 feet; width is 13 feet; height is 8 feet.
- **Wood species–**Wood content (by volume) is 75 percent Douglas-fir and 25 percent western hemlock.
- **Packing ratio–**Pile is relatively compact; about 20 percent wood-to-volume ratio (0.20).
- **Fuel consumption–**90 percent of the wood mass will be consumed.
- **Emission factors–**The pile is "average" in soil content and therefore will burn with a combustion efficiency of 0.88.

Step one—Total gross volume of the pile—

A. Select a representative pile shape: Pile shape is half-ellipsoid—shape code 6.

B. Calculate the gross volume:

Formula method: $\text{Volume} = \dfrac{\pi * 13 * 40 * 8}{6} = 2178 \text{ cubic feet}.$

Nomagram: Refer to figure 5 and follow the arrowed line to the X-axis at the upper right, where the gross volume equals about 2.2 thousand cubic feet.

Step two—Net volume of the woody biomass—

Formula method: Net volume = Gross volume x packing ratio; therefore,

2178*0.20 = 435.6 cubic feet.

Nomagram: Refer to figure 6 and follow the arrowed line to the Y-axis at the left, where the net volume equals about 435 cubic feet.

Step three—Density or weighted-average density of the wood

Formula method: The pile is 75 percent Douglas-fir and 25 percent western hemlock. Refer to table 2 for the densities of Douglas-fir (28.1 lb/ft^3) and western hemlock (26.2 lb/ft^3). Calculate the weighted average:

(0.75*28.1)+(0.25*26.2)=27.63 pounds per cubic foot.

Nomagram: Refer to figure 7, where the diagonal line connecting Douglas-fir and western hemlock intersects the vertical line representing 75 percent Douglas-fir at about 27.7 lb/ft^3 (on the Y-axis).

Step four—Consumable (oven-dry) mass of wood—

Formula method: Net wood mass = net wood volume x wood density

435.6*27.63 = 12,036 pounds or ≅ 6 tons.

Nomagram: Refer to figure 6 and follow the arrowed line to the curves on the right, then down to the X-axis, where the net mass of wood is approximately 6.0 tons.

Step five—Proportion of mass consumed—

Formula method: Mass consumed = net mass x percent consumed

6.0*0.90 = 5.4 tons.

Step six—Mass of particulate matter produced—

Formula: Total emission = mass consumed x emission factor.

Emission factors: Referring to figure 8, for an "average" pile:
PM = 27 lb/ton
PM10 = 20 lb/ton
PM2.5 = 17 lb/ton

Calculate: PM: 5.4 tons*27 lb/ton=145.8 pounds
PM10: 5.4 tons*20 lb/ton=108.0 pounds
PM2.5: 5.4 tons*17 lb/ton=91.8 pounds

Appendix 2: Full-Sized Copies of the Nomagrams

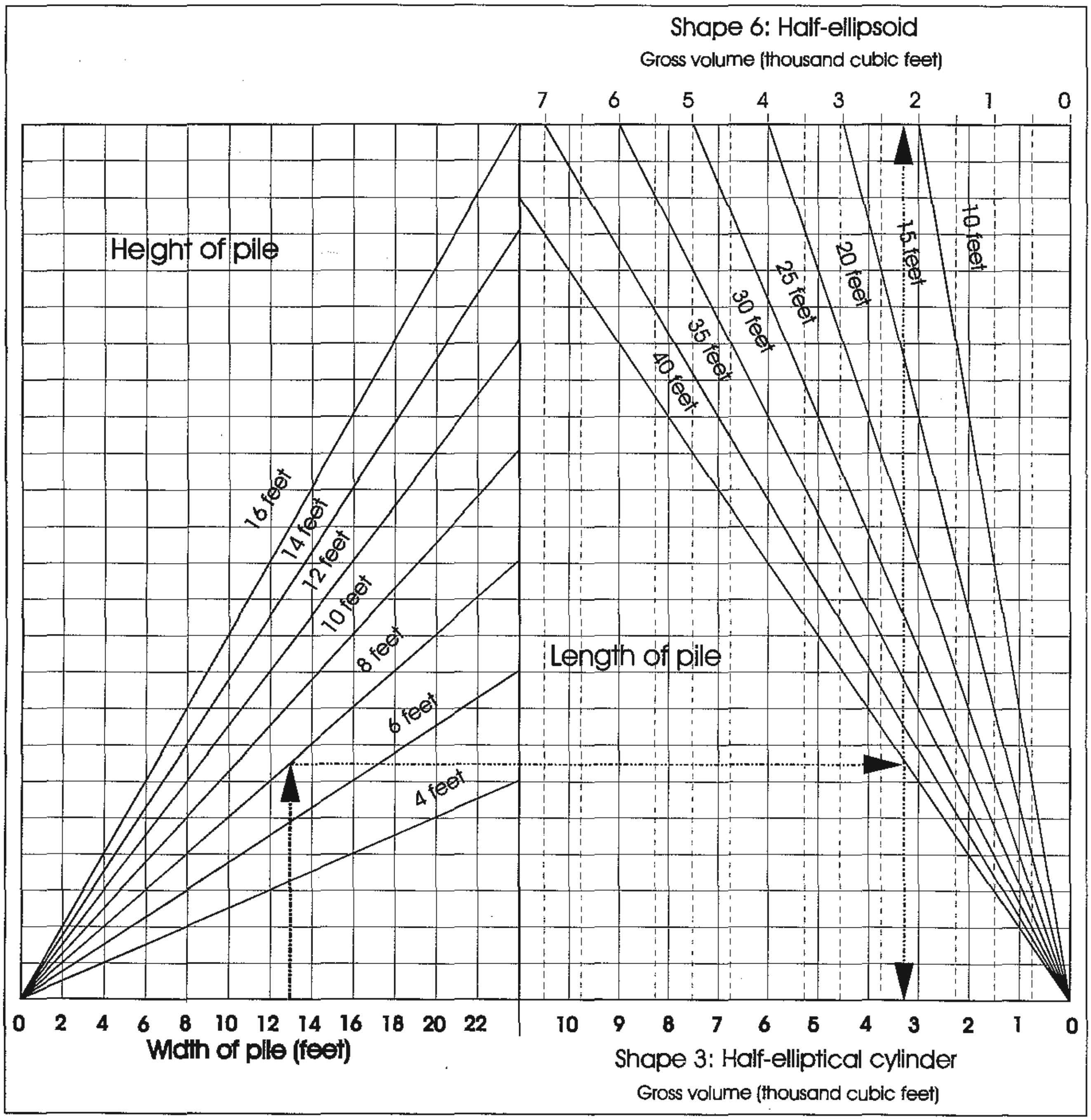

Figure 5—The nomagram assists in determining gross pile volumes (X-axes at upper and lower right) for shape codes 6 and 3, respectively, by width (X-axis at lower left), height (diagonal lines at left), and length (diagonal lines at right).

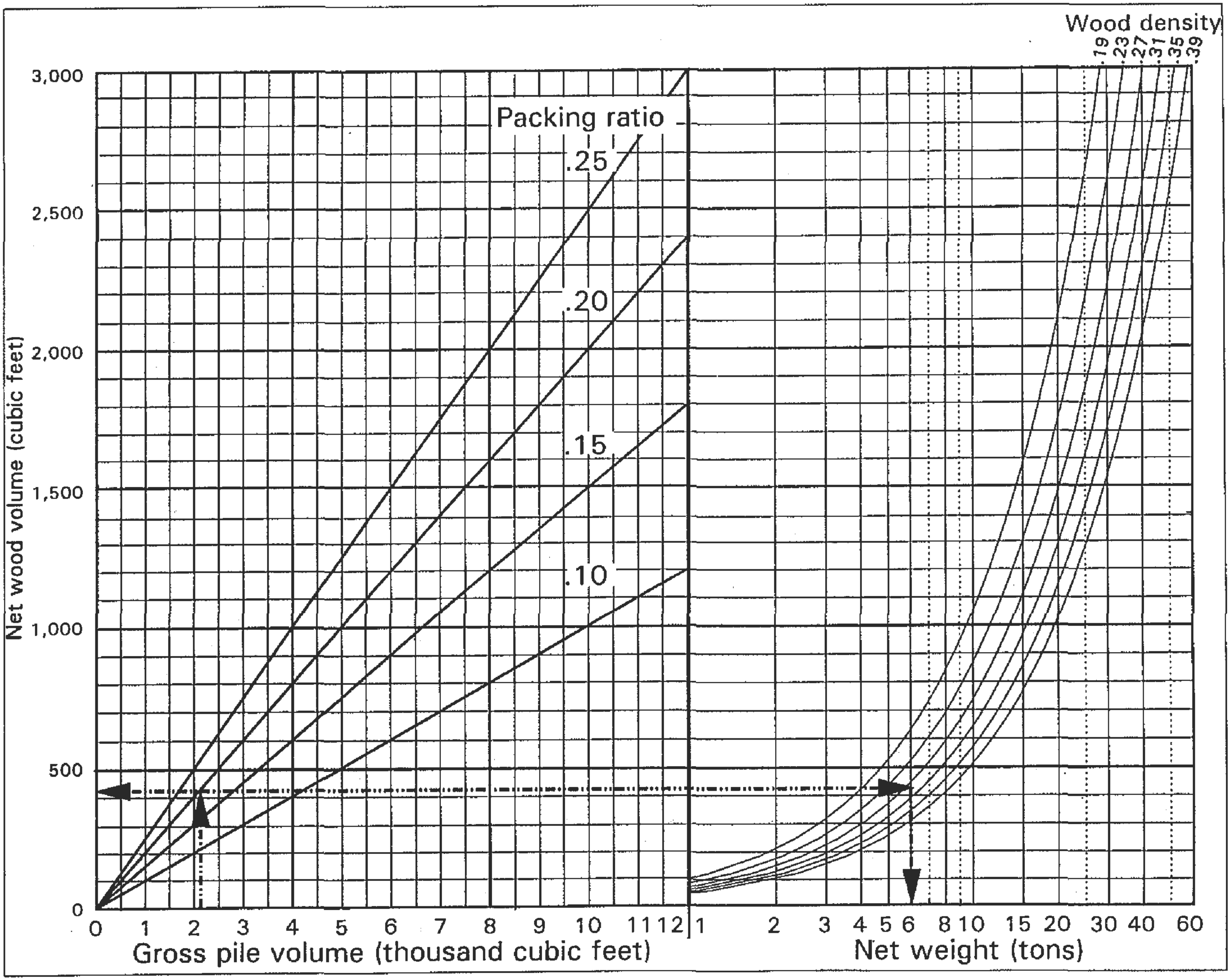

Figure 6—The nomagram assists in determining net mass of the wood (X-axis at right) by using gross pile volume (X-axis at left), packing ratio (diagonal lines at left), net wood volume (Y-axis), and wood density (curves at right).

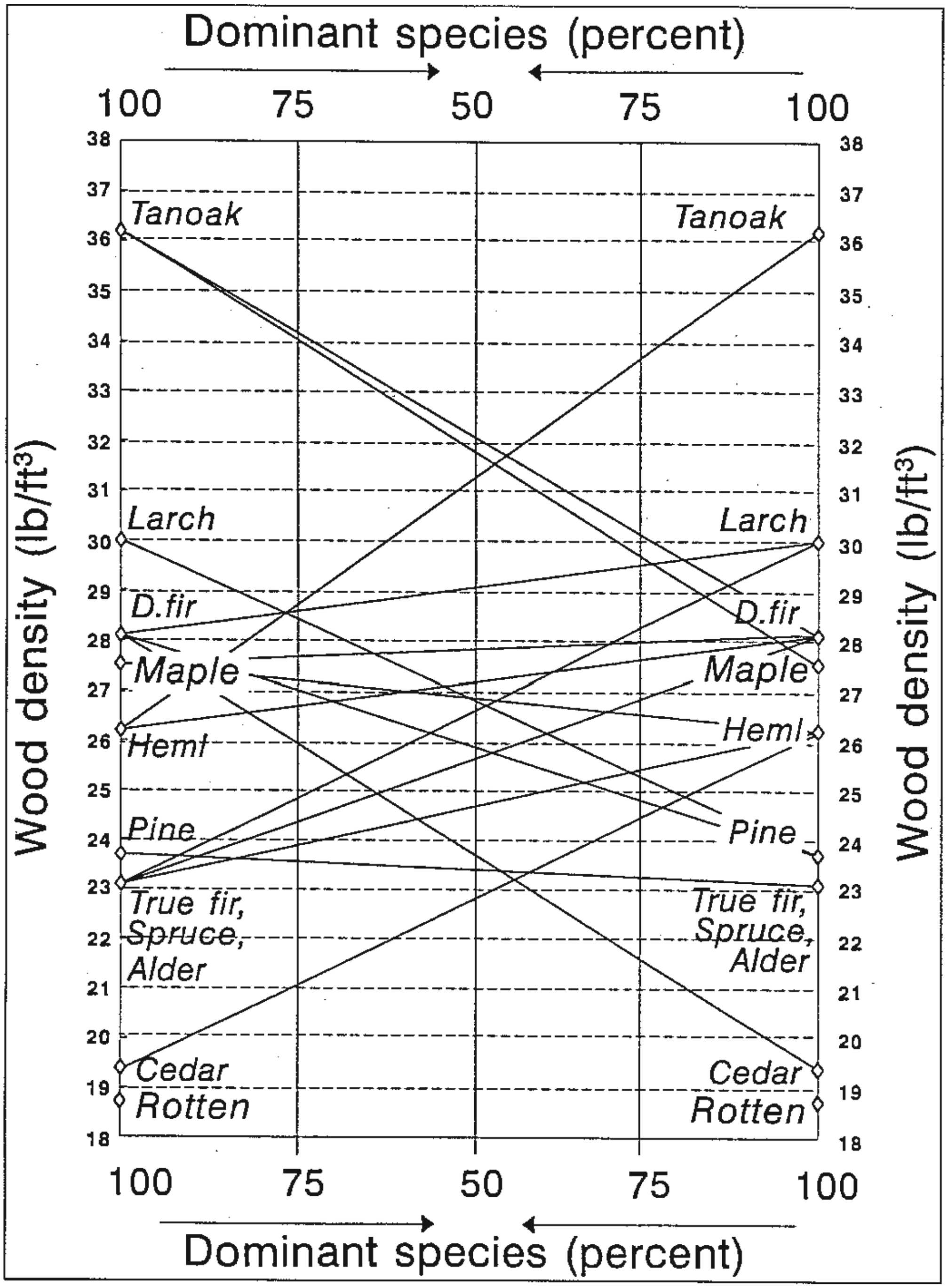

Figure 7—The nomagram can be used to calculate a weighted-average wood density (Y-axes) for two species by finding a point along a diagonal line representing two species that intersects with a vertical line indicating the correct proportion of the two species.

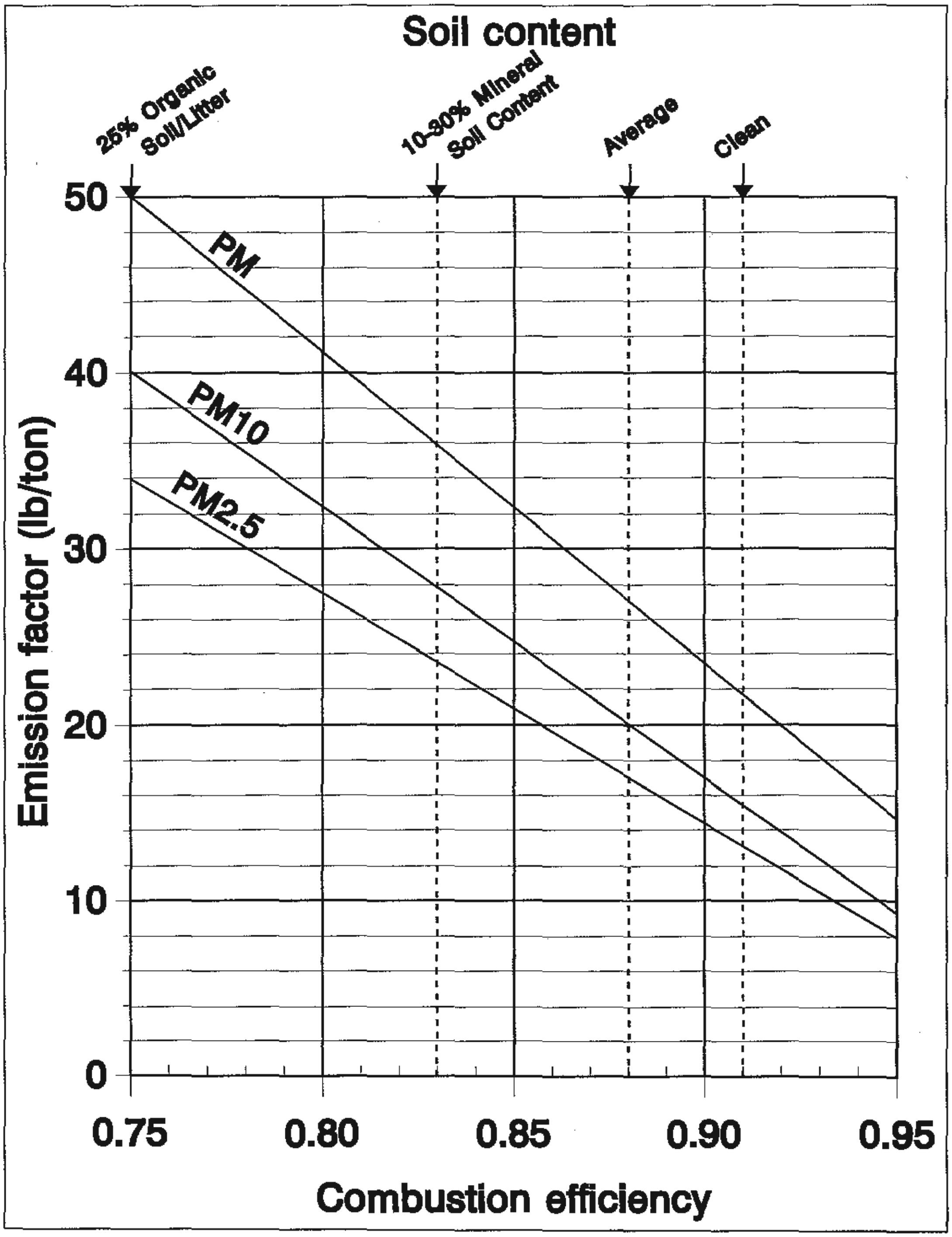

Figure 8—An appropriate emission factor for PM, PM2.5, and PM10 can be determined from knowledge of the relative amount of soil in the pile.

Hardy, Colin C. 1996. Guidelines for estimating volume, biomass, and smoke production for piled slash. Gen. Tech. Rep. PNW-GTR-364. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 17 p.

Guidelines in the form of a six-step approach are provided for estimating volumes, oven-dry mass, consumption, and particulate matter emissions for piled logging debris. Seven stylized pile shapes and their associated geometric volume formulae are used to estimate gross pile volumes. The gross volumes are then reduced to net wood volume by applying an appropriate wood-to-pile volume packing ratio. Next, the oven-dry mass of the pile is determined by using the wood density, or a weighted-average of two wood densities, for any of 14 tree species commonly piled and burned in the Western United States. Finally, the percentage of biomass consumed is multiplied by an appropriate emission factor to determine the mass of PM, PM10, and PM2.5 produced from the burned pile. These estimates can be extended to represent multiple piles, or multiple groups of similar piles, to estimate the particulate emissions from an entire burn project.

Keywords: Fuel, emissions, piled slash, smoke management.

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#2101

MODELING CARBON STORES IN OREGON AND WASHINGTON FOREST PRODUCTS: 1900–1992

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Abstract. A new model, FORPROD, for estimating the carbon stored in forest products, considers both the manufacture of the raw logs into products and the fate of the products during use and disposal. Data for historical patterns of harvest, manufacturing efficiencies, and product use and disposal were used for estimating the accumulation of carbon in Oregon and Washington forest products from 1900 to 1992. Pools examined were long- and short-term structures, paper supplies, mulch, open dumps, and landfills. The analysis indicated that of the 1,692 Tg of carbon harvested during the selected period, only 396 Tg, or 23%, is currently stored. Long-term structures and landfills contain the largest fraction of that store, holding 74% and 20%, respectively. Landfills currently have the highest rates of accumulation, but total landfill stores are relatively low because they have been used only in the last 40 years. Most carbon release has occurred during manufacturing, 45% to 60% lost to the atmosphere, depending upon the year. Sensitivity analyses of the effects of recycling, landfill decomposition, and replacement rates of long-term structures indicate that changing these parameters by a factor of two changes the estimated fraction of total carbon stored less than 2%.

1. Introduction

Eighteen years after Baes et al. (1977) first posed the question, uncertainty remains about the role of terrestrial biota in the global carbon cycle. On one hand, reconstruction of past land-use change indicates that the terrestrial biota is a net source of 0.4 to 2.6 Pg ($Pg = 10^{15}$ g) C year^{-1} (Dale et al., 1991; Dixon et al., 1994). On the other hand, 'deconvolution' studies (which estimate terrestrial flux by subtracting atmospheric increases and ocean uptake from the efflux of fossil fuels) indicate that the terrestrial biota is currently a small sink of less than 0.3 Pg (Post et al., 1990).

This discrepancy of 0.7–2.9 Pg C year^{-1} could be caused by several factors. First, uncertainty remains about the carbon uptake rate of oceans (Post et al., 1990; Tans et al., 1990; Watson et al., 1991). Second, major uncertainties also remain concerning land-use estimates. Some studies have indicated that carbon flux from non-tropical forests is close to being balanced (Houghton et al., 1987), others that some non-tropical areas may be carbon sinks (Kauppi et al., 1992; Kurtz et al., 1992; Turner et al., 1993). The differing estimates may result from the different definitions of the aerial extent of ecosystems, and different data for disturbance rates, carbon stores associated with living biomass (Brown et al., 1989), and soil carbon (Schlesinger, 1984; Post et al., 1982). Carbon stores in many ecosystems may change, as when fuel accumulates after fire suppression, without a change

Climatic Change **33:** 521–550, 1996.

in land cover-type (Brown et al., 1991; Houghton, 1991). Finally, estimates of atmospheric fluxes may differ because major pools such as soil, woody slash, and forest products are treated inconsistently.

While many earlier studies have provided insight into ecosystem factors controlling carbon balance, they cannot be used for estimating atmospheric fluxes because they exclude forest products (Armentanto and Ralston, 1980; Cooper, 1983; Cropper and Ewel, 1987) or they have modeled them in a simple fashion with a constant rate of product loss to the atmosphere (Houghton et al., 1983; Harmon et al., 1990; Dewar, 1991; Hall and Uhlig, 1991; Kurz et al., 1992). The latter analyses, while more inclusive, contain many uncertainties and do not present the basis for determining the rates of manufacturing efficiency and forest-product life spans. Harmon et al. (1990) calculated the mix of products from published conversion factors that describe the flow of raw materials through the manufacturing process. Kurtz et al. (1992) used a similar approach to determine the mix and then modeled the long-term accumulation of these materials. Despite increased realism, neither of these two studies allowed for changes in manufacturing efficiencies, product use, or disposal over time.

To refine estimates of the carbon flux associated with land-use change, we have developed an analysis system that historically reconstructs the flow of carbon into and out of forest products. This paper describes this new model which is called FORPROD (Forest Products). While FORPROD is currently applied to the Pacific Northwest, it can be used in any region where basic timber-harvest and manufacturing data are available. FORPROD is part of a larger study designed to estimate the effect of land-use change and timber harvest on the carbon balance of Oregon and Washington. It estimates stores of carbon in forest products as part of the larger system of models that predicts changes in carbon stores within the forest ecosystem after timber harvest (Cohen et al., 1992, 1994). FORPROD considers (1) the amount of raw material (i.e., logs) that is converted to products (e.g., lumber) during manufacturing, and (2) the accumulation of forest products as they are used or disposed. Products considered by the model are lumber, plywood, paper (including paper board), mulch, and fuel. The fate of these major forest products in use as short- and long-term structures, paper supplies, mulch, open dumps, and landfills is followed. Processes considered during use are decomposition, replacement of structures, and recovery and recycling of disposed paper and wood into new products. The data come from Oregon and Washington, which have produced approximately 20% of the forest products in the United States for the last half century (Powell et al., 1993).

First we give an overview of harvest and manufacturing – the sources of data, assumptions about them, and conversions. Second, we describe the model. Third, the parameters tested in sensitivity analyses are discussed. We were particularly interested in the sensitivity of the model to historical change because such change has commonly been ignored in past studies. Given that manufacturing efficiency and the longevity of forest products and wastes have generally increased with time, carbon release may be substantially underestimated if the parameters are defined only by the most recent period. Fourth, harvested carbon is tracked through manufacture of products and disposal. Finally, we use the model to make a preliminary estimate of the carbon that has been stored in forest products produced in Oregon and Washington from 1900 to 1992.

2. Harvest and Manufacturing Overview

Our analysis included only the fate of logs harvested for industrial purposes within Oregon and Washington and not wood harvested for firewood, despite the fact that such carbon is rapidly released to the atmosphere. There are few statistics on the volume of firewood, as it is generally harvested on a small scale (i.e., for individual households). We also did not consider the fate of logs imported to Oregon and Washington for manufacturing. Our assessment of the effect of timber harvest on carbon sequestration in the two states is designed to couple changes in the forest ecosystem to the fate of the forest products, and to solve the flux to the atmosphere by mass balance. For this approach to work, however, we must consider a closed system; inclusion of carbon harvested outside the location of interest would create an open system that could not be internally balanced. Finally, we did not include the use of fossil fuels for harvesting and processing carbon into forest products. The release of fossil-fuel carbon is usually considered separately from releases related to land-use (Houghton et al., 1983; Dewar, 1991; Hall and Uhlig, 1991); we therefore follow this convention and consider only the fate of carbon produced within the forest ecosystem.

The model first converts harvested tree volumes to carbon and then estimates the fraction of raw materials converted to forest products. These values are then used by the carbon-stores portion of FORPROD to estimate the input rates to the various forest-product pools. The model can be used in two modes, one with a constant rate of manufacturing efficiency, the other with a time series of changing rates of manufacturing efficiency. In the standard simulation we used the latter approach.

2.1. HARVEST OF RAW MATERIALS

Predicting the mass of forest products produced for a given year first requires that the volume of boles harvested be entered into FORPROD. We therefore compiled published harvest statistics for Oregon and Washington from 1900 to 1992 (Johnson, 1941a, b; Moravets, 1949a, b; Wall, 1972; Warren, 1993). As FORPROD does not consider the fate of logs used for firewood, we did not include firewood in the analysis. Besides historical records, FORPROD can also use output from land-use models for the volume of trees removed from forests.

As FORPROD tracks the fate of carbon, the reported units of wood volume must be converted into carbon. The first step was to convert Scribner board feet to total cubic-foot volume (ft^3), which required data for the mixture of species and the size of the logs for the most exact conversion factors (Hartman et al., 1976). Unfortunately these data are not reported with harvest statistics, and therefore the conversion factors had to be approximated by regressing the reported cubic-foot volume of growing stock against the reported board-foot volume of saw timber (Bassett and Oswald, 1983; Gedney et al., 1986a, b, 1987, 1989; MacLean et al., 1992). That analysis gave an average conversion factor for common conifer species in Oregon and Washington of 0.234, within a range of 0.221 to 0.265, depending upon species. We used an average conversion factor weighted by the growing stock volume of each species reported in recent timber surveys (Gedney et al., 1986a, b, 1987, 1989; MacLean et al., 1992). The equation used to convert Scribner board foot volume (VolSbft) to cubic-foot volume (Volcft) was:

$$\text{Volcft} = 0.234 * \text{VolSbft}$$

The cubic-foot volume of wood harvested was then converted to the total cubic-meter volume (Volcm) by:

$$\text{Volcm} = 0.028 * \text{Volcft}.$$

The mass of organic matter harvested as wood (OGMWood) was calculated by multiplying cubic-meter volume by wood density (DenWood) of the major species:

$$\text{OGMWood} = \text{DenWood} * \text{VolCm}.$$

The density for current forest conditions was calculated by weighting the wood density of each species (Maeglin and Wahlgren, 1972) by the proportion of the growing stock it comprised in recent timber surveys (Gedney et al., 1986a, b, 1987, 1989; Maclean et al., 1992). The mean density of logs harvested east of the Cascade Mountains was 0.40 Mg m^{-3}; west of the Cascade Mountains it was 0.43 Mg m^{-3}. We then calculated density for earlier periods, finding that it has changed little over the last 50–60 years: 0.435 Mg m^{-3} for west-side forests (Andrews and Cowlin, 1934) and 0.389 Mg m^{-3} for east-side forests (Cowlin and Wyckoff, 1944). Finally, the mass of organic matter of wood (OGMWood) was multiplied by 0.52, the carbon content of coniferous wood, to convert the carbon mass of wood (CWood) (Wilson et al., 1987; Birdsey, 1992):

$$\text{CWood} = 0.52 * \text{OGMWood}.$$

2.2. LOG DISPOSITION

Once harvested, Oregon and Washington trees are used chiefly as saw logs for lumber production, veneer logs for plywood production, and pulp logs for paper

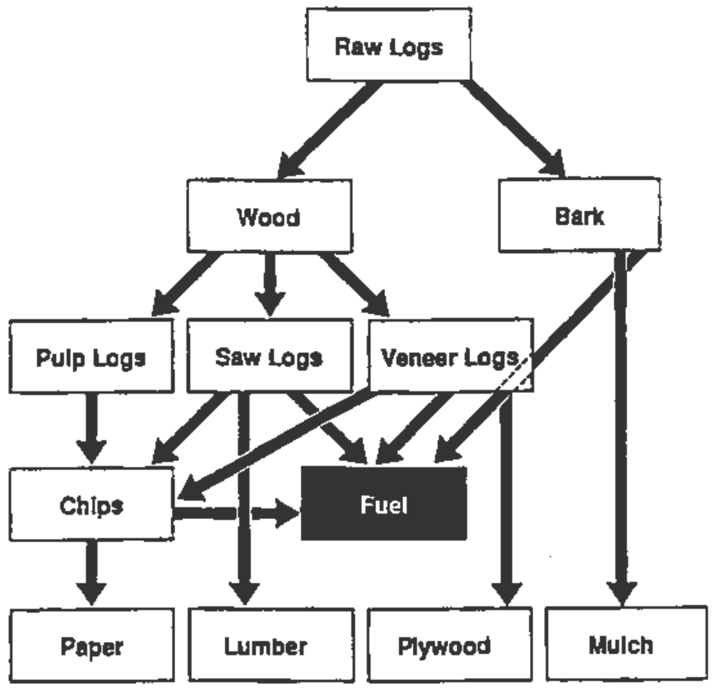

Figure 1. FORPROD flow of carbon through harvest and manufacturing. Fuels are released to the atmosphere in the year of harvest.

production (Figure 1). We excluded other minor uses, such as for railroad ties and poles (< 1% of total). We assumed that exported logs were used in the same way as logs used within the United States.

The equations for calculating carbon mass in saw logs (SawLog), veneer logs (VenLog), and pulp logs (PulpLog) were:

$$\text{SawLog} = \text{FSawLog} * \text{CWood}$$

$$\text{VenLog} = \text{FVenLog} * \text{CWood}$$

$$\text{PulpLog} = \text{FPulpLog} * \text{CWood}.$$

where FSawLog, FVenLog, and FPulpLog are the fractions of each used in any given year.

Changes in the use of logs over time was compiled from published harvest reports (Moravets, 1950; Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard, 1984; Howard and Ward, 1988; Larsen, 1990, 1992; Howard and Ward, 1991). In years in which there were no reports, we used linear interpolation for estimating values.

2.3. BARK REMOVAL AND PROCESSING

Before logs are used for lumber, plywood, or pulp production, the bark is removed. The mass of carbon in bark (CBark) was calculated as:

$$\mathrm{CBark} = \mathrm{FBark} * (\mathrm{CWood}/(1 - \mathrm{FBark}))$$

where FBark is the fraction of logs that is bark. The fraction varies among species (Wilson et al., 1987), so averages were calculated by multiplying the portion of growing stock of a species by the fraction in bark. The values derived were 15% for logs east of the Cascade Mountains, 12% for logs west of the mountains.

Bark is currently used for mulch (BMulch), hogged fuel (BFuel), and chips (BChips) which were calculated as:

$$\mathrm{BMulch} = \mathrm{FMulch} * \mathrm{CBark}$$

$$\mathrm{BFuel} = \mathrm{FBFuel} * \mathrm{CBark}$$

$$\mathrm{BChips} = \mathrm{FBChips} * \mathrm{CBark}$$

where FMulch, FBFuel, and FBChips are the fractions of bark being used for each in any given year. Historical patterns of bark use were compiled from the literature (Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992). Linear interpolation was used when data were missing. We assumed that before 1960, when reporting began, bark was primarily used as fuel.

2.4. LUMBER PRODUCTION

The primary products produced from saw logs are lumber, chips for paper production, and hogged fuel. The rest is disposed waste. During lumber production, a large fraction of wood waste is generated in the form of slabs, sawdust, planer shavings, and defective lumber. We assumed that this material was disposed of as either chips or hogged fuel. In reality, some of it was either decomposed or incinerated without energy recovery. As the consequences for carbon stores of these two processes were similar to consequences for hogged fuels, we combined the three flows into a fuel category.

The production of lumber (Lumber), chips (SLChip), and hogged fuel (SLFuel) from saw logs was calculated as:

$$\mathrm{Lumber} = \mathrm{FLumber} * \mathrm{SawLog}$$

$$\mathrm{SLChip} = \mathrm{FSLChip} * \mathrm{SawLog}$$

$$\mathrm{SLFuel} = \mathrm{FSLFuel} * \mathrm{SawLog}$$

where FLumber, FSLChip, and FSLFuel are the fractions of saw logs being used for each in any given year. Historical changes in saw-log processing efficiency (Hodgson, 1931; Corder et al., 1972; Lane et al., 1973a, Hartman et al., 1976; Willits and Fahey, 1988; Briggs, 1993) and waste disposition (Hodgson, 1931; Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Hartman et al., 1976; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992) were compiled from historical summaries of the forest-products sector. In years without reported data, we used linear interpolation for estimating values.

2.5. PLYWOOD PRODUCTION

The primary products produced from veneer logs are plywood, hogged fuel, chips for paper production, and dimensional wood from cores left after veneer peeling. We combined plywood and lumber from peeler cores as one product. Wood waste resulting from plywood production was disposed as either chips or hogged fuels. As with sawlogs, veneer wood waste that was decomposed or incinerated was treated as hogged fuel.

The production of plywood (Plywood), chips (VLChip), and hogged fuel (VLFuel) from veneer logs (VenLog) was calculated as:

$$\text{Plywood} = \text{FPlywood} * \text{VenLog}$$

$$\text{VLChip} = \text{FVLChip} * \text{VenLog}$$

$$\text{VLFuel} = \text{FVLFuel} * \text{VenLog}$$

where FPlywood, FVLChip, and FVLFuel are the fractions of veneer logs being used for each in any given year. Historical changes in the efficiency of veneer-log processing (Corder et al., 1972; Lane et al., 1973b, Woodfin, 1973, Hartman et al., 1976; Adams et al., 1986; Brigs, 1993) and waste disposition (Gedney, 1956; Cowlin and Forster, 1965; Manock et al., 1970; Corder et al., 1972; Schuldt and Howard, 1974; Bergvall et al., 1975; Howard and Hiserote, 1978; Howard, 1984; Howard and Ward, 1988, 1991; Larsen, 1990, 1992) were compiled from summaries for the forest-products sector. In years without reported data, we used linear interpolation for estimating values.

2.6. PAPER PRODUCTION

During the processing of chips and pulp logs into paper, material is lost. The overall efficiency of paper production depends strongly on the process used. Although the efficiency of each pulping process has remained relatively constant with time, the proportion of paper produced by each process has changed markedly. To take into consideration these historical changes, we calculated a weighted average efficiency

Table I

Efficiency of the eight types of softwood pulping processes used to predict average efficiency

Pulping process	Efficiency (%)
Mechanical pulping	95
Soda pulping	87
Defibrating	85
Semichemical	70
Screenings/off quality	50
Sulfate-bleached	45
Sulfate-semibleached	50
Sulfate-unbleached	55
Sulfite-bleached	43
Sulfite-unbleached	48
Dissolving and special alpha	35

for Oregon and Washington and the United States by multiplying the efficiency (Smook, 1982) of each of the main categories of wood pulping processes (Table I) by the respective quantities of pulp produced each year.

The treatment of waste from wood pulp production also varies with the process. Waste from sulfite and sulfate pulping, the major processes in Oregon and Washington, is burned as fuel and to recover sulfur. Other waste is digested in settling ponds or disposed in landfills. In the model, we assumed that material not resulting in paper formation was burned, or that it decomposed rapidly in waste-water treatment.

The amount of paper produced each year (PapIn) was calculated as:

$$\text{PapIn} = \text{PapCR} * \text{EffPP} * (\text{PulpLog} + \text{SLChip} + \text{VLChip})$$

where EffPP is the efficiency of converting chips to paper for each year as determined above, and PapCR is the reduction in carbon content brought about by the paper-manufacturing process. For all forest products except paper, the carbon content was assumed to be equal to that of raw wood (52%). But since cellulose is the primary product of paper manufacturing, and the carbon content of pure cellulose is 23% lower than that of whole wood, carbon stored in paper products was adjusted to an average content of 40%.

2.7. WASTE DISPOSAL

Since 1900, paper and finished wood products have been disposed of in primarily four different ways: open dumps, sanitary landfills, incineration, and recycling. Flows to landfills, incinerators, and open dumps were determined from published reports. Records of the fraction of waste disposed of in open dumps were poor, so we noted when sanitary landfills began and when open dumps were closed to account for the transition from one type of disposal to the other (Collins, 1972; Baum and Parker, 1973; Waste Age, 1979; DeGreare, 1982; EPA, 1984; Liptak, 1991). For example, in the United States, the use of sanitary landfills was not accepted as the proper means to dispose of solid waste until after 1945 (Ham, 1972). We therefore assumed that prior to this time, solid waste was largely disposed in open dumps. The conversion rate of open dumps to sanitary landfills appears to have been low until the Resource Conservation and Recovery Act of 1976 (DeGreare, 1982; EPA, Office of Solid Waste, 1984). We therefore assumed that open dump conversion greatly increased after that point and was largely completed by 1980 (Collins, 1972; Liptak, 1991).

There are also few quantitative estimates of the amount of waste disposed by incineration. Data on the number of cities with incinerators indicate that their use increased between 1900 and 1940, but then declined as many municipalities converted to landfilling (Baum and Parker, 1973; Rathje, 1989). The first estimate of the fraction of waste incinerated is for 1960 when 30.8% of all municipal solid waste was disposed in this manner (EPA, 1990). We therefore assumed that at its peak in the 1940's, incineration would have accounted for a slightly higher fraction of waste. The decline in incineration appears to have continued from 1960 to the mid-1980's, with 20.6% incinerated in 1970, and 14.2% incinerated in 1988 (EPA, 1990). We assumed a linear rate of change over this period. Since 1988, an increase in the fraction of waste incinerated has been driven primarily by the need for waste volume reduction and energy recovery (Kiser, 1991; Schmidt, 1990).

Prior to 1960, the degree of paper recycling is difficult to document. We therefore assumed that 5% of all paper waste was recycled between 1900 and 1940, and that between 1940 and 1960 there was a linear increase from 5% to 18%, the latter value being the first reported value we could find (Liptak, 1991). After 1960 we used the time series reported by Franklin Associates (1988, 1993) and Rathje (1989) to estimate paper recycling rates.

3. Estimating Carbon Stores with FORPROD

Carbon stores in forest-product pools were tracked in short-term structures, long-term structures, paper supplies, mulch, and waste in open dumps and sanitary landfills (Figure 2). 'Mulch' refers to bark or sawdust that is composted or spread directly on the soil; 'open dumps' are disposal sites in which rates of biological decomposition and combustion are high; 'sanitary landfills' are sites with no combustion and low rates of biological decomposition. Changes in the pools are estimated with difference equations having a time step of 1 year. Input to short-term and long-term structures, mulch, and paper supplies are from the manufacturing subroutines previously described. Lumber and plywood production is divided into material added to short-term structures or long-term structures. The former include

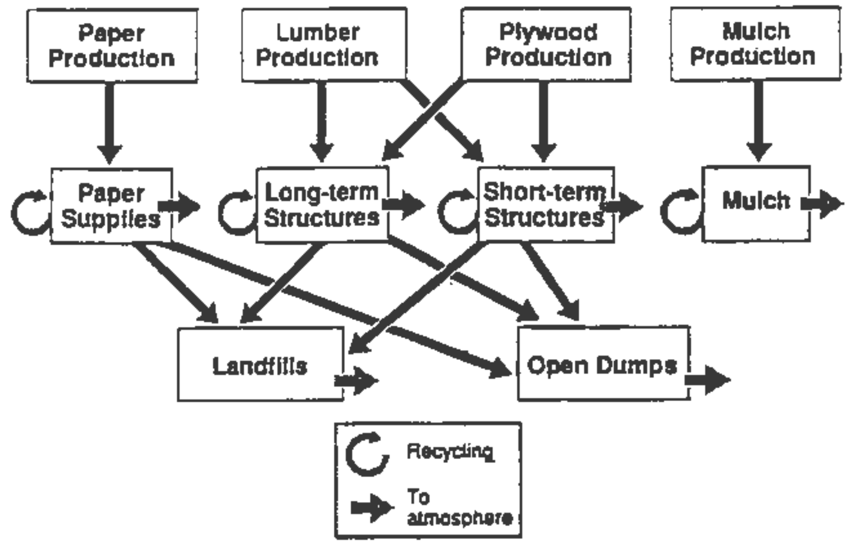

Figure 2. FORPROD flow of forest products during use and disposal. Recycling returns some material to the pool source and some to the atmosphere. All pools lose carbon to the atmosphere from decomposition or combustion.

wood in fences in decay-prone environments or in products such as pallets that have a short life span (< 20 years). The latter include buildings and other forms of wood with life-spans exceeding 20 years. All paper supplies, including paperboard, are tracked.

Waste (W) lost in disposal and decomposition of products is influenced by the rate of recycling and recovery into new products. We assumed that products recovered from a given source would be used in a similar way, that is, that paper would be recovered as paper, short-term structures as short-term structures, and long-term structures as long-term structures.

The following sections give the assumptions and equations for each FORPROD pool. Table II summarizes the values of the parameters used in the standard simulation.

3.1. MULCH

The change in mulch stores (Mulch) are calculated as:

$$\Delta\text{Mulch} = \text{MulIn} - \text{MulDK} * \text{Mulch}$$

Where MulIn is the annual input of mulch and compost as predicted from manufacturing functions and MulDK is the decomposition-rate constant. We assumed a decomposition-rate constant of 0.10 year^{-1}, a value slightly higher than that for bark in a natural setting (Harmon and Sexton, 1995).

Table II

Values for decomposition, replacement, and recycling parameters used in the standard simulation (See text equations in Section 3 for details of forest-product pools)

Process pool	Parameter	Value
Decomposition-rate constants		
Mulch	MulDK	0.10 $year^{-1}$
Short-term structures	SSDK	0.05 $year^{-1}$
Long-term structures	LSDK	0.01 $year^{-1}$
Open dump	DumpDk	0.30 $year^{-1}$
Landfill	LFillDk	0.005 $year^{-1}$
Replacement-rate constants		
Short-term structure	SWasteMax	0.10 $year^{-1}$
Long-term structures	LWasteMax	0.01 $year^{-1}$
Paper	PWasteMax	0.60 $year^{-1}$
Recycling recovery rate		
Short-term wood structure	WRcvr	90%
Long-term wood structure	Wrcvr	90%
Paper	PRcvr	90%

3.2. SHORT-TERM STRUCTURES

The change in short-term structures (SStr) is a function of input from lumber and plywood and loss from decomposition in use and replacement:

$$\Delta SStr = SSIn - SSDK * SStr - SWaste * SStr$$

where SSIn is the input from lumber and plywood, SSDK is the in-place decomposition-rate constant of short-term structures, and SWaste is the rate constant of replacement. SSIn is estimated from the production of lumber and plywood.

Because there is little direct data for the fraction of lumber and plywood used in short-term structures, we estimated that all wood used for shipping and half of the wood used in manufacturing would be used in short-term structures, which would mean 18% and 5%, respectively, were used in short-term structures between 1962 and 1986 (Haynes, 1990), such that

$$SSIn = 0.18 * Lumber + 0.05 * Plywood.$$

SWaste is a function of the rate of recycling (WRcycl) and rate of recovery into new forest products (WRcvr) so that

$$SWaste = SWasteMax * (1 - WRcycl * WRcvr)$$

where SWasteMax is the maximum rate of replacement of short-term structures. This equation reduces the flow of waste to zero only when all material is recycled and completely recovered to new products. In this analysis we assumed that 95% of all short-term structures would be replaced within 30 years (SWasteMax = 0.10 year^{-1}), and that the decomposition-rate constant would be 0.05 year^{-1} (95% decomposing in 60 years). We also assumed that 90% of the recycled material would be recovered as 'new' short-term structures and that the remaining 10% would be disposed in open dumps, landfills, or incinerators.

3.3. LONG-TERM STRUCTURES

The change in long-term structures (LStr) is a function of input from lumber and plywood and loss from decomposition in use and replacement:

$$\Delta \text{LStr} = \text{LSIn} - \text{LSDK} * \text{LStr} - \text{LWaste} * \text{LStr}$$

where LSIn is the input from lumber and plywood, LSDK is the in-place decomposition-rate constant of long-term structures, and LWaste is the rate constant of replacement. LSIn is estimated from the production of lumber and plywood, so that

$$\text{LSIn} = 0.82 * \text{Lumber} + 0.95 * \text{Plywood}$$

where coefficients are the compliment of those used to predict the fraction going to short-term structures.

LWaste is a function of the rate of lumber and plywood recycling and rate of recovery into new forest products so that

$$\text{LWaste} = \text{LWasteMax} * (1 - \text{WRcycl} * \text{WRcvr})$$

where LWasteMax is the maximum rate of replacement of long-term structures. As with short-term structures, this equation reduces the flow of waste to zero only when all material is recycled and completely recovered to new products. In this analysis we assumed that 95% of all long-term structures would be replaced within 300 years (LWasteMax = 0.01 year^{-1}) (Marin, 1978), and that the in-place decomposition-rate constant would be 0.01 year^{-1} (95% decomposing in 300 years). We also assumed that 90% of the recycled material would be recovered as 'new' long-term structures and that the remaining 10% would be disposed in open dumps, landfills, or incinerators.

3.4. PAPER STORES

The changes in stores of paper supplies (Paper) are a function of input from paper production (PapIn) and loss from disposal (PWaste):

$$\Delta \text{Paper} = \text{PapIn} - \text{PWaste} * \text{Paper}$$

PWaste is a function of the rate of paper recycling (PRcycl) and rate of recovery into new paper products (PRcvr), so that

$$\text{PWaste} = \text{PWasteMax} * (1 - \text{PRcycl} * \text{PRcvr})$$

where PWasteMax is the maximum rate of paper disposal. This equation reduces the flow of paper waste to zero only when all paper is recycled and completely recovered into 'new' paper. In this analysis we assumed that 95% of all paper supplies would be replaced within 5 years (PWasteMax $= 0.60$ year^{-1}), that 90% of recycled paper would be recovered as 'new' paper, and that 10% of the recycled paper would be disposed in open dumps, landfills, or incinerators.

3.5. OPEN DUMPS

Before the advent of sanitary landfills, paper and wood products in open dumps underwent rapid decomposition or combustion. The model accounts for the transition from open dumps to sanitary landfills. Changes in open dump stores (Dump) are a function of input from short- and long-term structures and paper supplies minus the removal from decomposition and combustion:

$$\Delta\text{Dump} = \text{LDump} * \text{LStr} + \text{SDump} * \text{SStr} + \text{PDump} * \text{Paper} - \text{DumpDk} * \text{Dump}$$

where LDump, SDump, and PDump are, respectively, the flows of waste from long- and short-term structures and paper to dumps, and DumpDk is the combined decomposition and combustion-rate constant for material in open dumps. The flow of waste into dumps depends on the amount of waste incineration and the flow of waste to landfills. For example, LDump, the rate at which long-term structural waste is added to dumps, is calculated as

$$\text{LDump} = \text{FWDump} * \text{LWaste} * (1 - \text{WoodIncin})$$

where FWDump is the fraction of wood waste going to dumps, LWaste is the rate at which long-term structures are replaced (as calculated under loss from long-term structures), and WoodIncin is the fraction of wood being incinerated. The other flows to dumps are calculated in a similar manner.

In this analysis we assumed that 95% of the material added to open dumps would decompose or be burned within 10 years, therefore we used a DumpDk rate-constant of 0.30 year^{-1}.

3.6. LANDFILLS

In modern sanitary landfills, solid waste is strongly compacted, covered, or capped with a layer of soil in a dry, anaerobic, and acidic environment. Little or no decay takes place (Rathje, 1989; Liptak, 1991), thus little carbon reenters the atmosphere.

Changes in landfill stores (LFill) are calculated as the difference between input from paper, short- and long-term structures, and the material decomposed:

$$\Delta \text{LFill} = \text{LLFill} * \text{LStr} + \text{SLFill} * \text{SStr} + \text{PLFill} * \text{Paper} - \text{LFillDk} * \text{LFill}$$

LLFill, SLFill, and PLFill are, respectively, the flows of waste from long- and short-term structures and paper to landfills, and LFillDk is the decomposition-rate constant for material in landfills. Although much of the carbon leaving landfills is in the form of methane (CH_4), no differentiation is made in the model. We did not partition flows from landfills into CO_2 and CH_4 for several reasons. First, there are few data on the rate of CO_2 versus CH_4 production during the course of decomposition. Second, even if decomposers produced only CH_4 in landfills, an undetermined and potentially large fraction may be converted to CO_2 by energy recovery or other combustion processes. As these uncertainties have no influence on carbon stores, we have defered this aspect of the problem until better data are gathered.

The flow of waste into landfills is calculated in a similar manner to the flow into open dumps and depends on the amount of waste incineration and the alternative flow of waste to open dumps. For example, PLFill, the rate paper is added to landfills, is calculated as:

$$\text{PLFill} = (1 - \text{FPDump}) * \text{PWaste} * (1 - \text{PaperIncin})$$

where FPDump is the fraction of paper disposed in open dumps, PWaste is the fraction of paper replaced as calculated under paper stores, and PaperIncin is the fraction of paper waste incinerated. The other flows to landfills are calculated in a similar manner.

4. Sensitivity Analyses

The simulation using the best estimate of parameters is called the 'standard run'. Some of the parameters (see Table II) used in this run were constant over the entire period, whereas others varied over time. The variations in the latter set of parameters represented the best or most likely historical reconstruction of trends over the simulation period. Additional simulation runs were made to test the sensitivity of the model to parameters of major concern. The details of each run are described in the following sections, named after the parameter that was tested. In all of these tests, standard-run values were used except for the parameter in question.

4.1. TRANSITION TO LANDFILLS

To assess the sensitivity of simulations to the flow of waste to open dumps versus landfills, we considered three scenarios: our best reconstruction of the time of transition from open-dump to landfill disposal, a transition 5 years earlier than estimated, and a transition 10 years earlier than estimated. Unless specifically noted, the standard run was the 'best reconstruction' scenario.

4.2. LANDFILL DECOMPOSITION

The effect of three rates of decomposition of waste in landfills was explored because there are no quantitative measurements of this process and we were unable to determine an upper limit on the expected lifetime of landfill material. We therefore examined a low-decomposition scenario in which 95% decomposition occurred within 1200 years, a medium-decomposition scenario in which it occurred in 600 years, and a high-decomposition scenario in which it occurred in 300 years. The high, medium and low rate-constants were 0.01, 0.005, and 0.0025 year^{-1}, respectively. The medium landfill decomposition rate-constant was used for the standard run.

4.3. RECYCLING RATES

Although recycling rates for paper have been compiled annually since 1960 (Franklin Associates, 1993) the actual rates are debatable because some 'recycled' paper may be used for fuel or products subject to high rates of decomposition (e.g., hydromulch). We explored the effects of recycling by doubling and halving the reported rates.

4.4. LONG-TERM STRUCTURE REPLACEMENT

There are few estimates of replacement rates of long-term structures. Longevities of 100–150 years are often used (Harmon et al., 1990), but the only rigorous survey we found indicated a longevity of 300 years (Marcin, 1978). We used three rate-constants of replacement to examine the effects of this parameter: 0.01 year^{-1} (the standard run), one half of that value (0.005 year^{-1}), and double that value (0.02 year^{-1}).

4.5. TEMPORAL VARIATION

In the standard run, the values of some parameters, such as manufacturing efficiencies and waste disposal, varied over the simulation period. In many past studies, the values were held constant over the simulation period. In this set of simulations, we explored the effect of holding the parameters constant. Two fixed sets of values derived from the standard run were used: parameters specific to 1970, and parameters specific to 1990.

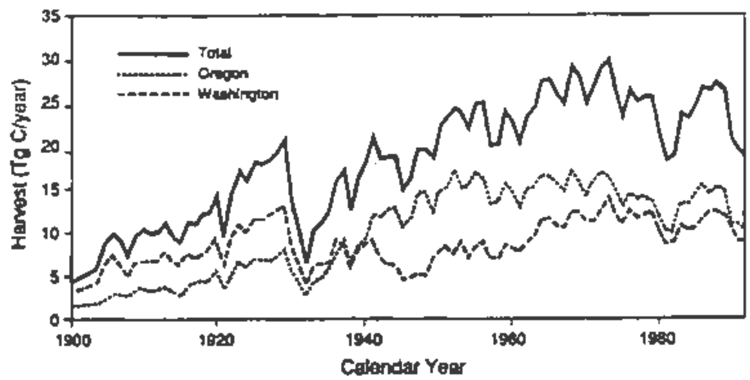

Figure 3. Historical reconstruction of the total wood and bark carbon harvested in Oregon and Washington between 1900 and 1992.

5. Harvest and Manufacturing Patterns, 1900–1992

The total amount of tree harvest from Oregon and Washington forests from 1900 to 1992 for use in wood products excluding firewood is estimated to be 1,692 Tg (Tg = 10^{12} g). The amount of carbon removed increased from an estimated 4 Tg year^{-1} in 1900 to a high of 29.9 Tg year^{-1} in 1973 (Figure 3). Since 1945, the harvest of carbon for the wood products industry from these two states has averaged 23.9 Tg year^{-1}. Fluctuations in harvest have been primarily due to economic cycles in the United States, the largest fluctuation occurring during the Great Depression in the 1930's.

5.1. RAW LOG USE

The primary use of harvested logs has been as saw logs for lumber production (Figure 4). The use of veneer logs for plywood production was relatively minor until the 1950's, when building construction increased. Pulp logs have been a minor component of the timber harvest in Oregon and Washington throughout the entire period, peaking in 1962 at approximately 16% of all logs. Since 1960, pulp logs have comprised an average of 9.8% of all logs harvested in Oregon and Washington.

5.2. BARK USE

Bark has been used primarily for fuel (Figure 5). In the mid-1960's a growing market for bark mulch arose, and since 1965 it has averaged 14.5%. Most of the remainder has been burned as hogged fuel or waste.

5.3. SAW LOG USE

The largest change in saw log production over the last 90 years has been in the use of mill waste and not in milling efficiency, as one might assume (Figure 6A).

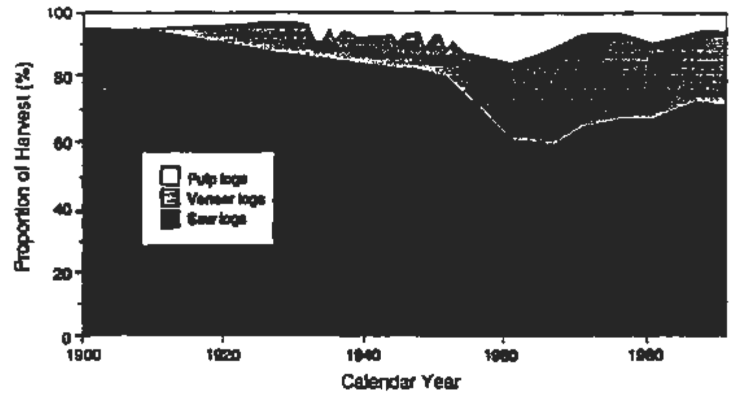

Figure 4. Historical reconstruction of the utilization of raw logs for lumber, plywood, and paper production in Oregon and Washington between 1900 and 1992.

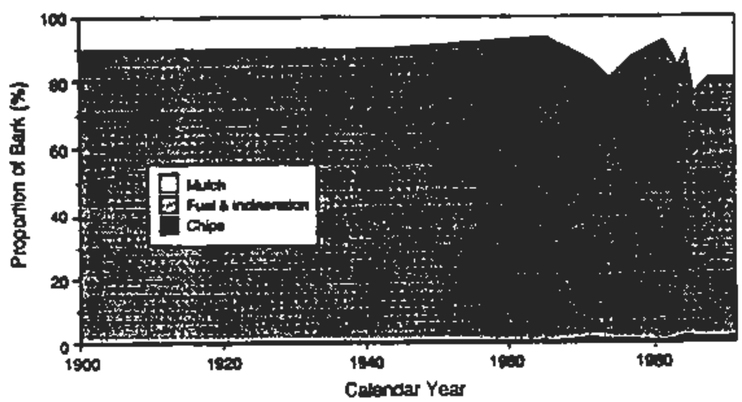

Figure 5. Historical reconstruction of the disposition of bark waste in Oregon and Washington between 1900 and 1992.

In the first half of this century, the percentage of lumber produced from saw logs in Oregon and Washington was approximately 52%. After 1945, production fluctuated, but generally declined to 40% by 1950. A decline in efficiency between 1945 and 1950 was due to additional processing, such as planing that reduced the amount of lumber by 10% (Corder et al., 1972), redefining of board-foot lumber measurement to smaller dimensions, and use of logs of smaller diameter. After the 1950's, saw mill efficiency has gradually increased because of technological improvements (Adams et al., 1986).

Saw log residue, averaging 48% of the wood, was primarily burned as waste or as hogged fuel until the late 1940's. In the mid to late 1940's, the use of the residue for pulp increased with the introduction of the sulfate pulping process. A tightening log supply and technological improvements in log barkers and chippers made it possible to use the saw log residue from Douglas-fir (*Pseudotsuga menziesii*

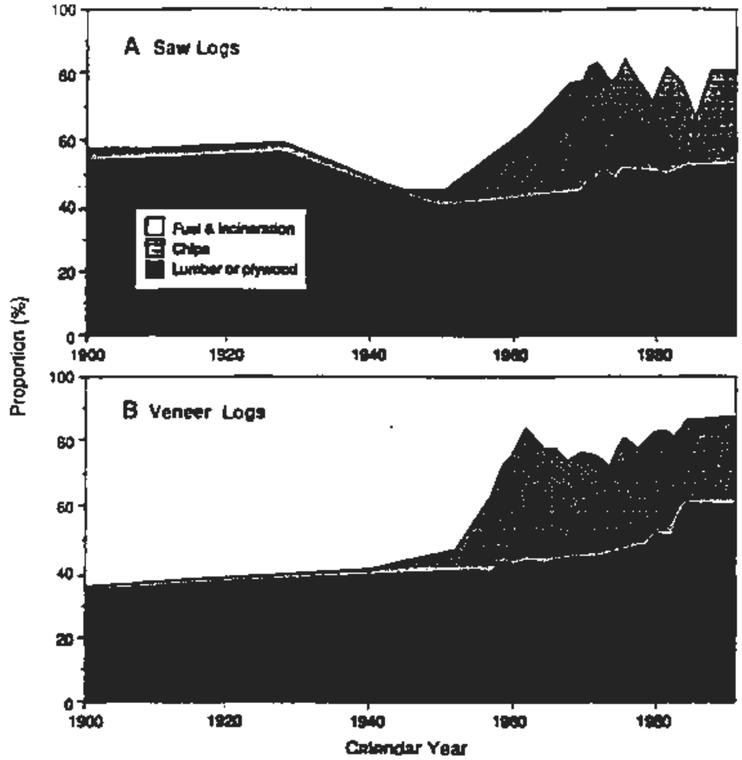

Figure 6. Historical reconstruction of the disposition of (A) saw logs and (B) veneer logs in Oregon and Washington, between 1900 and 1992.

(Mirb.) Franco] for pulping (Gedney, 1956). The use of log waste residue for chips increased until 1960, and since that time has averaged 27%.

5.4. VENEER LOG USE

As with saw logs, the largest change in veneer log use involved the disposition of the waste (Figure 6B). The percentage of plywood produced from veneer logs in Oregon and Washington slowly increased with efficiency from 40% in 1940 to 50% in 1980. Since the mid-1980's, technological improvements have increased the efficiency of plywood mills to approximately 61%.

Veneer log residue was primarily burned as waste or used as hogged fuel until the late 1940's, when there was an increase in the chipping of wood waste for paper production for the same reasons as for chipping of saw log residues. The fraction of veneer logs used for chips increased until 1960, and since that time has averaged 31%.

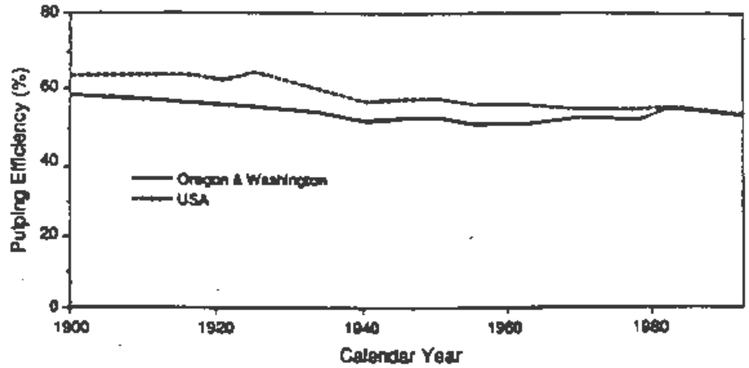

Figure 7. Estimated changes in papermaking efficiency between 1900 and 1992 in (A) Oregon and Washington and (B) the entire United States.

5.5. PAPER PRODUCTION

The overall efficiency of producing pulp for paper products declined slightly from approximately 60% to 52% between 1900 and 1992 (Figure 7). This is a trend not only for Oregon and Washington, but for the United States as a whole. The efficiency decline is due to an increase in paper production from sulfate pulping processes and a decrease in the proportion of mechanical pulping.

5.6. COMBINED PRODUCTION

The combined mass of forest products manufactured from 1900 to 1992 is primarily associated with the change in the mass harvested (Figure 8), product mass ranging from 2.04 Tg year^{-1} in 1900 to a high of 17.01 Tg year^{-1} in 1973. Changes in manufacturing efficiency and use of milling waste have also been important, and in some periods have counteracted the influence of harvest levels on production. During 1930 to 1950, for example, harvest levels increased 4-fold, but overall manufacturing efficiency (defined as the ratio of product output to harvest) declined from approximately 50% to 40%. Since 1950, the overall manufacturing efficiency has increased steadily (approximately 61% in 1992) because of changes in individual manufacturing efficiencies and use of wood waste for paper production, and the increase has partially offset the generally lower harvests during 1975 to 1992.

As might be expected from the disposition of raw logs, lumber has been the primary forest product from the two states over the period examined, although the proportion of lumber in total products has declined from 89% in 1900 to 53% in 1992. The decrease in fraction of total output has been caused, in part, by the increase in plywood production, which has remained at approximately 20% of total output since 1960. Construction materials have therefore been the major output over the period, ranging from 73% to 89% of total production. Perhaps the largest cause of the decreased importance of lumber has been the increase in paper production

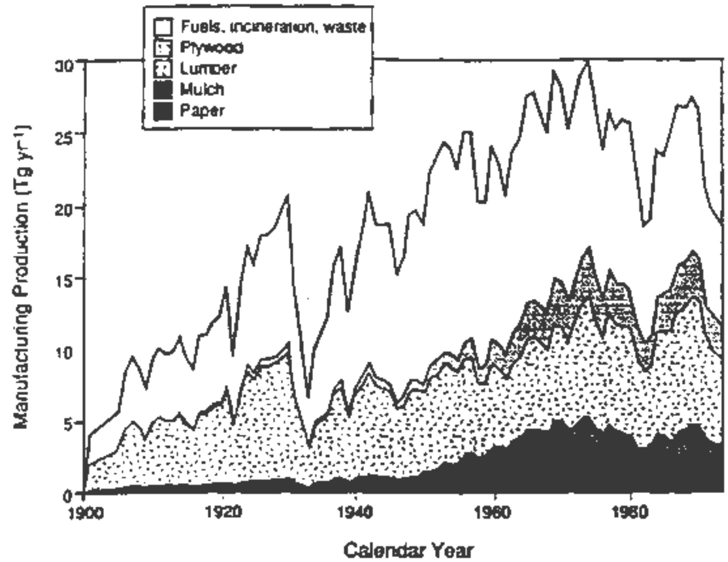

Figure 8. Historical reconstruction of manufacturing production for logs harvested in Oregon and Washington between 1900 and 1992.

since 1950. Before then, paper comprised < 10% of the total product output. The increased use of wood waste for chips after 1950, however, greatly increased paper production to a peak 29% of total production in the 1960's. Since then, paper has been approximately 20% of all product output.

5.7. WASTE DISPOSAL

In the first 70 years of this century, open dumps and incineration were used for most of the wood and paper products disposed (Figure 9). The conversion of open dumps to sanitary landfills appears to have been gradual between 1940 and 1970 and then rapid into the 1980's.

After a long period of decline between 1940 and 1985, during which the fraction of waste incinerated apparently dropped from 35% to 5%, the fraction started to increase, 17% being incinerated in 1991 (Kiser, 1991) and 25% incineration predicted for 1992 (Schmidt, 1990).

Recycling of paper waste in the United States has increased gradually since 1960, when records began to be kept. The percentage of paper and paperboard recycled in the U.S. has steadily risen: 18.1% in 1960 (Liptak, 1991), 20.6% in 1970, 26.7% in 1980, and 38.1% in 1992 (Franklin Associates, 1993). The recycling of wood waste appears to have been minimal until the late 1980's. For example, Portland, OR, has shifted from recycling none of its wood waste in 1985 to 18% in 1992.

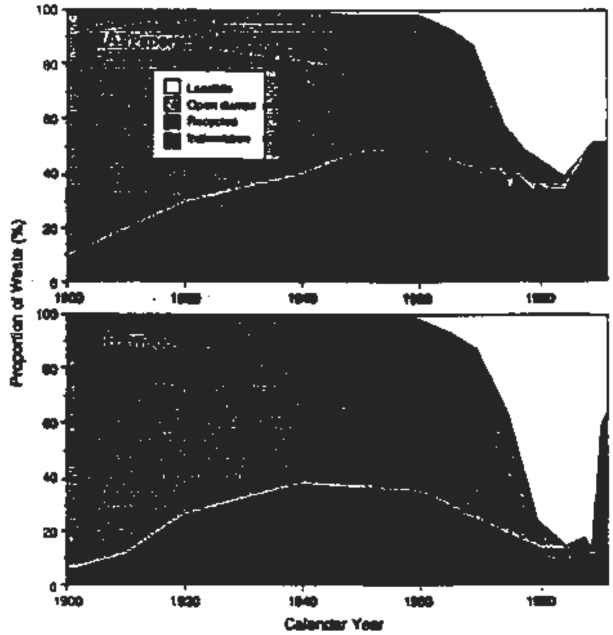

Figure 9. Historical reconstruction of the fate in Oregon and Washington between 1900 and 1992 of (A) paper waste and (B) wood waste.

6. FORPROD Estimates of Carbon Stores, 1900–1992

6.1. RESULTS WITH STANDARD SIMULATION

Of the 1,692 Tg of carbon that has been harvested between 1900 and 1992, the standard simulation indicated that 396 Tg or 23% remains in storage. The largest storage pool has been long-term structures, which, by the end of the period examined, comprised 74% of the total stores (Figure 10). Although landfills rank second, that pool comprised a smaller fraction (20%) than we originally anticipated, probably because landfills have been a major disposal site only for the last two decades. All other pools together contained 6% of the total stores, and some pools, such as paper and mulch, contained less than 1%.

The analysis indicates that, despite nearly a century of timber harvest, few forest product pools have reached a steady state. The overall rate of increase of forest-product carbon stores from 1900 to 1992 was 4.3 Tg year^{-1}. From 1972 to 1992, the rate was 6.02 Tg year^{-1}, indicating that, if anything, the rate of forest-product accumulation is increasing, largely because of the growth of the landfill pool, which had average net accumulations of 0.33 Tg year^{-1} between 1952 and 1972 and 3.46 Tg year^{-1} between 1972 and 1992. In contrast, the net accumulation rate in long-term structures has increased only slightly over those two periods, from 3.2 Tg year^{-1} to 3.65 Tg year^{-1}.

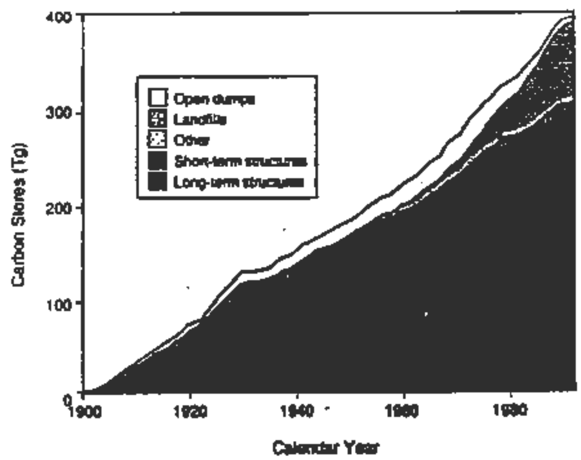

Figure 10. Accumulation of carbon in forest products produced in Oregon and Washington between 1900 and 1992, as estimated by FORPROD.

6.2. EFFECT OF LANDFILL DISPOSAL

Forest-product stores were affected more by the transition from open dumps to landfills than by landfills decomposition-rate constants. Shifting the time of transition forward 5 years and 10 years from the standard scenario gave predictions of 405 and 414 Tg, respectively (Table III), an increase of 2.3% and 4.5%, respectively, over the standard simulation store of 396 Tg. Relative to the cumulative harvest, the discrepancy is even smaller ($< 1\%$), indicating it had little effect on the overall result.

6.3. EFFECT OF LANDFILL DECOMPOSITION

The sensitivity analysis indicated that the landfill decomposition-rate constant, one of the most difficult parameters to estimate, did not greatly influence the overall result (Table III). The rate constants of 0.0025, 0.005, and 0.01 year^{-1} yielded total forest-product stores of 398, 396, and 393 Tg, respectively, as of 1992, a change of $\pm 1\%$ of total stores and $< 0.1\%$ of the cumulative harvest.

6.4. EFFECT OF RECYCLING

Increasing and decreasing the recycling rates had an unanticipated result (Table III). Although doubling the rate increased paper stores from 3.97 Tg to 4.56 Tg, it lowered total stores from 396 to 389 Tg. Halving the recycling rate had the opposite effect, increasing total stores to 400 Tg. Modifying the rate of paper recovery did not modify this trend. This counterintuitive result stems from the fact that paper in landfills lasts much longer than paper as product, so that there is a slight carbon

Table III

Effect of varying selected parameters on estimates of carbon stores in forest products

Test	Total stores	Percent change from standard run
Standard run	396	–
Landfill decomposition		
0.0025 year^{-1}	398	+0.5
0.010 year^{-1}	393	−0.8
Recycling		
Halved	400	+1.0
Doubled	389	−1.8
Landfill transition		
5 years earlier	405	+2.2
10 years earlier	414	+4.5
Long-term structure replacement		
0.005 year^{-1}	422	+6.6
0.02 year^{-1}	357	−9.9

gain without recycling. Relative to the total store, the increase is minor ($< 1\%$) and might be offset by the effects of a reduced demand for virgin fiber.

6.5. EFFECT OF LONG-TERM STRUCTURE REPLACEMENT

The rate constant of long-term structure replacement had the greatest effect on FORPROD simulations (Table III). Decreasing the rate constant from the standard simulation value of 0.01 year^{-1} to 0.005 year^{-1} caused the 1992 total forest-product stores to increase from 396 Tg to 422 Tg, a change of 6.6%. Although this is a large increase in forest-product stores, it only represented a 1.5% increase relative to the cumulative harvest. Likewise, an increase in the replacement rate constant to 0.02 year^{-1} resulted in a decrease in overall stores to 357 Tg, and a 10% decrease of forest-product stores.

6.6. EFFECT OF TEMPORAL VARIATION

The effect of holding efficiency rates constant varied with the data period (Figure 11). When 1970 values were used, overall stores were close to those with the standard simulation, a total of 364 Tg in 92 years. This is an 8% underestimate, probably due to the lower use of landfills in the 70's. Much larger discrepancies were introduced with the 1990 values, which gave a total store of 594 Tg in 92 years, 50% larger than values with the standard simulation.

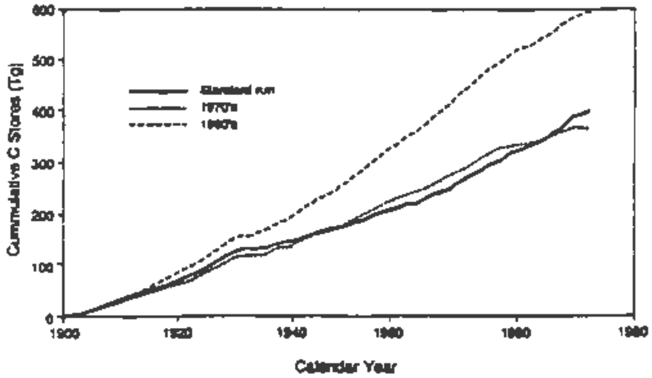

Figure 11. Total accumulation of forest products estimated by FORPROD when parameters varying with time in the standard run (see Figures 4–8) are held constant.

The pools that differed most among these three simulations were long-term structures and landfills. The 1990 rates gave the largest difference among stores in long-term structures: 273, 295, and 345 Tg with 1970, standard, and 1990 values, respectively. The difference for landfill stores was even greater: 43, 78, and 232 Tg with 1970, standard, and 1990 data, respectively. The latter results indicate that while knowing the exact time of transition from open dumps to landfills is not important, modeling with landfills as the sole disposal site is untenable.

In an earlier study, Harmon et al. (1990) estimated that 45% of harvested carbon ends up in long-term storage pools with an average loss of 1.5% year^{-1}. Applying those values here indicates a total carbon store in 1992 of 401 Tg. While total stores over the 92-year period estimated with the two methods are comparable, the earlier study overestimated carbon stores in some years after 1938 by as much as 54 Tg. Although the overall trend is the same, results for a given year may be significantly inaccurate.

7. Discussion and Conclusions

The overall carbon balance of a region depends on net changes in carbon pools such as living vegetation, detritus, soils, and forest products. As our analysis for pools other than forest products is incomplete, it would be misleading to calculate an overall balance. Nonetheless, our analysis of the forest products pools is important for reconstruction of regional carbon flux. It indicates that, despite the large mass of carbon (1,692 Tg) harvested in Oregon and Washington, only a small fraction (23%) is currently stored in forest products. This fraction is probably higher than average for the United States because paper production is more important in other regions (e.g., in the Southeast). The fraction of net stores is probably also high relative to that found in developing regions where manufacturing efficiency is low and decay in use may be greater.

The estimated rate of current accumulation of forest products manufactured from logs harvested in Oregon and Washington was approximately 6 Tg year^{-1}. Our analysis indicates that, far from being in balance, forest products resulting from harvest in the Pacific Northwest will continue to accumulate if harvest levels remain constant. For comparison with other studies of forest products, this absolute accumulation rate can be placed in relative terms with respect to harvest mass and current product stores. Conversion to relative terms indicates a net accumulation of approximately 25% of harvest mass and a growth rate of the forest-products pool of approximately 2% year^{-1}. These values are considerably lower than estimates for the Canadian forest sector (Kurz et al., 1992) of a net accumulation of 50% of harvest mass and a growth rate of the forest-products pool of 4% year^{-1}. The difference may be attributable to the predominance of paper as a product and of landfills as a store in the Canadian forest sector. A study of future forest-product stores from timber harvest in Finland (Karjalainen et al., 1995), in which current harvest levels were extrapolated 50 years into the future, indicated that approximately 38% of the harvest would be in net storage and that forest-products pool would have a relative growth rate of 1.5% year^{-1} from years 30 to 50.

Given the early stage of forest-product modeling, it is difficult to determine whether these differences are due to the dynamics of the systems or to variation in the modeling approaches. The former would be more interesting and meaningful; however, differing assumptions about waste deposition (i.e., transition from open dumps to landfills), landfill decomposition rates, and recycling may obscure real differences in system dynamics. We can distinguish some differences by comparing the manufacturing efficiency estimated by the models. Karjalainen et al. (1995) estimate an overall efficiency of 68%, and Kurz et al. (1992) give individual efficiencies for products that indicate an overall efficiency of 38%, if lumber and paper production (approximately 16.6 Tg) is divided by harvest mass (44 Tg). Our study, which includes past as well as current periods, estimates a quite comparable range of 40% to 61%. It is interesting that the study showing the lowest manufacturing efficiency (Kurz et al., 1992) had the greatest rates of accumulation and net storage. These differences in system response are therefore likely to be caused by treatment of forest products in use rather than in manufacturing.

While it is possible to use average rates of manufacturing efficiency and waste disposition over a given period of interest, our analysis indicates that this method introduces major inaccuracies in temporal patterns of accumulation, particularly when the transition from open-dump to landfill waste-disposal is not included. This may partially explain the large proportion (50%) of forest products found to be stored in landfills by Kurz et al. (1992), who assumed landfills since 1946 were the primary waste deposition site. If one assumes that landfills are not important until 1970 (as in the United States), then they would store about half the value estimated by Kurz et al. (1992) and comprise 25% of the total stores. Our sensitivity analysis of wastes generated from Oregon and Washington wood products indicate a similar effect. The assumption that landfills were the primary deposition site increased the share of forest products stored landfills from 20% in the standard simulation to 39%.

In contrast to its sensitivity to the transition from open dumps to landfills, the FORPROD model was relatively insensitive to the rates of recycling, landfill decomposition, and long-term structure replacement. Doubling and halving those parameters led to less than 10% change in total stores of forest products. Karjalainen et al. (1995) performed a sensitivity analysis similar to ours by altering product life-spans, recycling rates, and landfill decomposition rates. They found that changing the product life-span 10% resulted in $< 3\%$ change in total stores. Similarly they found that increasing recycling 50% increased stores $< 2\%$, and that doubling the flow to landfills from 25% to 50% of all waste increased total stores 10%. The largest change resulted from increasing the landfill decomposition rate from 1% to 10% year^{-1}, which decreased stores 20%.

The insensitivity of forest-products models to most parameters may be due to the fact that substantial amounts of carbon are lost to the atmosphere during manufacturing, when approximately 40% to 60% is lost within a few years of harvest, leaving a relatively small fraction to be stored for a long period. These models may also be insensitive to these parameters because they generally involve internal transfers to pools that sequester carbon. The sensitivity of the models to the manufacturing parameters is fortuitous, because those parameters have the best historical data. In contrast, the fate of paper and wood wastes appears to be a key focus for future research. Once the uncertainty regarding paper and wood waste is resolved, the role of forest products in the overall global carbon balance can be assessed.

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The outcome is in the assumptions: analyzing the effects on atmospheric CO_2 levels of increased use of bioenergy from forest biomass

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Abstract

Recently, several studies have quantified the effects on atmospheric CO_2 concentration of an increased harvest level in forests. Although these studies agreed in their estimates of forest productivity, their conclusions were contradictory. This study tested the effect of four assumptions by which those papers differed. These assumptions regard (1) whether a single or a set of repeated harvests were considered, (2) at what stage in stand growth harvest takes place, (3) how the baseline is constructed, and (4) whether a carbon-cycle model is applied. A main finding was that current and future increase in the use of bioenergy should be studied considering a series of repeated harvests. Moreover, the time of harvest should be determined based on economical principles, thus taking place before stand growth culminates, which has implications for the design of the baseline scenario. When the most realistic assumptions are used and a carbon-cycle model is applied, an increased harvest level in forests leads to a permanent increase in atmospheric CO_2 concentration.

Keywords: atmosphere, bioenergy, carbon, climate change, Faustmann, impulse response functions

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Introduction

The literature draws attention to the fact that the conversion of natural habitats to cropland leads to release of carbon, thus creating a biofuel carbon debt with a potential payback period of several decades or even centuries (see, for example, Gurgel *et al.*, 2007; Fargione *et al.*, 2008; Searchinger *et al.*, 2008; Melillo *et al.*, 2009; Gibbs *et al.*, 2010; Lapola *et al.*, 2010).

The articles mentioned, however, studied biofuels based on fast-growing crops, in which the biomass harvested within 1 year is replaced by a new crop. In that case, the CO_2 released by combustion of the biomass could, for practical purposes, be ignored because the growth of the new crop requires the capture of the same amount of CO_2 within 1 year.

The issue becomes more complex if the source of bioenergy is a forest. The rotation period of a boreal forest stand is usually 70–120 years. Hence, a century might be required for the regrowth of a harvested boreal forest stand and recapture of the amount of CO_2 released originally. Despite this considerable time lag, recent studies have considered wood fuels from boreal forests as being carbon neutral, thus ignoring the amount of CO_2 released by the combustion of that wood (see, for example, Bright & Strømman, 2009; Sjølie *et al.*, 2010).

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Keeping in mind that the carbon intensity of wood fuels is approximately at the level of coal, it is obvious that, from a methodological perspective, ignoring these emissions is not satisfactory. A body of literature has thus emerged that accounts for the amount of CO_2 released from combustion of biomass from forests and other slow-growing sources of biomass (see, for example, Manomet Center for Conservation Sciences, 2010; Cherubini *et al.*, 2011a,b; McKechnie *et al.*, 2011; Holtsmark, 2012).[1]

The conclusions of the articles mentioned vary significantly. For example, Holtsmark (2012) found that increasing the harvest of a forest permanently lowered the carbon stock of the forest and, consequently, permanently heightened the amount of CO_2 in the atmosphere. In contrast, Cherubini *et al.* (2011a,b) found that the CO_2 concentration in the atmosphere was *lower* 60–70 years after harvesting a relatively slow-growing forest than if the forest had not been harvested. Figure 1 illustrates these differences. The dashed line (left axis) depicts the atmospheric CO_2 that remains after harvest and combustion of a stock of biomass containing one metric ton of carbon, as found by Cherubini *et al.* (2011a). The solid line (right axis) shows the corresponding result in the work of Holtsmark (2012), in which increased harvest levels were predicted to

[1]Haberl *et al.* (2012a,b) and Schulze *et al.* (2012) include further references and discuss the implications of this literature.

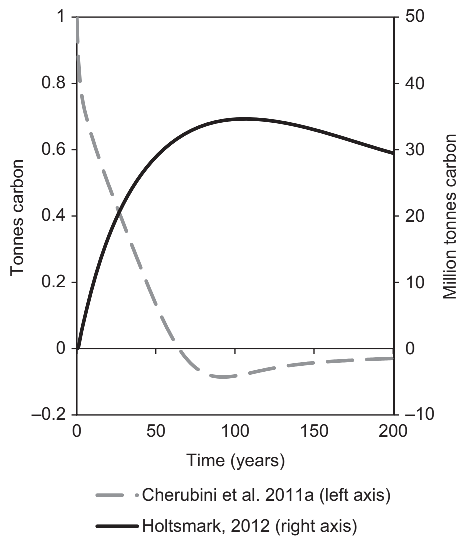

Fig. 1 The dashed line (left axis) shows the atmospheric carbon that remains at time *t* after a single harvest event at time *t* = 0, according to Cherubini *et al.* (2011a). The solid line (right axis) shows the atmospheric carbon that remains after a series of subsequent harvest events as a result of the application of an impulse response function to the results of Holtsmark (2012).

increase the amount of CO_2 in the atmosphere in the long term.[2]

The different conclusions reached in these papers are explained by different methodological choices or assumptions. Therefore, an analysis of the importance of different simplifications and methodological choices is needed. Here, I will focus on four methodological choices.

1 Some studies consider a single harvest event occurring at the present time, with no biomass to be harvested in the future. However, a single harvest event performed at the present time will not produce any biomass in the future and is, therefore, not satisfactory if one wants to gather knowledge related to the consequences of the increased use of biomass presently *and* in the future. A single harvest event performed at the present time will not produce the required biomass if one aims to replace fossil fuels with biomass on a permanent basis. I will, therefore, demonstrate the effects of the replacement of a single harvest approach with a permanently increased harvest approach.
2 In some studies, it is assumed that a rotation period ends when the growth of the trees has culminated. Other studies take into account that, since the publication of the work of Faustmann (1849), and even earlier,[3], forest economists have known that a commercial forester will not postpone harvest until the growth of the trees has culminated, but will usually harvest at an earlier stage, following the so-called Faustmann rule. I will demonstrate the effects of the application of a rotation-period length that is in accordance with this rule.
3 Taking into account that harvest usually takes place in stands that are still growing, the baseline scenario becomes important. Not all studies take into account that the harvest scenario should be measured against a baseline scenario (with no harvest) in which the trees are still growing, thus capturing CO_2 from the atmosphere. I will demonstrate the importance of the use of a realistic baseline scenario along these lines.
4 In some studies, it is assumed, for simplicity, that the CO_2 released from the combustion of biomass accumulates and remains in the atmosphere forever. In other studies, an impulse response function is applied that models the ability of the ocean and of the terrestrial biosphere to absorb CO_2 from the atmosphere.

Table 1 provides an overview of how the five studies on the bioenergy from forests mentioned deal with these methodological choices. The approach of Cherubini *et al.* (2011a,b) was the inclusion of an impulse response function in the analysis, whereas the other studies listed applied a simple accumulation of CO_2. However, Cherubini *et al.* (2011a,b) and Manomet Center for Conservation Sciences (2010) considered a single harvest event exclusively. The methodology used for the construction of the baseline scenarios also varied.

To demonstrate quantitatively how the methodological choices influence the conclusions of this type of study, I will use the articles of Cherubini *et al.* (2011a) and Holtsmark (2012) as the starting point, adjust their methodological choices, and demonstrate the consequences of these adjustments. In contrast with the approach of Cherubini *et al.* (2011a), Holtsmark (2012) considered the consequences of permanently increasing harvest levels by studying a series of harvests. Moreover, Holtsmark (2012) took into account that the harvest usually takes place before the growth of the stand culminates and how the baseline scenario then should be designed. Holtsmark (2012), however, ignored the decay functions of atmospheric CO_2 and considered, for simplicity, accumulated emissions exclusively.

[2]See the red curve in Fig. 4, page 423, in Holtsmark (2012). To achieve the somewhat different solid line in Fig. 1 here, the impulse response function of the Bern 2.5CC carbon-cycle model was applied; see Eqn (1).

[3]See the discussion of early contributions to this issue in Samuelson (1976) and Scorgie & Kennedy (1996).

Table 1 Methodological differences in five recent papers dealing with bioenergy from forest biomass

	Cherubini *et al.* (2011a)	Cherubini *et al.* (2011b)	Manomet Center for Conservation Sciences (2010)	McKechnie *et al.* (2011)	Holtsmark (2012)
Single harvest event or permanently higher harvest level?	Single	Single	Single	Permanent	Permanent
Does the no harvest baseline take growth and carbon capture in mature stands into account?	No	No	Yes	Yes	Yes
Is the time of harvest in accordance with the Faustmann rule?	No	Some of the scenarios	Yes	Yes	Yes
Impulse response function (IRF) or simple accumulation of CO_2?	IRF	IRF	Simple accumulation	Simple accumulation	Simple accumulation

This study builds a bridge between the approaches of these two studies by taking atmospheric decay functions into account, as in Cherubini *et al.* (2011a), and including the realistic baseline scenario and the multiple harvest approach of Holtsmark (2012) in the analysis.

This paper is organized as follows. The model and the basic methodological choices are presented in the next section, the results are presented in the third section, and the results are discussed in the fourth section, which also includes the conclusions of the study.

Materials and methods

Based on Forster *et al.* (2007) and the Bern 2.5CC carbon-cycle model, which those authors recommend, Cherubini *et al.* (2011a) applied the following atmospheric CO_2 decay function:

$$y(t) = \Delta_0 + \sum_{i=1}^{3} \Delta_i e^{(-t/\alpha_i)}, \tag{1}$$

where $y(t)$ represents the fraction of an initial pulse of CO_2 at time $t = 0$ that remains in the atmosphere at time t and where α and Δ_i are parameters (Table 2). The time unit is 1 year. The decay is caused by the uptake of CO_2 by the ocean and by the terrestrial biosphere. Cherubini *et al.* (2011a) considered two cases. In the first case, those authors did not take into account the oceanic absorption of anthropogenic CO_2 from the atmosphere, although they considered this effect in the second case. For the purpose of this study, only the latter case is considered, as it is the most realistic and, therefore, the most interesting case.

It is assumed that the harvesting of biomass from forests is followed by replanting and the growth of new biomass. Regrowth implies carbon capture from the atmosphere. Cherubini *et al.* (2011a) assumed that the growth and carbon capture of the stand after a harvest follow the analytic form:

$$g(\tau) = \left(2\pi\sigma^2\right)^{1/2} e^{-(\tau-\mu)^2/2\sigma^2}, \tag{2}$$

Table 2 Parameter values

			Cherubini *et al.* (their case with r = 100)	Present case
Δ_0	0.217	σ	25	37.5
Δ_1	0.259	μ	50	75
Δ_2	0.338			
Δ_3	0.186			
α_1	172.9			
α_2	18.51			
α_3	1.186			

where σ and μ are parameters and τ is the age of the stand. It can be deduced that a parcel with a stand age τ has the following carbon stock.[4]

$$C(\tau) = \left(2\pi\sigma^2\right)^{1/2} \sum_{\tau'=0}^{\tau} e^{-(\tau'-\mu)^2/2\sigma^2}. \tag{3}$$

The carbon captured by biomass regrowth should be considered in terms of negative emissions. Negative emissions should be treated symmetrically regarding positive emissions. Thus, the decay function presented in (1) should be applied to these negative emissions exactly as it is applied to the positive emissions.

Consider, for example, a parcel replanted at time $t = 0$. The carbon captured at time t_1 would be $g(t_1)$, and at time t_2, i.e., t_2-t_1 periods later, a fraction $y(t_2-t_1)$ of these negative emissions, i.e., $-g(t_1)\cdot y(t_2-t_1)$, is remaining in the atmosphere.

Assume now that, at time $t = 0$, the age of the stand is τ_m and that harvesting proceeds at this time. Combustion of the extracted biomass causes a CO_2 emission pulse $C(\tau_m)$, which, for simplicity, is labeled as C in the following equation. Taking the regrowth function described in (2) into account, the amount of CO_2 in the atmosphere A_H (t) at time t, will be as follows:

[4] To show exactly how the numerical examples in the next section are constructed, I used discrete time in the theoretical model description as well.

$$A_H(t) = C \cdot y(t) - \sum_{t'=0}^{t} g(t')y(t-t'), \tag{4}$$

where the first term on the right-hand side represents what is left of the pulse in the atmosphere at t periods after harvesting, whereas the second term represents the effect of regrowth.

Thus far, I have followed the example of Cherubini *et al.* (2011a). However, the alternative to harvesting and combustion of biomass is to not harvest: i.e., letting the stand grow and capture more CO_2. In this case, the amount of CO_2 in the atmosphere would evolve as follows.

$$A_{\mathrm{NH}}(t) = -\sum_{t'=0}^{t} \mathrm{g}(\tau_{\mathrm{m}} + t')\mathrm{y}(t-t'). \tag{5}$$

Note the assumption of Cherubini *et al.* (2011a) that harvesting always takes place when the growth of the stand has culminated [see (c) in Fig. 2], which is the reason why those authors disregarded this effect. If we take this effect into account, the net effect of harvesting on the atmospheric carbon content will be as follows:

$$A_{\mathrm{S}}(t) = A_{\mathrm{H}}(t) - A_{\mathrm{NH}}(t). \tag{6}$$

The time at which harvesting takes place is a pertinent point. If we assume that the stock of trunks in the stand is proportional to the amount of biomass $C(t)$ and that the market interest rate is r, then, according to the Faustmann rule, a forest owner will harvest when the stand age τ satisfies the following equation.

$$\frac{C'(\tau)}{C(\tau)} = \frac{r}{1-e^{-r\tau}}. \tag{7}$$

As the interest rate approaches zero, (7) is reduced to

$$\frac{C'(\tau)}{C(\tau)} = \frac{1}{\tau}. \tag{8}$$

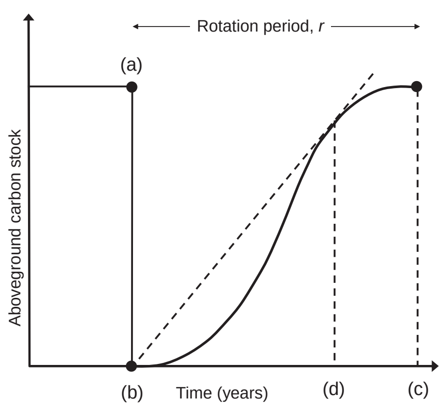

Fig. 2 This diagram is identical to Fig. 1 in Cherubini *et al.* (2011a), with the exception of the addition of the dashed lines. Cherubini *et al.* (2011a) assumed that harvest takes place at (c), whereas the Faustmann rule says that harvest usually will take place somewhere between (b) and (d).

Harvesting at a time at which τ satisfies (8) implies a maximum sustained yield (MSY) and harvesting at point (d) in Fig. 2. To the extent that the forest owner discounts future income, the rotation period will be shorter.

The intuition behind the Faustmann rule is as follows. The forest owner takes into consideration his opportunity to invest the harvest profit, creating postharvest periodic revenue of $rC(\tau)$. Postponing the harvest has an alternative cost corresponding to this revenue. This could easily be interpreted as that harvest should take place when τ satisfies the equation $C(\tau) = rC'(\tau)$. However, the Faustmann rule (7) also takes into account that, if the first harvest is postponed, all future harvests must also be postponed. This leads to Eqn (7), which implies an even earlier harvest than is indicated by the more simple equation $C(\tau) = rC'(\tau)$.

The application of the limiting case of the Faustmann rule described in (8) to the slower growing forest studied by Cherubini *et al.* (2011a), i.e., a forest with a rotation span of 100 years, implies that harvesting occurs when the stand is 70 years old. In other words, the slower growing forest considered by Cherubini *et al.* (2011a) is actually a relatively rapidly growing boreal forest. The rotation period for MSY in most Scandinavian forests is reportedly 70–120 years.

I shall, therefore, adjust the parametric assumptions to allow for a MSY rotation period of 100 years for the stand in question. I will accomplish this using the parameters $\sigma = 37.5$ and $\mu = 75$ (Table 2). Given these assumptions, the growth and carbon capture of the stand will culminate at a stand age of approximately 150 years. In other words, the stand will continue to grow and capture CO_2 from the atmosphere, as specified in Eqn (5), if it is not harvested after reaching maturity. The two compared (re)growth scenarios are shown in Fig. 3. The solid line traces the carbon stock of the stand if it is harvested at time $t = 0$, whereas the dashed line traces the carbon stock of the stand if its age is 100 years at time $t = 0$ and if it is not harvested.

Results

Single harvest event

First, consider the case studied by Cherubini *et al.* (2011a), with a rotation period of 100 years. The harvest gives rise to a pulse emission of one metric ton of carbon at time 0, which is recaptured completely by the regrowth of the stand over the next 100 years. After these 100 years, there is no further growth on the stand. The dashed line in Fig. 4 shows the atmospheric carbon remaining from this pulse, according to the calculations of those authors. Note that, after ca. 65 years, a lower carbon concentration in the atmosphere is estimated in the presence of a harvest event compared with the case without harvest. This is so because increased atmospheric CO_2 levels lead to an increase in the accumulation of carbon in the terrestrial ecosystems, as well as to an increase in oceanic CO_2 absorption.

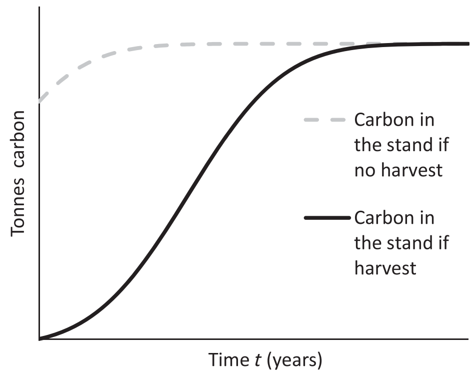

Fig. 3 Development of the carbon stock of a stand that is mature at time 0. The solid line represents the harvest case. The dashed line represents the no-harvest case.

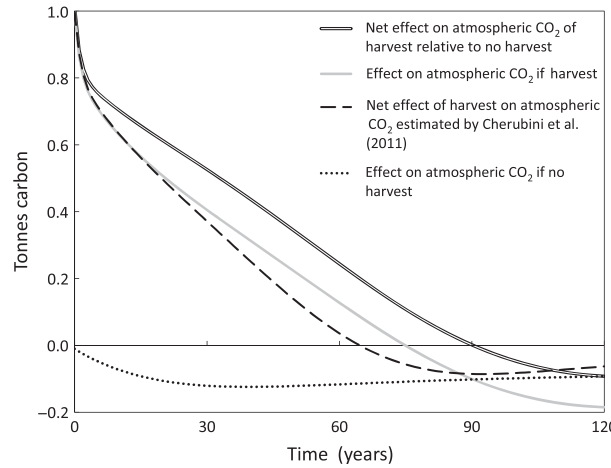

Fig. 4 The dashed line depicts the remaining atmospheric carbon for the methodology applied by Cherubini *et al.* (2011a), with a rotation period of 100 years. The gray, solid line represents the atmospheric carbon remaining with a slower growing stand with harvesting occurring at a stand age of 100 years. In both cases, harvesting of this stand at time 0 is assumed to cause an emission pulse of one ton of carbon. The dotted curve traces the effect on atmospheric carbon levels in the no-harvest case, whereas the double-line curve shows the net effect of harvest compared with no harvest.

As argued in the previous section, when dealing with a boreal forest, it would be appropriate to consider a MSY rotation period of 100 years and culmination of growth after approximately 150 years, which would be consistent both with the Faustmann rule and with a typical boreal forest stand. The harvest of this forest stand at time 0 is assumed to lead to a pulse of emission of one ton of carbon. The gray, solid line in Fig. 4 shows the level of atmospheric carbon from the pulse that remains in this case; cf. Eqn (4).

The question of the use of an appropriate baseline arises at this point. As Cherubini *et al.* (2011a) assumed that there is no further growth on the stand in the no-harvest case, there is no change in atmospheric carbon in their baseline scenario. The scenario is different if it is assumed that there is continued growth in the no-harvest case. The dotted curve in Fig. 4 traces the effect on atmospheric CO_2 levels in the no-harvest case and corresponds to Eqn (5). This curve dips below zero because there is no emission pulse at time $t = 0$, although carbon is still captured by continued growth after this time point.

Our interest is related to the *net* effect of harvesting on atmospheric CO_2 levels. This can be computed by subtracting the amount of atmospheric carbon in the no-harvest case from the amount of atmospheric carbon in the case with harvest; cf. Eqn (6). The result is the double-line curve in Fig. 4. Compared with the case studied by Cherubini *et al.* (2011a), this case gives a somewhat longer period of enhanced levels of atmospheric CO_2.

Multiple harvest events

The numerical examples presented in the previous section measure the effect of a *single harvest event*. However, IPCC documents, such as Chum *et al.* (2012), envisage a permanent increase in the use of bioenergy and, accordingly, a higher harvest rate. Therefore, in the following paragraphs, I will consider a case in which the harvest events described in the previous section take place every year on a permanent basis.

Consider now a forest with an age structure such that every year one parcel, each with a growth function described by Eqns (1) and (2), reaches the stand age τ_m and is, therefore, considered mature and ready for harvest. The net effect on atmospheric carbon of harvesting a stand every year compared with the case where the parcels are left unharvested, is given by the following equation.

$$A(t) = \sum_{t'=0}^{t} A_S(t'). \tag{9}$$

The function $A_S(t)$ is defined in Eqn (6). Given the numerical assumptions, the expression is shown by the solid line depicted in Fig. 5. Other than the difference in scale (million tons and tons of carbon), the solid line shown in Fig. 5 is not far off the corresponding result that is obtained when the impulse response function is applied to the data of Holtsmark (2012), which is indicated by the dotted curve shown in Fig. 5.

To have intuition to the above described results, study the dashed curve shown in Fig. 5, which is identical to the double lined curve depicted in Fig. 4. These curves show that the effect of a single harvest on atmospheric CO_2 levels is a two-stage process. During the first stage, the level of atmospheric CO_2 is higher than it would have been in the absence of harvest, whereas the reverse is true in the second stage. The observation

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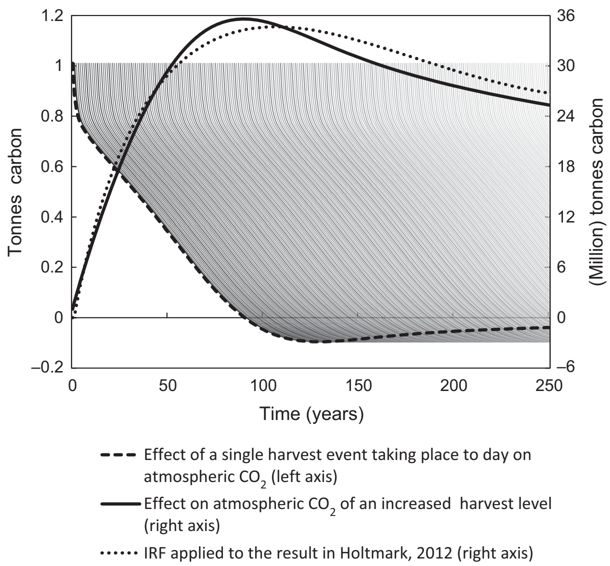

Fig. 5 The dashed curve (left axis) shows the net effect on atmospheric carbon of a single harvest event taking place today compared with the no-harvest case. The set of thin curves depicts similar net effects of subsequent annual harvest events. The thick solid line (right axis) shows the total net atmospheric carbon that remains after this series of identical annual harvest events. The dotted curve (right axis) represents the effect of an increased harvest level, as described in Holtsmark (2012).

that the negative effect in the second stage is smaller than the positive effect during the first stage is important to predict the outcome of a series of harvest events.

Next, consider the case in which harvest takes place annually. Every year, there is a pulse of emissions of 1 ton of carbon with subsequent regrowth on the stand. The set of thin curves shown in Fig. 5 represent the effects of these subsequent annual harvest events. The net effect on atmospheric CO_2 of this series of harvest events is calculated via vertical summation of this set of curves and the dashed curve. This gives the solid line depicted in Fig. 5, which is measured on the right axis.

Note that the dashed curve converges toward zero, whereas the solid line converges toward 19 tons of carbon (result not shown here). Hence, a single harvest event has no long-term effect on atmospheric carbon, whereas a permanently increased harvest level will increase atmospheric CO_2 permanently. It follows that an increased harvest level is not a carbon-neutral activity not even in the long term, whereas a single harvest event is a carbon-neutral activity in the long term.

Discussion

The realization that wood fuels are not carbon neutral gives rise to a number of methodological questions or assumptions regarding the manner via which CO_2 emissions from wood fuels should be modeled. In this study, I have focused on four methodological choices. First, I analyzed whether the consideration of a single harvest event is sufficient when the consequences of the increased use of biomass presently and in the future are to be analyzed. Second, I analyzed whether the assumption that the rotation period ends when the growth of the trees has culminated is satisfactory. Third, I analyzed the manner via which the baseline no-harvest scenario should be constructed. Finally, I studied the importance of including impulse response functions in the analyses.

The work of Cherubini *et al.* (2011a) was used as a starting point to evaluate the importance of these methodological choices. The approach of those authors of using an impulse response function was adopted. However, their model was adjusted taking into account that harvest usually takes place before the growth of the trees has culminated. The baseline (no harvest) scenario was adjusted accordingly. Finally, a single harvest approach was supplemented with a multiharvest approach, which reflects the fact that the policy proposal to be analyzed addresses the question of whether biomass should be harvested at the current time *and* in the future.

The numerical simulations provided information on the importance of these methodological choices. First, they showed that the results change fundamentally when a single harvest approach is replaced with a multiharvest approach reflecting a permanently increased harvest level. A single harvest approach could lead to the conclusion that wood fuels are carbon neutral in the long term, but not in the short term, whereas a multiharvest approach leads to the conclusion that wood fuels are not carbon neutral, neither in the long term nor in the short term. The multiharvest approach revealed that a permanently increased harvest level leads to a permanent increase in atmospheric carbon also when a realistic carbon-cycle model is taken into account.

Second, it was found that the consideration that harvest usually takes place before growth of the trees has culminated and the consequent adjustment of the baseline have a significant effect on the results, although they are not changed fundamentally.

Third, the results of Holtsmark (2012) were adjusted by incorporating an impulse response function in the analyses. This approach did not change the results fundamentally. Using simple accumulation of CO_2 in the atmosphere in this type of study is an approximation that is acceptable.

Another question, which was not discussed here, concerns the extent to which the increased harvest of a forest may reduce atmospheric carbon if the extracted biomass

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replaces fossil energy sources. For a discussion of this question, see Holtsmark (2012) and McKechnie *et al.* (2011).

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LETTERS

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Regional carbon dioxide implications of forest bioenergy production

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Strategies for reducing carbon dioxide emissions include substitution of fossil fuel with bioenergy from forests[1], where carbon emitted is expected to be recaptured in the growth of new biomass to achieve zero net emissions[2], and forest thinning to reduce wildfire emissions[3]. Here, we use forest inventory data to show that fire prevention measures and large-scale bioenergy harvest in US West Coast forests lead to 2–14% (46–405 Tg C) higher emissions compared with current management practices over the next 20 years. We studied 80 forest types in 19 ecoregions, and found that the current carbon sink in 16 of these ecoregions is sufficiently strong that it cannot be matched or exceeded through substitution of fossil fuels by forest bioenergy. If the sink in these ecoregions weakens below its current level by 30–60 g C m^{-2} yr^{-1} owing to insect infestations, increased fire emissions or reduced primary production, management schemes including bioenergy production may succeed in jointly reducing fire risk and carbon emissions. In the remaining three ecoregions, immediate implementation of fire prevention and biofuel policies may yield net emission savings. Hence, forest policy should consider current forest carbon balance, local forest conditions and ecosystem sustainability in establishing how to decrease emissions.

Policies are being developed worldwide to increase bioenergy production as a substitution for fossil fuel to mitigate fossil fuel-derived carbon dioxide emissions, the main cause of anthropogenic global climate change[4,5]. However, the capacity for forest sector bioenergy production to offset carbon dioxide emissions is limited by fossil fuel emissions from this activity (harvest, transport, and manufacturing of wood products) and the lower energy output per unit carbon emitted compared with fossil fuels[6]. Furthermore, forest carbon sequestration can take from decades to centuries to return to pre-harvest levels, depending on the initial conditions and amount of wood removed[7]. The effects of changes in management on CO_2 emissions need to be evaluated against this baseline. Consequently, energy policy implemented without full carbon accounting and an understanding of the underlying processes risks increasing rather than decreasing emissions[4,8].

In North America, there is increasing interest in partially meeting energy demands through large-scale forest thinning[5], with the added benefit of preventing catastrophic wildfire and concurrent carbon loss[3]. Although forest thinning can be economically feasible, sustainable, and an effective strategy for preventing wildfire where risk is high[9,10], it remains unresolved whether this type of forest treatment can satisfy both the aims of preventing wildfire and reducing regional greenhouse gas emissions.

For both aims to be satisfied, it needs to be shown that: (1) reduction in carbon stocks due to thinning and the associated emissions are offset by avoiding fire emissions and substituting fossil fuel emissions with forest bioenergy, (2) the change in management results in less CO_2 emissions than the current or 'baseline' emissions, and (3) short-term emission changes are sustained in the long term. Determination of baseline forest sector carbon emissions can be accomplished by combining forest inventory data and life-cycle assessment (LCA[6]) that includes full carbon accounting of net biome production (NBP) on the land in addition to carbon emissions from bioenergy production and storage in wood products. NBP is the annual net change of land-based forest carbon after accounting for harvest removals and fire emissions.

Our study focused on the US West Coast (Washington, Oregon and California), a diverse region owing to the strong climatic gradient from the coast inland (300–2,500 mm precipitation per year) and a total of 80 associated forest types, ranging from temperate rainforests to semi-arid woodlands (Supplementary Table S1). The region is divided into 19 distinct ecoregions[11] on the basis of climate, soil and species characteristics, and includes a broad range of productivity, age structures, fire regimes and topography. Mean net primary production of the forest types range from 100–900 g C m^{-2} yr^{-1} (this study), falling within the global range of 100 to 1,600 g C m^{-2} yr^{-1} reported for temperate and boreal forests[12]. Forest land ownership is divided fairly evenly between public and private sectors having different management histories and objectives that affect forest carbon dynamics[13].

Carbon sequestration rates vary greatly across the region, with mean net ecosystem production (NEP; photosynthesis minus respiration) ranging from -85 g C m^{-2} yr^{-1} in the dry Northern Basin to more than 400 g C m^{-2} yr^{-1} in the mesic Coast Range. After accounting for fire emissions and substantial harvest removals, regional NBP remains a significant sink of 26 ± 3 Tg C yr^{-1} or 76 ± 9 g C m^2 yr^{-1}, similar to the US average[14] and estimates for the member states of the European Union[15]. Sixteen of the 19 ecoregions, representing 98% of the forest area in the region are estimated to be carbon sinks (Fig. 1a; exceptions are drier ecoregions where annual productivity is low and fire emissions are relatively high). Thus, the observed regional sink is not solely due to the region's highly productive rainforests, which occupy 15% of the area. Within the region, California's NBP is higher than that of Oregon and Washington (107 versus 53–61 g C m^{-2} yr^{-1}), primarily owing to differences in NEP (Supplementary Table S2) and harvest between similar forest types within the same ecoregions that cross state boundaries (Supplementary Discussion and Table S3).

In addition to current management or business as usual (BAU, characterized by current preventive thinning and harvest levels), we designed three treatments (Supplementary Fig. S1a) to reflect the varying objectives of potential forest management systems: forest fire prevention by emphasizing removal of fuel ladders

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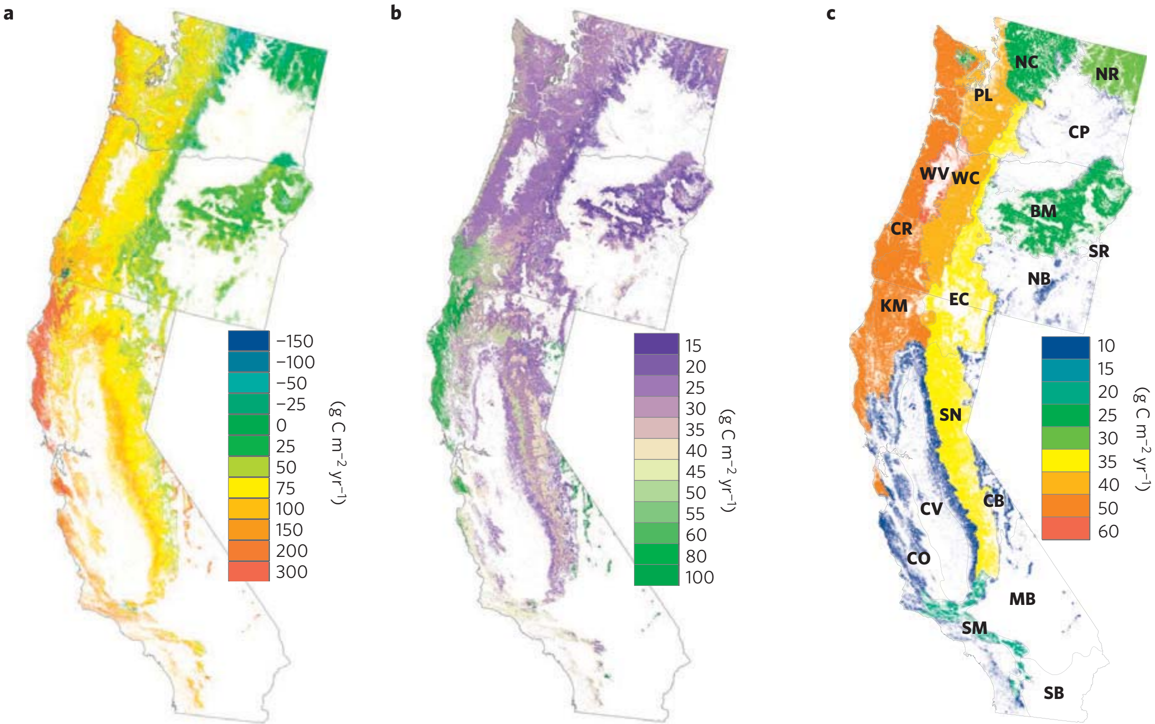

Figure 1 | Maps of US West Coast NBP and uncertainty for current and threshold conditions. a, Current NBP or BAU; positive values (warm colours) indicate forest sinks whereas negative values (cool colours) are carbon sources to the atmosphere. **b**, The current NBP uncertainty estimates that were calculated using Monte Carlo simulations of mean forest type values for the components of NBP (net ecosystem productivity, fire and harvest) combined with the uncertainty associated with remote sensing land cover estimates. **c**, The amount NBP would need to decrease to reach a threshold NBP where bioenergy management may result in emission decreases to the atmosphere. BM, Blue Mountains; CB, Central Basin; CO, California Chaparral and Oak Woodlands; CP, Columbia Plateau; CR, Coast Range; CV, Central California Valley; EC, East Cascades; KM, Klamath Mountains; MB, Mohave Basin; NB, North Basin and Range; NC, North Cascades; NR, Northern Rockies; PL, Puget Lowlands; SB, Sonoran Basin; SM, Southern California Mountains; SN, Sierra Nevada; SR, Snake River; WC, West Cascades; WV, Willamette Valley.

('fire prevention') in fire-prone areas, making fuel ladder removal economically feasible by emphasizing removal of additional marketable wood in fire-prone areas ('economically feasible'), or thinning all forestland regardless of fire risk to support energy production while contributing to fire prevention ('bioenergy production'). Removals are in addition to current harvest levels and are performed over a 20-year period such that 5% of the landscape is treated each year. Our reliance on a data-driven approach versus model simulations strengthens our analysis in the short term, but limits our ability to make long-term predictions. Extending our study beyond a 20-year timeframe would overstretch data use because current forest growth is unlikely to represent future growth due to changes in climate, climate-related disturbance, and land use[16,17].

In our study region, we found that thinning reduced NBP under all three treatment scenarios for 13 of the 19 ecoregions, representing 90% of the region's forest area. The exceptions where NBP was not reduced were primarily due to high initial fire emissions compared to NEP (for example, Northern Basin and North Cascades; Supplementary Fig. S2). The dominant trend at the ecoregion level was mirrored at the regional level, with the bioenergy production scenario (highest thinning level) resulting in the region becoming a net carbon source (Supplementary Table S2 and discussion of state-level estimates). Regionally, forest biomass removals exceeded the potential losses from forest fires, reducing the *in situ* forest carbon sink even after accounting for regrowth, as found in previous studies with different approaches or areas of inference[8,18]. Because we have assumed high reductions in fire emissions for the areas treated in each scenario, it is unlikely we are underestimating the benefit of preventive thinning on NBP.

It is important to recognize that even if the land-based flux is positive (a source) or zero (carbon neutral), decreases in NBP from BAU can increase CO_2 emissions to the atmosphere. LCA was used to estimate the net emissions of carbon to the atmosphere in each treatment scenario (Supplementary Fig. S1b and Tables S4 and S5). LCA at the ecoregion level revealed that emissions are increased for 10 out of 19 of the ecoregions (Fig. 2), representing 80% of the forest area in the region. The combination of *in situ* and wood-use carbon sinks and sources emit an additional 46, 181 and 405 Tg C to the atmosphere over a 20-year period (2–14% increase) above that of the BAU forest management scenarios for the fire prevention, economically feasible, and bioenergy production treatments, respectively (Fig. 3).

Sensitivity analysis of our results to a range of fire emission reductions, energy conversion efficiencies, wood product decomposition rates and inclusion of wood substitution showed that carbon emissions varied by −10 to 28% from the optimum values across the scenarios, depending on the combination of assumptions (Supplementary Discussion and Table S6). The analysis revealed that an increase in estimated current fire emissions (which effectively reduces the baseline sink) may decrease total atmospheric C emissions in the fire prevention scenario, but only given optimum conditions for all of the other parameters (for example 100% energy efficiency). Nevertheless, if fire frequency and intensity increase in the future[19], emissions savings through forest bioenergy production may become possible, especially in ecoregions where the sink is already weak.

Previous case studies showed that harvesting an old-growth forest in the Pacific Northwest[20] or increasing the thinning removals of temperate forests is likely to deteriorate the forest and wood

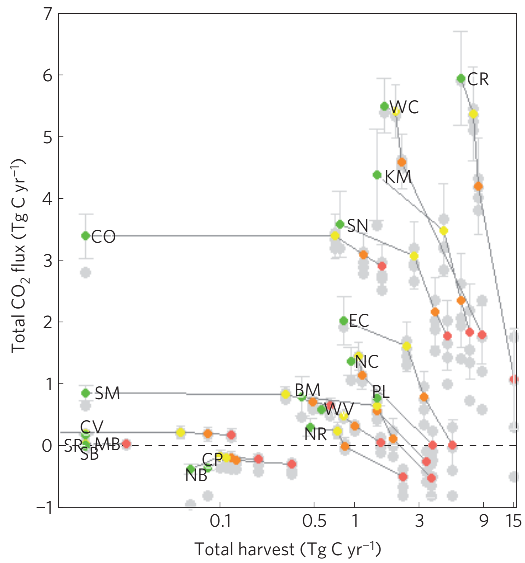

Figure 2 | Life-cycle assessment carbon emission trends by ecoregion under various management scenarios. The *x* axis is the total harvest (BAU+treatment) and the *y* axis is the total CO_2 flux in Tg C yr^{-1} for each ecoregion. Coloured circles represent each scenario (green, BAU; yellow, fire prevention; orange, economically feasible; red, bioenergy production). Grey circles are the values for each sensitivity analysis set of parameters and the error bars represent the estimate uncertainty. The locations of the ecoregions indicated by labels are shown in Fig. 1a. For most ecoregions, the treatments increase emissions to the atmosphere.

product carbon stock[21]. However, these studies were limited to a handful of sites, relied primarily on modelled results[3,18] and did not account for the energy requirements of forest management and wood processing nor for the potential to substitute fossil fuels with bioenergy. We build on these results by including all ecoregions, all age classes (not just old-growth), three treatments including bioenergy production, and sector-based LCA. We found that even though forest sector emissions are compensated for by emission savings from bioenergy use, fewer forest fires, and wood product substitution, the end result is an increase in regional CO_2 emissions compared to BAU as long as the regional sink persists.

To determine a threshold NBP for which bioenergy management reduces atmospheric CO_2 emissions compared with BAU, we applied the same assumptions as used in the LCA. We found that if the NBP drops by 50–60 g C m^{-2} yr^{-1} in currently productive ecoregions or 15–30 g C m^{-2} yr^{-1} in currently less productive ecoregions, bioenergy management would come with CO_2 emissions savings compared to BAU (Fig 1c). Aggregating the ecoregion thresholds translates into a regional mean NBP of 45 g C m^{-2} yr^{-1} or a 41% reduction on average. Reductions in NBP may occur due to increased mortality and/or decreased growth due to climate, fire, or insect outbreaks. However, reductions in NBP from increased harvest do not qualify because harvest increases emissions; wood carbon enters the products/bioenergy chain, where subsequent losses occur. We cannot predict from the data when the threshold NBP would occur because a high resolution process-based model with the ability to incorporate future climate, nitrogen deposition, age dynamics, disturbance and management would need to be used, which is beyond the scope of this study.

Ecoregion threshold NBP is dependent on the scenario treatment removals and area because the fire prevention treatment targets only those areas most likely to burn. For example, to reduce emissions in the Sierra Nevada, baseline NBP would have to decrease by as much as 84 g C m^{-2} yr^{-1} for the bioenergy production scenario versus only 13 g C m^{-2} yr^{-1} for the fire prevention scenario. In ecoregions where current sinks are marginal or weakened by climate, fire, or insect outbreaks there may be a combination of harvest intensity and bioenergy production that reduces forest sector emissions. In nine of the ecoregions where forests are carbon neutral or a source of CO_2 to the atmosphere and/or fire emissions are high for BAU, total CO_2 emissions under the fire prevention scenario could be reduced compared with BAU. They provide examples where management strategies for carbon emission reduction or sequestration should differ from the majority of the region; a one-size-fits-all approach will not work[22]. Also, large areas in the Northern Rockies (for example, Colorado and Wyoming) are at present experiencing increases in forest mortality due to beetle-kill, a trend which could continue in a warmer climate[23]. These areas may already be at or below the threshold NBP; if so, they could benefit from targeted bioenergy implementation. However, simply lowering current regional harvest intensities in areas where NBP is not weakened also reduces emissions (Supplemental Discussion and Fig. S3). Finally, as we have assumed large-scale implementation of these strategies in addition to BAU harvest, we may be overestimating future harvest even though harvest has declined significantly since 1990 because of restrictions placed on harvest on federal lands as part of the Northwest Forest Plan. If the strategies were used to substitute for BAU harvest, the outcome on NBP would be much different (that is, increased for the fire prevention scenario).

Our study is one of the first to provide full carbon accounting, including all of the sinks and sources of carbon emissions from the

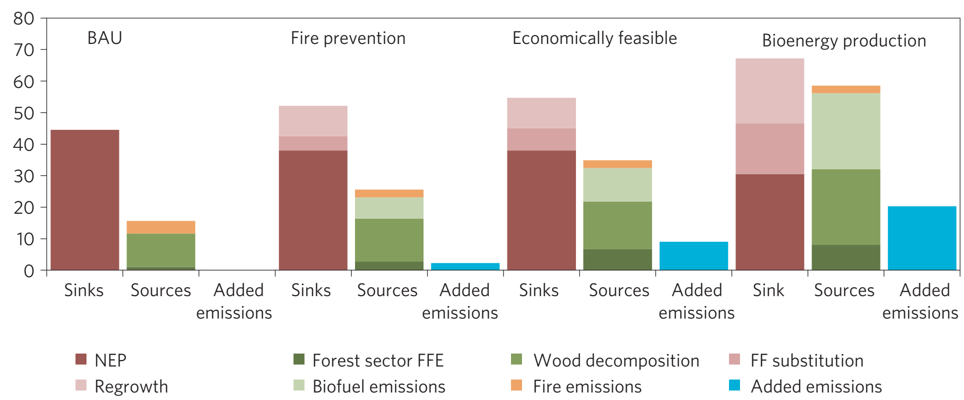

Figure 3 | Total US West Coast forest sector carbon sinks, sources and added emissions relative to BAU under various management scenarios. Units are in Tg C yr^{-1}. Life-cycle assessment estimates account for changes in carbon on land in addition to emissions associated with production, transport and usage of wood, and substitution and displacement of fossil fuel emissions associated with use and extraction. BAU results in the lowest anthropogenic emissions from the forest sector.

forestry sector and the current *in situ* sink, for such a large area. Given the diversity of woody ecosystems in the study region, ranging from highly productive temperate rainforests to less productive semi-arid woodlands, the trends in response probably apply to other temperate regions globally (Supplementary Table S1) where forests are at present a strong net carbon-sink (for example, Eastern US, China and Europe), although the extent of the effect remains to be established.

Greenhouse-gas reduction plans call for up to 10% reductions in emissions by 2020 and forest-derived fuels are being proposed as a carbon-neutral solution to reducing energy emissions. In all of our proposed scenarios, increases in harvest volume on the US West Coast will on average result in regional emission increases above current levels, although there are a few ecoregions where the tested scenarios could result in emission savings. As long as the current *in situ* NBP persists, increasing harvest volumes in support of bioenergy production is counterproductive for reducing CO_2 emissions. In this study region, the current *in situ* NBP in tree biomass, woody detritus and soil carbon is more beneficial in contributing to reduction of anthropogenic carbon dioxide emissions than increasing harvest to substitute fossil fuels with bioenergy from forests.

Although large uncertainty remains for regional forecasts to year 2050 or 2100, it is expected that forest carbon sinks will diminish over time because of ageing of the forests, saturation of the CO_2-fertilization and N-deposition effects, and increased mortality due to climate or insects[24,25]. This would require new assessments to identify management options appropriate for each situation. Carbon-management is not the sole criteria that should be considered when planning forest management. Our findings should thus also be evaluated against other ecosystem services, such as habitat, genetic and species diversity, watershed protection, and natural adaptation to climate change.

Methods

We quantified forest sequestration rates and test forest thinning scenarios across the region using a data-intensive approach which, for the first time, takes into account the diversity of forest characteristics and management. We combined Landsat remote sensing data with inventories and ancillary data to map current forest NEP, NBP, and changes in NBP with three thinning scenarios. The approach can be applied at multiple scales of analysis in other regions.

We combined spatially representative observational data from more than 6,000 federal Forest and Inventory Analysis plots (see Supplementary Methods and Table S7) with remote-sensing products on forest type, age and fire risk[26], a global data compilation of wood decomposition data and 200 supplementary plots[13] to provide new estimates of US West Coast (~34 million hectares) forest biomass carbon stocks (Supplementary Table S8), NEP (the balance of photosynthesis and respiration) and NBP (the *in situ* net forest carbon-sink accounting for removals). We included all forestland in our analysis, across all age classes (20–800 years old) and management regimes. Plot values were aggregated by climatic region (ecoregion), age class and forest type, and this look-up table was used to assign a value to each associated 30 m pixel.

We use regional combustion coefficients to determine fire emissions. Only 3–8% of live tree biomass is actually combusted and emitted in high severity fire in the Pacific Northwest[28], contrary to other studies that report much higher emissions because they assume 30% of all aboveground woody biomass is consumed[27]. Although the latter contradicts extensive field observations[28,29] and modelling studies[30] in the region, we included 30% as the upper-end combustion factor in our sensitivity analysis (Supplementary Table S9).

In addition to the spatially explicit estimates of stocks and fluxes under current management or BAU (current forest harvest), three treatments were designed (fire prevention, economically feasible and bioenergy production; Supplementary Fig. S1a) to reflect the varying objectives of potential future forest management over the next 20 years; within the proposed time period for CO_2 reductions in the US. Areas were prioritized for treatment by fire risk and frequency. The proposed treatments result in additional harvest removals because we assume the current harvest rate for wood products will continue in the future. We limit our specific analysis to the short term because this is the timeframe suitable for policymakers, effectiveness of fire protection treatments, and an appropriate use of the data-driven approach. However, to investigate conditions (for example, sink saturation) that could invalidate our short-term results in the long term, we also calculated the *in situ* NBP at which the atmosphere may benefit from bioenergy removals.

Last, we studied the net effects of the thinning treatments on atmospheric CO_2 by LCA of carbon sources and sinks that includes the post-thinning NBP and wood use (harvest, transport, manufacturing, decomposition, wood product substitution, conversion and use of bioenergy, and displacement of fossil fuel extraction emissions; Supplementary Fig. S1b and Table S4,S5).

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Author contributions

T.W.H. designed and implemented the study with guidance from B.E.L. and S.L. T.W.H., S.L. and B.E.L. co-wrote the paper and S.L. contributed to parts of the analysis. C.W. provided essential data and methods for the analysis and valuable comments on the manuscript.

Additional information

The authors declare no competing financial interests. Supplementary information accompanies this paper on www.nature.com/natureclimatechange. Reprints and permissions information is available online at http://www.nature.com/reprints. Correspondence and requests for materials should be addressed to T.W.H.

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Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest☆

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Abstract

Fire exclusion has led to an unnatural accumulation and greater spatial continuity of organic material on the ground in many forests. This material serves both as potential fuel for forest fires and habitat for a large array of forest species. Managers must balance fuel reduction to reduce wildfire hazard with fuel retention targets to maintain other forest functions. This study reports fuel consumption and changes to coarse woody debris attributes with prescribed burns ignited under different fuel moisture conditions. Replicated early season burn, late season burn, and unburned control plots were established in old-growth mixed conifer forest in Sequoia National Park that had not experienced fire for more than 120 years. Early season burns were ignited during June 2002 when fuels were relatively moist, and late season burns were ignited during September/October 2001 when fuels were dry. Fuel loading and coarse woody debris abundance, cover, volume, and mass were evaluated prior to and after the burns. While both types of burns reduced fuel loading, early season burns consumed significantly less of the total dead and down organic matter than late season burns (67% versus 88%). This difference in fuel consumption between burning treatments was significant for most all woody fuel components evaluated, plus the litter and duff layers. Many logs were not entirely consumed – therefore the number of logs was not significantly changed by fire – but burning did reduce log length, cover, volume, and mass. Log cover, volume, and mass were reduced to a lesser extent by early season burns than late season burns, as a result of higher wood moisture levels. Early season burns also spread over less of the ground surface within the burn perimeter (73%) than late season burns (88%), and were significantly patchier. Organic material remaining after a fire can dam sediments and reduce erosion, while unburned patches may help mitigate the impact of fire on fire-sensitive species by creating refugia from which

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EXHIBIT 2

these species can recolonize burned areas. Early season burns may be an effective means of moderating potential ecosystem damage when treating heavy and/or continuous fuels resulting from long periods of fire exclusion, if burning during this season is not detrimental to other forest functions.

Keywords: Burning season; Conifer forest; Duff; Organic matter; Surface fuel; Woody fuel

1. Introduction

Fire exclusion in mixed conifer forests throughout western North America has led to an unnatural accumulation of twigs, branches, logs, litter, and duff, on the forest floor (Parsons and DeBenedetti, 1979; van Wagtendonk, 1985). Due to the lack of fire and increasing tree densities, the spatial continuity of these surface fuels is now also greater (Miller and Urban, 2000). In addition, more of the large downed logs are in a highly decayed state (Skinner, 2002). When ignited, heavy fuels can contribute to extreme wildfire behavior (Arno, 2000; Brown et al., 2003) with potentially detrimental ecosystem consequences (van Wagtendonk, 1985; Stephens, 1998). The heat released by consumption of heavy fuels may cause torching of nearby trees and the embers released by the torching of trees and burning of decayed snags can lead to long-distance spot fires. Rotten logs are readily ignited by embers and are therefore also important in propagating spot fires.

Besides acting as fuel and potentially influencing fire behavior, organic material on the forest floor provides habitat for a large number of forest species, including small mammals (Tallmon and Mills, 1994; Carey and Johnson, 1995; Ucitel et al., 2003; McCay and Komoroski, 2004), reptiles (James and M'Closkey, 2003), amphibians (Bunnell, 1995), and invertebrates (Harmon et al., 1986; Torgersen and Bull, 1995). The presence of organic matter also influences geomorphic processes. Litter and duff aids in water infiltration and reduces the potential for erosion (Agee, 1973). A strong correlation has been found between post-burn watershed sediment yield and the percentage of forest floor exposed by burning (Benevides-Solorio and MacDonald, 2001; Johansen et al., 2001). Logs and other woody debris can dam and retain sediments on slopes and plays an important role in stream channel dynamics (Harmon et al., 1986; Naiman et al., 2002).

With organic matter on the forest floor acting as fuel, habitat, and providing structural integrity to the forest ecosystem, managers are often faced with conflicting considerations (Brown and See, 1981; Brown et al., 2003; Ucitel et al., 2003). Prescription burning is a commonly used method to treat fuels, but fuel reduction targets to reduce wildfire hazard must be balanced with fuel retention targets to maintain habitat and other forest functions. If too much fuel is removed, the heat released may damage trees excessively and the loss of organic matter may lead to erosion and reduced abundance and diversity of fire-sensitive species (Kauffman and Martin, 1989). Conversely, prescribed fires that consume little of the available fuel may not adequately reduce fire hazard. Achieving such a balance can be particularly challenging when fuel loading is high.

The net ecosystem effect of burning, whether by wildfire or prescribed fire, is often closely tied to the amount of heat released. Heat released is in turn proportional to the amount of available fuel (Alexander, 1982; Johnson and Miyanishi, 1995; Whelan, 1995), but fuel moisture, the physical structure of the fuel bed, weather conditions, and a myriad of other factors lead to a high degree of variability in patterns of consumption and subsequent fire effects (Alexander, 1982; Martin and Sapsis, 1992). The excessive litter, duff, and woody debris found in many areas of the Sierra Nevada where fire has been actively suppressed can result in long-duration heating when fire is returned to the system. In the mixed conifer forest, a significant proportion of the ''fine fuel'' – litter and smaller twigs and stems – is consumed at the flaming front (flaming combustion), leading to a pulse of heat release that has the greatest impact above ground (i.e. canopy scorch on affected trees). The duff layer is typically consumed through smoldering combustion after the flaming front has passed (Kauffman and Martin, 1989). In areas where the duff layer is thick, this smoldering combustion may be of long duration and generate substantially more heat than flaming combustion (Kauffman and Martin, 1989). Because a significant portion of the heat generated by smoldering combustion is transferred downward (Frandsen and Ryan, 1986; Hartford and Frandsen, 1992), soil and below ground processes are often most strongly impacted. Fire can also persist for long periods in large logs. Decayed logs are more likely to be completely consumed by fire than freshly fallen logs (Brown et al., 1985; Kauffman and Martin, 1989; Skinner, 2002), potentially producing a large amount of heat energy.

Even if extensive crown scorch is avoided with the first burn after a period of fire suppression, the heat produced can injure the cambium, kill roots and lead to the death of even large overstory trees (Ryan and Frandsen, 1991; Swezy and Agee, 1991; Stephens and Finney, 2002). In addition, the greater spatial continuity of fuels may cause fire to burn over a greater proportion of the ground surface. Historically, frequent fires are believed to have kept fuel loads relatively low and the lack of fuel continuity contributed to a highly patchy pattern of fire spread (Swetnam, 1993). The patchiness of fire spread under historical conditions may have been important in reducing the impact of fire on fire-sensitive species by creating abundant refugia from which these species could rapidly recolonize burned areas.

The amount of fuel consumed and percentage of the area burned can be controlled to some extent by varying the fuel moisture and weather conditions that prescription burns are conducted under. In similar mixed conifer forests, Kauffman and Martin (1989) reported that early season burns ignited one month after the last spring precipitation event consumed only 15% of the total available fuel, while early fall burns when fuel moisture was much lower consumed 92% of the total available fuel. Percentage consumption of the litter and duff in early and late season burns was significantly correlated with the moisture content of the lower duff layer. Fuel consumption can also vary by the tree species contributing most of the fuel. Agee et al. (1978) noted that pine litter could be effectively reduced by burning in spring, summer, or fall, but drier summer or fall conditions were required to reduce the more compact white fir (*Abies concolor*) and giant sequoia (*Sequoiadendron giganteum*) litter.

Prior to the policy of fire suppression, fires in the mixed conifer zone of the Sierra Nevada burned a given area approximately every 4–40 years (Kilgore and Taylor, 1979; Swetnam, 1993; Caprio and Swetnam, 1995; Skinner and Chang, 1996). In Sequoia and Kings Canyon National Parks, prescription burning has been used to reduce fuels and restore natural ecosystem processes since the late 1960s (Kilgore, 1973). Most of this burning has been done during the fall months, which is within or after the period when the majority of land area is likely to have burned prior to European settlement (mid-summer to early fall) (Caprio and Swetnam, 1995). Early season (late spring/early summer) burns were historically uncommon and usually associated with dry years. Fires in the fall are desirable from a fire management perspective because they are typically followed by the onset of seasonal rain and snow and therefore require less monitoring. However, fall fires potentially have more impact on air quality in the adjacent Central Valley (Cahill et al., 1996), due to stable atmospheric patterns common at this time of year. A greater proportion of the prescription burning in Sequoia and Kings Canyon National Parks has, in the past few years, been conducted earlier in the season under more favorable smoke dispersal conditions.

The purpose of this study was to evaluate differences in surface fuel consumption, fire coverage (proportion of area burned), and coarse woody debris dynamics with early season and late season prescribed fires, to help managers refine burning prescriptions for this vegetation type. The findings are especially relevant to the first restoration burn after a long period of fire suppression.

2. Materials and methods

2.1. Study site description

Three replicate early season prescribed burn, late season prescribed burn, and unburned control units were established in a completely randomized design in Sequoia National Park (Fig. 1). The study site was located on a northwest-facing bench above the Marble Fork of the Kaweah River, adjacent to the Giant Forest sequoia grove, at elevations ranging from 1900 m to 2150 m above sea level. Each unit was 15–20 ha in

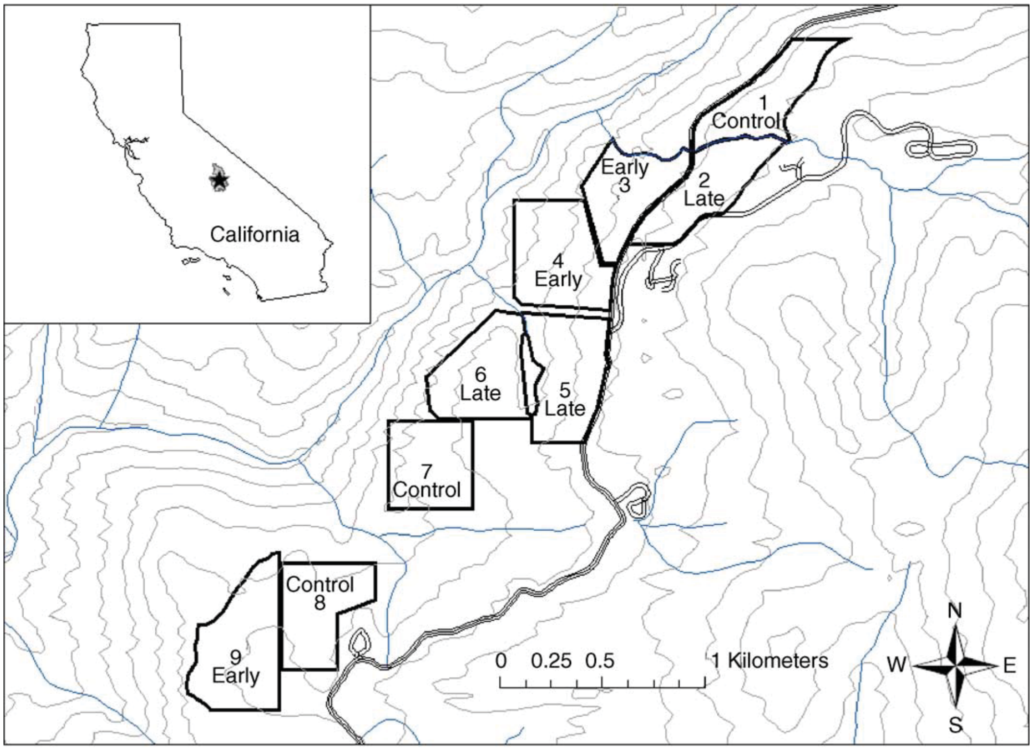

Fig. 1. Map showing location of the early and late season prescribed fire treatment areas in Sequoia National Park, California. The contour interval is 60 m.

size. Tree species in this old-growth mixed conifer forest were, in order of abundance, white fir, sugar pine (*Pinus lambertiana*), incense cedar (*Calocedrus decurrens*), red fir (*A. magnifica* ssp. *shastensis*), Jeffrey pine (*P. jeffreyi*), ponderosa pine (*P. ponderosa*), dogwood (*Cornus nuttallii*), and California black oak (*Quercus kelloggii*). Pre-treatment tree density and basal area averaged 714/ha and 66.5 m^2/ha, respectively. More than half of the trees (370/ha) had a diameter at breast height (dbh) $>$10 cm and numerous large trees were present (41 trees/ha with a dbh $>$80 cm). Cross-dating of wood sections containing fire scars collected from snags indicated that the pre-settlement fire return interval in the study area ranged between 15 and 40 years but the last major fire occurred in 1879 (Caprio and Knapp, unpublished data).

Early season burns were conducted 20 and 27 June 2002 and late season burns were conducted 28 September, 17 and 28 October 2001. Weather data (ambient air temperature, relative humidity, wind speed, and wind direction) were taken hourly immediately prior to and during the burns using a belt weather kit. Conditions were similar during burns within burning season treatment. Ambient air temperature was somewhat higher during the early season burns (range = 16–22 °C) than during the late season burns (range = 13–18 °C). Relative humidity and wind speed ranged from 44 to 68% and 0 to 8 km/h, respectively, during the early season burns and 20 to 63% and 0 to 7 km/h, respectively, during the late season burns. The period of relative humidity $<$40% during the late season burns was confined to the morning of one burn (17 October) and occurred as a temperature inversion dissipated. Relative humidity for much of this burn was within the range experienced during the others.

Ignition was accomplished using drip torches and was initiated at the highest elevation within each burn unit. Three and sometimes four ignition specialists spaced 10–15 m apart walked perpendicular to the slope from higher to lower elevations igniting strips and spot-igniting fuel ‘‘jackpots’’. Burns were mainly strip head fires of low to moderate intensity. With the exception of occasional single small trees that torched, fire was predominantly on the surface.

2.2. Fuel moisture

Fuel moisture measurements were made at the time of ignition for each burn. Woody fuels of different size classes, in addition to litter and duff, were collected in different microenvironments within the burn unit and separately placed into air-tight plastic bags or nalgene bottles. The larger woody fuels were obtained by cutting 1–2 cm wide cross sections out of logs with a chain saw. Samples were returned to the lab, weighed wet, dried in a mechanical convection oven at 85 °C for 48 h, and weighed again. Because several of the duff samples collected prior to one of the early season burns contained a significant amount of mineral soil, separate duff samples were re-collected shortly after the burn in an adjacent unburned forest area with similar aspect, species composition, and canopy cover.

2.3. Surface fuel loading

Mass of surface fuel (dead and down woody fuels plus litter and duff) was estimated both prior to treatment and following treatment using Brown’s planar intercept method (Brown, 1974). Two 20 m transects were installed at each of 36 spatially referenced points located on a 50 m grid within each unit. The direction of the first transect was based on a random bearing (n), and the second transect was placed $n + 120^{\circ}$ from the first. The proximal end of each transect was offset 2 m from the gridpoint to avoid disturbance in the area of the grid point. Number of intercepts of 1-h (hour) (0–6 mm) and 10-h (>6–25 mm) fuels were counted along the first 2 m of the transect, while 100-h (>25–76 mm) fuels were counted along the first 4 m of the transect. The 1000-h fuels (>76 mm) were counted along the entire length of the transect. Diameter, species, and decay class (sound or rotten) of each 1000-h log was noted. A log was considered rotten if it could be dented or broken up with a kick. The maximum height above the ground of elevated dead woody fuel was measured in three adjacent 33 cm long sections in the center of the transect. Litter and duff depth measurements were also taken at three spots along the transect (5 m, 10 m, and 15 m). Depth measurements were made 50 cm to the right of the transect prior to treatment and 50 cm to the left of the transect post-treatment. Because so little of the forest floor was composed of freshly cast leaf and needle material at the time of sampling, we defined litter as both the freshly cast and fermentation layers (fermentation layer = cemented together by fungal growth but the shape and structure of needles, etc. still visible). The duff layer was anything below the fermentation layer down to mineral soil. Fuel loads were calculated using formulas of Brown (1974) with individual tree species constants for bulk density, squared quadratic mean diameter, and non-horizontal correction from van Wagtendonk et al. (1996, 1998). The individual species constants were weighted by the proportional basal area of tree species in the study area. Total litter and duff fuel mass was estimated using fuel depth to weight relationships developed for the study area (described below).

At the time of the second census (post-burn), the total transect length covering areas that burned, did not burn, or were composed of rock were mapped along each Brown’s transect. Patchiness of the burn pattern was estimated by calculating the average number and average size of unburned patches. Brown’s transects in the early season burn units were surveyed shortly after the burns and in the same growing season, while the late season burns were followed by snowfall and could not be evaluated until the following spring. The fuel reduction estimates for the late season burns were therefore corrected for the amount presumed to have been added over the winter and prior to the fuel survey. Because late season burns consumed nearly the entire litter and duff layers where fire passed over the surface (see duff pin methods, next paragraph), all litter, duff, and small woody fuels on burned ground were assumed to have fallen since the burns and were not considered in the calculation of post-burn fuel estimates. Large woody fuel pieces that obviously fell post-burn (i.e. lying in a burned area but showing no visual evidence of combustion) were also not considered. Few large woody fuel pieces fell over the winter in the unburned controls, and these were identified by comparison with pre-treatment data. Other fuel categories in the unburned controls were not similarly corrected, but their amounts were presumed to have been negligible (far more fuel was added to the late season burn plots over the winter due to loss of scorched needles and instability of partially consumed snags).

To more accurately evaluate litter and duff consumption in areas where fire burned, duff pins consisting of 30 cm nails or 75 cm sections of rebar were pounded into and flush with the forest floor and extending into the mineral soil. Four duff pins were installed adjacent to each grid point. Shortly after each burn, pins were reexamined and distances from the top of the duff pin to the top of remaining unburned forest floor material as well as the total distance from the top of the pin to mineral soil were measured.

2.4. Litter and duff depth: weight relationships

Forest floor samples were collected across the study area prior to treatment to develop a regression equation relating forest floor depth to forest floor mass. A 30 cm × 30 cm metal frame was pushed into the forest floor 5 m from the end of one fuel transect per gridpoint, at a random bearing. Litter and duff was excavated using a metal cutter and composition of the litter was scored visually as belonging to one of the three following categories; >80% short needle (*Abies* sp. and *Calocedrus decurrens*), >80% long needle (*Pinus* sp.), and mixed. Litter and duff were bagged separately. To ensure collection of all organic material, duff was collected past the mineral soil surface and later washed to remove the soil and rock portion. After the forest floor sample was removed, the depth of each layer was measured at the center of each side of the excavated square and averaged by layer for that sample. All litter and washed duff samples were dried at 85 °C in a mechanical convection oven for 48 h. After weighing the litter samples, all woody fuels with a diameter less than 7.6 cm were removed from the sample and weighed (woody fuels larger than 7.6 cm were not collected—the sampling frame was moved if the sampling point intersected with a section of woody fuel larger than 7.6 cm). Weights of woody fuels were subtracted from the total sample weight in developing the litter and duff depth: weight relationships.

2.5. Other fuels

Estimates of live fuel mass were not taken because the biomass contained within the understory (tree seedlings, grasses, forbs, and shrubs) was minimal relative to mass of dead and downed surface fuel. Although these live fuels did often burn and occasionally resulted in locally more intense fire activity, the overall contribution to fire effects was likely very low.

2.6. Coarse woody debris

Additional measurements were made on larger logs in order to obtain a better understanding of changes in habitat value, such as cover and volume, that could not be gained from Brown's transect data. Course woody debris (CWD) data were collected using methods similar to those described in Bate et al. (2002). A 4 m × 20 m strip plot was established along the second Brown's transects at every other gridpoint, with the transect forming the centerline of the plot. All logs or portions of logs that were at least 1m in total length and with a large end diameter of at least 15 cm (in or out of the plot) were counted and large end and small end diameters measured. If a log extended outside the plot, diameters were measured at the line of intercept with the plot boundary and the CWD piece. Logs were assumed to end when the diameter fell below 7.6 cm. Logs were not measured if more than half of the log was buried within the forest floor material. Two log lengths were measured—the length within the plot area, and total length. Log number was estimated as a count of logs with midpoints falling within the boundaries of the plot.

2.7. Data analysis

Separate fuel depth to weight regression equations were calculated for litter and duff composed primarily of fir, primarily of pine, and mixed species. In all calculations, the y-intercept was assumed to be equal to zero. The hypothesis of no difference between slopes of the lines for the three forest floor categories was tested using equations given in Zar (1999).

Fuel moisture of different classes and the percentage of residual litter and duff remaining in areas that burned were summarized at the experimental unit level and arcsine square root transformed prior to analysis using one-way ANOVAs with treatment (early season burn and late season burn) as the sole factor. Differences among treatments in fuel and CWD variables were evaluated with analysis of covariance (ANCOVA), using the pre-treatment numbers as a covariate. The treatment $\times$ covariate interaction was included in the model as well in cases where it was statistically significant. Linear contrasts, set a priori, were used to estimate the effect of burning (burns versus unburned control), and the effect of season of burning (early versus late). If the treatment $\times$ covari-covariate interaction was significant, the contrasts were calculated on the interaction at the level of the mean of the covariate. Differences between burning treatments in percentage of area burned, number of unburned patches per 20 m, and unburned patch size were evaluated using one-way ANOVAs. While both the average number of unburned patches and average unburned patch size variables did not require transformation, average percentage of area burned was arcsine square root transformed prior to analysis. A statistical significance level of $P < 0.05$ was used for all tests. Calculations were made using either SYSTAT v. 10 (SPSS Inc., Chicago, IL) or SAS v. 8 (SAS Institute, Cary, NC).

3. Results

3.1. Fuel moisture

Fuels within all size categories were significantly wetter during the early season burns than during the late season burns (Table 1). The difference in moisture was especially pronounced for large woody fuels and duff. Early season fuel moisture was for most woody fuel categories somewhat higher than the range within which Sequoia and Kings Canyon National Parks usually conducts prescribed burns in this vegetation type (Table 1). While woody fuels in the late season were within the prescription range, the 1000 h fuels were on the dry end of the prescription (Table 1).

3.2. Fuel loading and consumption

Separate regression coefficients for the depth to weight relationship were initially calculated for the three tree overstory categories—short needled, long needled, and mixed. However, neither the slope coefficients for the three litter categories nor the slope coefficients for the three duff categories were found to differ significantly from each other. Therefore, all data were combined and single equations were calculated for the litter and duff layers. A significant linear relationship with high r^2 was found between depth and mass for both litter and duff fuel samples (Fig. 2).

Prior to treatment, total fuel load averaged 191.6 Mg/ha across treatments (Table 2). Over half of this fuel (105.7 Mg/ha) was found in the litter and duff layers. Large logs (>7.6 cm diameter) comprised the majority of the woody fuels (77.5 Mg/ha), and 69% were classified as rotten. All surface fuel categories were significantly reduced by either early or late season burning, relative to the unburned control (Table 3). However, significantly less total fuel was consumed by early season burns (Table 3). The early season and late season burns consumed 67% and 88% of the available surface fuel, respectively. When broken down into individual surface fuel categories, significantly less was consumed for most with early

Table 1
Percentage moisture of fuels at the time of early season and late season prescribed burns in Sequoia National Park

Fuel type (fuel diameter)	Fuel moisture (%)		P-value	Fuel moisture prescription range (%)
	Early season (June 2002)	Late season (September/October 2001)		
1 h (0–0.6 cm)	13.5	8.9	0.018	5–12
10 h (0.6–2.5 cm)	12.7	8.8	0.001	6–13
100 h (2.5–7.6 cm)	16.9	10.4	0.032	7–14
1000 h (>7.6 cm)	26.4	10.6	0.020	10–20
Litter	22.5	11.3	0.011	–
Duff	37.9	11.7	0.002	–

Litter was considered the freshly cast and fermentation layers, while duff was considered the humus layer. Statistical significance of difference between treatments was tested using analysis of variance after arcsine square root transformation.

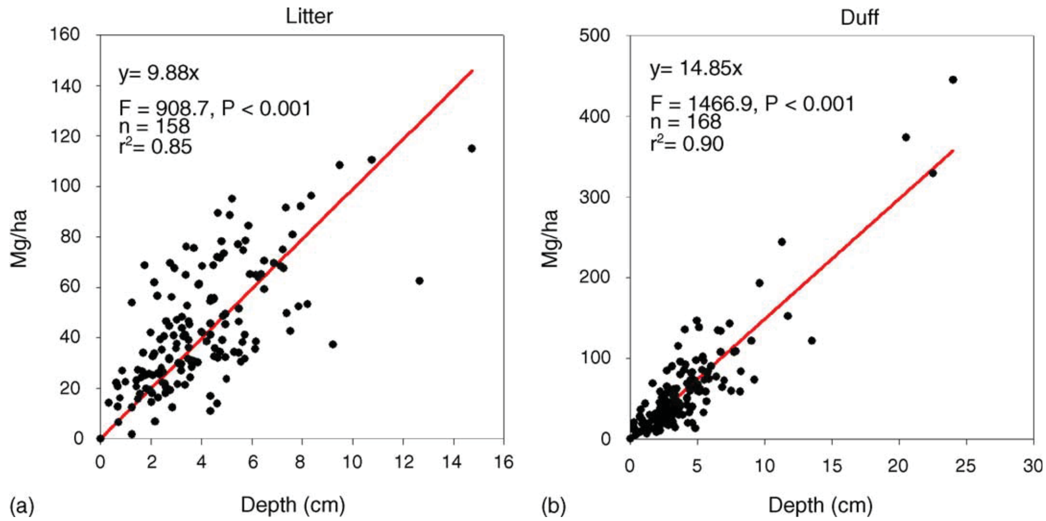

Fig. 2. Depth to mass regressions for (a) litter (freshly cast and fermentation layers) and (b) duff (humus layer) from systematic collections of litter and duff made throughout the study area. Samples were dried at 85 °C for two days before weighing.

season burns, and differences in the 10 h and 1000 h categories were nearly statistically significant. Average height of woody surface fuel above the forest floor was significantly reduced by fire, but there was no difference between the early and late season prescribed fire treatments (Table 3).

Less fuel consumption by early season burns was due to both a significantly greater amount of residual fuel remaining in areas that burned (Fig. 3a), and significantly less area burned within the fire perimeter (Fig. 3b). Early season burns left approximately five times more litter and duff unconsumed in areas where fire passed over the forest floor than late season burns. Early season burns were also significantly patchier (Fig. 3c) and the size of these unburned patches tended to be smaller (Fig. 3d).

3.3. Coarse woody debris

Large quantities of coarse woody debris were found in the study area. Prior to the prescribed burns, number of downed logs averaged 173/ha (91 with a diameter <30 cm and 82 with a diameter ≥30 cm) and covered an average of 4.3% of the ground surface area (Table 4). The total length of logs averaged 1064 m/ha, with a total volume of 190 m^3/ha (Table 4). Log mass averaged 61.7 Mg/ha, less than the 78.6 Mg/ha of 1000 h fuel estimated with Brown's transects. The

Table 2
Mean mass (standard error in parenthesis) of different fuel categories and height of woody fuels above the litter surface prior to and after treatment by early season and late season prescribed burns

Treatment	Time of survey	1 h (<0.6 cm) (Mg/ha)	10 h (0.6–2.5 cm) (Mg/ha)	100 h (2.5–7.6 cm) (Mg/ha)	1000 h (>7.6 cm) (Mg/ha)	Litter (L&F layers) (Mg/ha)	Duff (H layer) (Mg/ha)	Fuel total (Mg/ha)	Fuel height (cm)
Unburned	Pre-treatment	1.4 (0.04)	2.8 (0.1)	4.8 (0.3)	95.8 (11.0)	40.5 (4.8)	66.8 (5.6)	212.0 (17.3)	10.6 (1.5)
Early burn	Pre-treatment	1.1 (0.1)	2.7 (0.2)	4.7 (0.3)	70.6 (6.6)	42.8 (0.7)	59.5 (5.9)	181.3 (12.2)	11.3 (1.2)
Late burn	Pre-treatment	1.0 (0.1)	2.4 (0.2)	4.4 (0.3)	66.2 (9.5)	38.0 (1.8)	69.5 (2.9)	181.4 (13.5)	9.4 (0.2)
Unburned	Post-treatment	1.1 (0.02)	2.5 (0.1)	5.0 (0.2)	86.5 (9.8)	37.9 (1.4)	76.5 (0.9)	209.4 (10.2)	12.2 (1.4)
Early burn	Post-treatment	0.3 (0.04)	0.7 (0.1)	1.6 (0.1)	31.0 (4.2)	7.7 (0.3)	18.4 (2.2)	59.7 (5.3)	4.0 (0.6)
Late burn	Post-treatment	0.1 (0.04)	0.2 (0.1)	0.3 (0.1)	15.0 (1.3)	2.0 (0.5)	4.9 (1.8)	22.5 (3.2)	4.0 (1.1)

Table 3
Significance of analysis of covariance results for fuel size categories and fuel height after application of the burning treatments

Effect	d.f.	*P*-value							
		1 h (<0.6 cm)	10 h (0.6–2.5 cm)	100 h (2.5–7.6 cm)	1000 h (>7.6 cm)	Litter (L&F layers)	Duff (H layer)	Fuel total	Fuel height
Covariate	1	0.019	0.031	0.001	0.051	<0.001	0.002	0.007	0.337
Treatment	2	<0.001	<0.001	<0.001	0.002	<0.001	<0.001	<0.001	0.003
Burn vs. unburned	1	<0.001	<0.001	<0.001	0.001	<0.001	<0.001	<0.001	0.001
Early vs. late	1	0.008	0.070	0.001	0.068	0.004	0.004	0.004	0.441
Error	5								

Pre-treatment data were used as a covariate. The treatment × covariate interaction was not significant for any of the dependent variables and was therefore not included.

difference is likely due to the more restrictive definition of CWD.

Burning treatments resulted in a significant reduction in all CWD measures except log number (Table 5). Many logs were not completely consumed by fire. While late season burns resulted in significantly greater reduction in log cover, log volume, and log mass compared to early season burns, reduction in log length and log number did not differ between burning season treatments (Table 5). This difference between CWD variables in response to burning season treatment may be related to the tendency of early season burns to consume just the outer layers of many of the larger logs. While the late season burns also often did not consume the entire log, a greater proportion of the wood circumference was typically consumed. The reduction in CWD mass between burning season treatments was similar for the

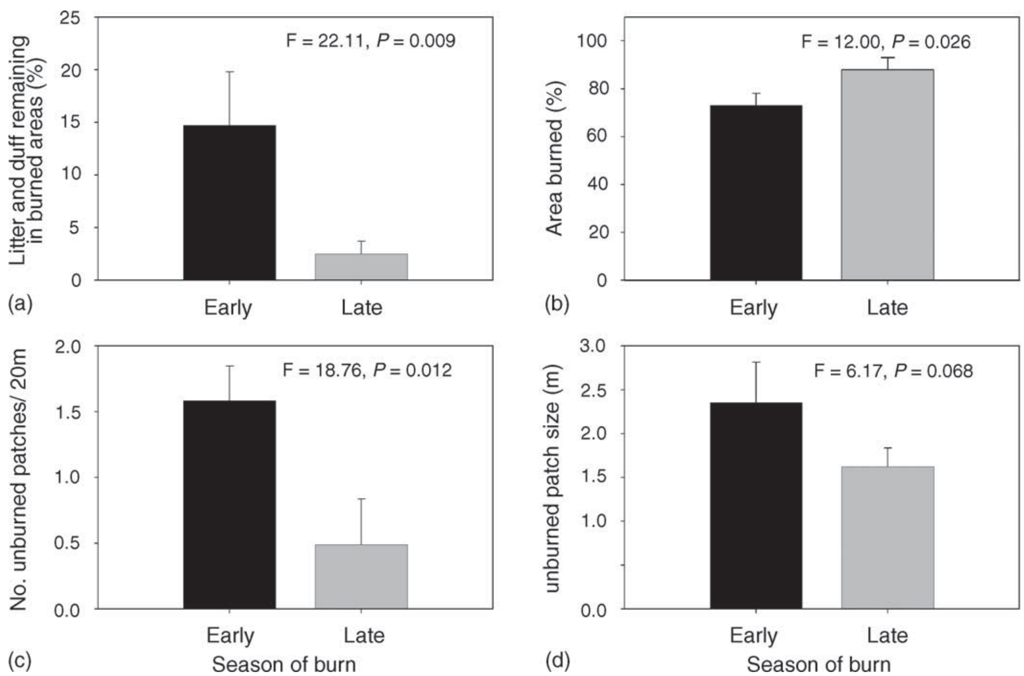

Fig. 3. Average percentage of residual litter and duff remaining in areas that burned (a), average percentage of area burned (b), average number of unburned patches within 20 m long Brown's fuel transects (c), and average size of unburned patches located within 20 m long Brown's fuel transects (d) in early season and late season prescribed fires.

Table 4
Means (standard errors in parentheses) of coarse woody debris attributes prior to and after treatment by early season and late season prescribed burns

Treatment	Time of survey	No. logs/ha, <30 cm diameter	No. logs/ha, ≥30 cm diameter	Log length (m/ha)	Log cover (%)	Log volume (m^3/ha)	Log mass (Mg/ha)
Unburned	Pre-treatment	90.3 (4.0)	108.8 (26.1)	1210.7 (128.3)	5.2 (0.6)	246.1 (19.6)	79.2 (6.4)
Early burn	Pre-treatment	134.3 (4.6)	76.4 (6.9)	1208.2 (51.5)	4.5 (0.7)	184.8 (48.6)	58.8 (14.8)
Late burn	Pre-treatment	48.6 (8.0)	62.5 (13.9)	772.2 (102.9)	3.3 (0.5)	138.5 (19.7)	47.2 (7.0)
Unburned	Post-treatment	111.1 (17.5)	104.2 (28.9)	1155.6 (102.0)	4.6 (0.5)	204.5 (22.8)	69.1 (8.5)
Early burn	Post-treatment	113.4 (11.6)	60.2 (6.1)	708.6 (35.8)	2.2 (0.1)	75.1 (51.3)	26.2 (2.6)
Late burn	Post-treatment	34.7 (10.6)	32.4 (6.1)	302.9 (57.3)	0.8 (0.1)	20.0 (2.1)	7.4 (0.9)

two measurement methods (percentage reduction of these components with early and late season burns averaged 55% and 77%, respectively, when measured using Brown's transects, and 56% and 84%, respectively, when measured using strip plot surveys).

4. Discussion

Fuel moisture was likely the main cause of differences in fuel consumption with early and late season burns. Because energy is necessary to drive off water before combustion is possible, more energy is required to propagate flaming combustion in moist fuels than dry fuels (Frandsen, 1987; Nelson, 2001). Consumption of large woody fuel is often quite high at moisture levels equal to or less than 10–15%, but less than half of these fuels are typically consumed when moisture levels exceed 25–30% (Brown et al., 1985). In this study, some logs were likely drier, while others, particularly partially rotten logs in shady locations, were likely considerably wetter than the average 26% moisture content of large logs (1000 h fuels) at the time of early season burns. Kauffman and Martin (1989) found that moisture content of the lower duff layer was the most important fuel or weather-related variable in multiple regression models of duff consumption. Little duff is consumed when the moisture content exceeds 110%, and the duff layer may burn independently of surface fire at a moisture content of less than 30% (Sandberg, 1980). Between these two values, consumption is related to both moisture content and heat of the surface fire (Reinhardt et al., 1991). Brown et al. (1985) reported an inverse linear relationship between duff moisture and percent duff consumption for mixed conifer forests in the northern Rocky mountains, and suggested that moisture content may become an even stronger predictor of consumption the deeper the duff layer.

Fuel moisture also influences fuel consumption through its effect on the amount of area within the

Table 5
Significance of analysis of covariance results for coarse woody debris attributes after application of the burning treatments

Effect	d.f. (treatment × covariate interaction included)	P-value					
		No. logs/ha, <30 cm diameter	No. logs/ha, ≥30 cm diameter	Log length (m/ha)	Log cover (%)	Log volume (m^3/ha)	Log mass (Mg/ha)
Covariate	1	0.132	0.353	0.070	0.009	0.007	0.007
Treatment	2	0.125	0.398	0.008	0.003	0.201	0.134
Treatment × covariate	2	–	–	–	–	0.038	0.024
Burn vs. unburned	1	0.122	0.251	0.003	0.001	0.004	0.003
Early vs. late	1	0.166	0.409	0.184	0.045	0.022	0.011
Error	5 (3)						

Pre-treatment data were used as a covariate. The treatment × covariate interaction was included when significant. In these cases, the contrasts for effect of treatments were calculated on the interaction at a value set to the mean of the covariate.

fire perimeter that burns. In fire simulation studies, Hargrove et al. (2000) reported that modeled fires under high fuel moisture conditions produced dendritic and patchy burn patterns, while at lower fuel moisture conditions, little of the landscape within the fire perimeter remained unburned. The model was based on fire ignition and spread in a gridded landscape where the probability of spread to neighboring fuels was evaluated in eight directions. The probability that fire will propagate to neighboring fuels (I) is reduced at higher fuel moisture levels. Interestingly, the maximum variability in fire burn pattern was predicted to occur near the critical threshold of $I = 0.25$, below which most fires remained small or went out. Using a different model, Miller and Urban (2000) also predicted that the functional connectivity of surface fuels would be reduced under higher fuel moisture conditions. Our findings of significantly reduced amount of area within the fire perimeter burned and greater patchiness of early season burns conducted under higher fuel moisture conditions are consistent with these model predictions. Slocum et al. (2003) similarly found that prescribed burns in Florida conducted under higher fuel moisture conditions were patchier than burns conducted when fuels were drier.

Based on fire scar dendrochronology data collected adjacent to our study area, Swetnam (1993) suggested that a fire-free interval as long as that seen today is likely unprecedented in the last 2000 years. By the time of our prescribed burns, a minimum of three to four cycles of fire had likely been missed. As a result, the fuel mass and CWD attributes reported here (log number, log length, log cover, log volume, and log mass) were likely considerably higher than what might have been present without fire suppression. The average of 191.6 Mg/ha of fuel found prior to the prescribed burns in this study was greater than fuel loadings reported for second growth and old-growth mixed conifer forests in northern portions of the Sierra Nevada by Kauffman and Martin (1989) (range, 74.8–163.9 Mg/ha). Keifer (1998) estimated the amount of pre-burn fuel to be 143.5 Mg/ha in several plots of mixed conifer/giant sequoia forest in Sequoia National Park that hadn't burned in over 40 years. A nearby mixed conifer that had also not experienced fire since pre-settlement times contained 210 Mg/ha of fuel (Mutch and Parsons, 1998), which is comparable to levels found in this study.

Accurate estimates of fuel mass and consumption are essential to predicting fire effects. Slopes of the litter and duff depth to weight regression relationships developed for this study were very similar to the estimates reported by van Wagtendonk et al. (1998) for white fir (litter: 9.88 versus 10.05 for this study and van Wagtendonk et al. (1998), respectively; duff: 14.85 versus 15.18 for this study and van Wagtendonk et al. (1998), respectively), helping to validate the accuracy of both sets of numbers. The 88% reduction in fuel mass recorded in the late season burn treatment was comparable to levels of consumption seen in other fires in mixed conifer forests conducted under dry fall conditions (Kauffman and Martin, 1989; Kilgore, 1972; Mutch and Parsons, 1998), slightly lower than the 91% fuel reduction reported for a dry early fall prescribed fire on a nearby southeast-facing slope in the same watershed (Stephens and Finney, 2002), and somewhat greater than an average consumption of 71% for multiple prescribed fires conducted under a range of fuel moisture conditions in Sequoia National Park (Keifer, 1998). Fuel reduction in the early season burns (67%), while still substantial, was within the range of values reported by Kauffman and Martin (1989) for late spring burns in Sierran mixed conifer forest (61–83%). Our estimate of the percentage of ground surface area within the fire perimeter that burned in the late season prescribed fires (88%) was very close to estimates of Kilgore (1972), who found that 80% of study plots within a late season prescribed fire unit were completely burned, while 14% of plots were partially burned. Similar data has, to our knowledge, not been collected in this vegetation type for early season burns.

With a complete understanding of fire effects often lacking, resource managers may seek to conduct prescription burning operations for restoring the process of fire to these forests that mimic historical fires that the trees and other forest organisms on a site evolved with (Moore et al., 1999; Stephenson, 1999). While the majority of land area historically burned during the dry late summer to early fall period, prescribed fires at the same time of year may now generate fire effects outside of the historical norm, due to the current high fuel loading conditions. These fire effects are potentially a function of not only of changes in the abundance of fuels, but the changes in the proportion of fuels that are in a highly decayed state. The dominant woody fuels in this system tend to decompose relatively rapidly. Harmon et al. (1987) reported a half life of only 14 years for white fir logs. However, with frequent low to moderate severity fires, large amounts of decomposed wood on the forest floor was likely historically uncommon (Skinner, 2002). Under dry fuel moisture conditions, decomposed logs are more likely to be completely consumed than sound, more recently fallen logs (Kauffman and Martin, 1989; Skinner, 2002; Stephens and Finney, 2002). The cracking and breakage of decomposed wood over time also increases the surface to volume ratio, leading to more rapid consumption and therefore potentially greater heat generation.

In addition to the high surface fuel loadings at the time of the burns, the spatial continuity of these fuels was also likely greater than found historically. Frequent fires are predicted to reduce fuel continuity (Miller and Urban, 2000), and historical fires were therefore likely quite patchy. This same finding can be inferred from Swetnam (1993), who reported a negative relationship between the proportion of trees exhibiting fire scars in any given year and the fire frequency. With more time between fires, the extra fuel buildup apparently aided in fire spread. It is likely that prescribed fires conducted under current levels of fuel continuity and under dry conditions where fire spread is not limited by fuel moisture will result in a greater proportion of the area within the fire perimeter burned, compared to historical fires.

By burning less of the landscape within the fire perimeter, the pattern of consumption of the early season fires was possibly more similar to historical fires. This patchiness may aide in the post-fire recovery of plant and animal populations, as the spatial distribution and size of unburned islands can be important for the recruitment and persistence of species that are sensitive to fire (Turner et al., 1997). Andrew et al. (2000) suggested that refuges provided by unburned logs may allow ant diversity to be maintained, even with frequent fuel-reduction fires. The abundance and distribution of unburned patches may also influence the probability of erosion. From rainfall simulation experiments, Johansen et al. (2001) found that sediment yields resulting from erosion did not change greatly whether 0% or 60–70% of the ground surface was exposed by burning. However, once the threshold of 60–70% of bare ground was exceeded, sedimentation increased sharply, possibly because of the greater probability of the connectedness of bare patches, which made infiltration and sediment capture less likely. The amount of bare ground exposed by early season burns in this study was close to the threshold value reported by Johansen et al. (2001), while the bare ground exposed by late season burns substantially exceeded this threshold. Such erosion simulations may be helpful for better defining target burn area percentages in prescribed fires.

While this study demonstrated that early season burns were not as effective at reducing fuel loading, less fuel consumption and less area within the fire perimeter burned may be beneficial for the recovery rate of important ecosystem components. In addition, more habitat for animal species dependent on CWD was maintained. However, the habitat value of charred but only partially consumed logs, relative to unburned logs, is unknown. Comparisons of these burns with historical fires are not possible, but the early season burns may have produced a landscape closer in many ways to that found after historical fires. The idea of utilizing early season burning as a tool to more gradually get back to the desired forest conditions is not new. Kilgore (1972) described two different strategies for reintroducing fire to the mixed conifer forest after a period of fire exclusion—either a relatively hot ‘‘restoration’’ burn that consumes a large proportion of the total fuel and results in significant mortality of trees, followed by additional burns at longer intervals (necessary because fine fuel accumulation will be slower with fewer remaining overstory trees), or a milder restoration burn followed by additional burns at shorter intervals. Both Arno (2000) and Allen et al. (2002) suggested that fire-induced damage could be reduced by successive burns starting with damp fuels. In the Sierra Nevada, higher fuel moisture conditions can be found both early in the burning season after snow melt, or following the first fall rains but prior to snowfall that persists on the ground. The latter conditions do not occur in all years, and the window of opportunity is typically narrow if it does. Thus, to meet burn area targets with currently available resources and burning strategies will likely continue to result in substantial burning being conducted during the early season.

Considering burning season as a tool to obtain the desired fire effects needs to also balance other factors that could be influenced by season. For example, earlier burns often occur during the growth or active phase of many organisms, which could potentially result in undesired impacts. Managers have sometimes elected not to conduct burns during bird nesting season, especially for sensitive species that nest in the forest understory (Robbins and Myers, 1992) and early season burns when conditions are moist may coincide with the peak of amphibian surface activity (Pilliod et al., 2003). However, as shown in this study, early season burns conducted under higher fuel moisture conditions also consume less of the forest floor and CWD that provides habitat for these species. Agee (1993) suggested that fires occurring during active growth phase of trees may be more injurious than fires occurring during the dormant season. Early season burns can lead to higher tree mortality by killing more of the fine surface roots of conifers (Swezy and Agee, 1991). In addition, McHugh et al. (2003) found that early season burns result in higher bark beetle activity and greater secondary mortality of some conifer species. All of these factors will need to be considered in decisions about the most appropriate time of year to conduct the first restoration burn after a period of fire suppression. Studies to evaluate potential impacts of burning season on these additional ecosystem components are in progress.

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Assessing the Methane Emissions from Natural Gas-Fired Power Plants and Oil Refineries

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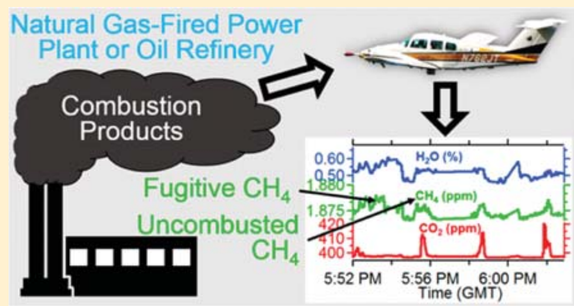
Supporting Information

ABSTRACT: Presently, there is high uncertainty in estimates of methane (CH_4) emissions from natural gas-fired power plants (NGPP) and oil refineries, two major end users of natural gas. Therefore, we measured CH_4 and CO_2 emissions at three NGPPs and three refineries using an aircraft-based mass balance technique. Average CH_4 emission rates (NGPPs: 140 ± 70 kg/h; refineries: 580 ± 220 kg/h, 95% CL) were larger than facility-reported estimates by factors of 21–120 (NGPPs) and 11–90 (refineries). At NGPPs, the percentage of unburned CH_4 emitted from stacks (0.01–0.14%) was much lower than respective facility-scale losses (0.10–0.42%), and CH_4 emissions from both NGPPs and refineries were more strongly correlated with enhanced H_2O concentrations (R^2_{avg} = 0.65) than with CO_2 (R^2_{avg} = 0.21), suggesting noncombustion-related equipment as potential CH_4 sources. Additionally, calculated throughput-based emission factors (EF) derived from the NGPP measurements made in this study were, on average, a factor of 4.4 (stacks) and 42 (facility-scale) larger than industry-used EFs. Subsequently, throughput-based EFs for both the NGPPs and refineries were used to estimate total U.S. emissions from these facility-types. Results indicate that NGPPs and oil refineries may be large sources of CH_4 emissions and could contribute significantly (0.61 ± 0.18 Tg CH_4/yr, 95% CL) to U.S. emissions.



INTRODUCTION

The abundance and accessibility of underground natural gas reserves, paired with rapid technological advancements in horizontal drilling and hydraulic fracturing techniques, have given rise to a booming natural gas industry and record-low natural gas prices. Natural gas is considered a cleaner fuel alternative to coal, producing roughly 56% the amount of CO_2 per unit of energy as coal,[1] and therefore, holds appeal as a "bridge fuel" during transition to renewable energy technologies.[2] Despite the environmental benefits of natural gas as an alternative fuel source, the primary constituent of natural gas is methane (CH_4), a relatively short-lived greenhouse gas with 28–34 and 84–86 times the cumulative radiative forcing of CO_2 on a mass basis over 100 years and 20 years, respectively.[3] Recent studies indicate that CH_4 leakage into the atmosphere may negate its advantages, for instance, a loss rate of 1.5% from natural gas production processes would increase the 20 year climate impact of natural gas by 50%.[4,5] Therefore, identifying significant sources of CH_4 emissions is imperative for effective development of methods to control emissions of greenhouse gases from the oil and natural gas industry.

While CH_4 emission rates from throughout the natural gas supply chain have been recently reported in the literature, there is less understanding regarding emissions from natural gas-fired power plants (NGPP) and crude oil refineries, both of which use large quantities of natural gas[6] and hence are potentially large sources of CH_4 emissions. Increased natural gas consumption by these facility-types has been driven by the combination of low natural gas prices and increased environmental regulations,[7,8] which for instance, has resulted in many coal-fired power plants in the U.S. converting to natural gas for energy generation.[9] Likewise, construction of new NGPPs is also rapidly rising, and in 2015 roughly 40% of new plants producing >1 megawatts (MW) of energy were natural gas-fired.[10] Furthermore, oil refineries are quickly shifting toward natural gas to fuel various equipment-types, including process and utility heaters, hydrogen generation units, and gas turbines, and consumed 893 200 million cubic feet of natural gas in 2014.[11,12]

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Here, we aimed to evaluate emissions from these major natural gas end users by performing a series of measurements of CH_4 and CO_2 emissions at three NGPPs and three oil refineries in Utah, Indiana, and Illinois using an aircraft-based mass balance technique. Hourly CH_4 and CO_2 emission rates are presented and used to obtain CH_4 emission factors (EF) in terms of CO_2 emissions for each facility ($EF_{facility}$). Co-location of CH_4 with CO_2 or H_2O emissions was assessed to understand if CH_4 emissions originated from combustion- or noncombustion-related equipment. Since NGPPs are only required to report combustion-related CH_4 emissions to regulatory agencies,[13,14] we calculated throughput-based CH_4 loss rates and heat input-based EFs for both stack-related combustion emissions as well as total facility emissions and compared our calculated EFs to the Environmental Protection Agency's (EPA) Greenhouse Gas Reporting Program's (GHGRP) default EF.[14] Throughput-based EFs were also calculated for the oil refineries measured in this study. Emissions for both NGPPs and refineries were then extrapolated to the U.S. national-scale using EFs calculated from this data set and U.S. activity factors from the EPA's Air Markets Program Data (AMPD) and Energy Information Administration to estimate total annual emissions from these facility-types. Results from this study support the existing need to better understand the potential of NGPPs and oil refineries as contributors to annual U.S. CH_4 emission totals, while also seeking to elucidate the source of emissions (e.g., combustion- or noncombustion-related) at these facilities.

EXPERIMENTAL METHODS

Flight Design and Emission Rate Quantification. Six flights were performed at three combined-cycle NGPPs and three refineries from July 30−October 1, 2015. To quantify facility emissions, an aircraft-based mass balance approach was used and flights were performed using Purdue's Airborne Laboratory for Atmospheric Research (ALAR, https://www.science.purdue.edu/shepson/research/BiosphereAtmosphereInteractions/alar.html), which is a twin-engine Beechcraft Duchess equipped with a Picarro Cavity Ring-Down Spectroscopy (CRDS, model G2401-m) analyzer for real-time, high frequency measurements of CH_4, CO_2, CO, and H_2O.[15−19] The aircraft is also outfitted with a Best Air Turbulence (BAT) probe for high-precision, three-dimensional wind measurements and a global positioning system/inertial navigation system (GPS/INS) for location tracking and wind measurements.[20] Both in-flight and on-ground CH_4 and CO_2 concentration calibrations were performed daily using three NOAA-certified gas cylinders and measurement precisions were ~0.15 ppm (CO_2) and ~1.4 ppb (CH_4).[21]

Prior to each mass-balance flight experiment (MBE), the facilities were circled in-flight to determine if emission of CH_4 or CO_2 was occurring, and if it could be unambiguously attributed to the target. To perform the experiment, a series of 8−14 horizontal transects was flown approximately 1−4 km downwind of the site. Each transect was flown at a unique altitude, ranging from as low to the ground as is safe to the top of the boundary layer and spaced approximately 50−100 m apart. The ends of each transect extended sufficiently past the edge of the plume to measure background air. For MBEs where the top transects do not capture the full height of the plume, a vertical profile was conducted to estimate the height of the boundary layer. The CH_4 emission rate was then calculated according to eq 1 based on previously described methods.[15−19]

$$\text{emission rate}_{CH_4} = \int_0^{z_i} \int_{-x}^{+x} \Delta A \cdot U_{\perp} \, dx dz \tag{1}$$

In eq 1, for each point along the transects, the enhancement of analyte (CH_4 or CO_2) concentration above background concentration, ΔA [mol/m^3], was multiplied by the perpendicular component of the wind speed, $U_\perp$ [m/s]. The resulting point-by-point flux values [mol/m^2-s] across each transect were interpolated to a two-dimensional gridded surface by kriging,[22] integrated laterally across the horizontal width of the plume ($-x$ to $+x$) and vertically from the ground (0) to the top of the boundary layer (z_i), to a resolution of 100 and 10 m, respectively, to provide CH_4 and CO_2 emission rates in [mol/s], which were then converted to [kg/h] to be consistent with industry units. Explanation of uncertainty determination is provided in the SI.

Emission Factor Determination. *Calculation of Facility-Based CH_4:CO_2 EFs ($EF_{facility}$) at NGPPs and Refineries.* U.S. inventories report annual CO_2 emissions for NGPPs and refineries, but currently do not account for CH_4 emissions from noncombustion-related processes. Therefore, a facility-wide CH_4 emission factor based on CO_2 emissions, $EF_{facility}$ [kg CH_4/kg CO_2] was determined for the three NGPPs and three refineries by dividing the mass balance-derived facility-wide CH_4 emission rate [kg/h] by the mass balance-derived facility-wide CO_2 emission rate [kg/h] (eq 2).

$$EF_{facility}\left[\frac{\text{kg } CH_4}{\text{kg } CO_2}\right] = \frac{CH_4 \text{ ER}_{facility}[\text{kg } CH_4]}{CO_2 \text{ ER}_{facility}[\text{kg } CO_2]} \tag{2}$$

$EF_{facility}$ was then multiplied by annual CO_2 emission rates (kg/h) reported to the GHGRP (for the NGPPs and refineries) and AMPD (for the NGPPs) to approximate annual CH_4 emissions from these facilities. This method assumes that the CH_4:CO_2 ratio is constant throughout the year for simplification. Note that the CH_4:CO_2 ratio was used as an EF due to high accuracy, hourly data for CO_2 emissions from the AMPD.

Calculation of Stack-Based CH_4:CO_2 EFs (EF_{stack}) at NGPPs. For the three NGPPs, emissions were also sampled exclusively from stacks, the primary source of combustion emissions, to derive a stack-based CH_4 emission factor based on CO_2 emissions, EF_{stack} [kg CH_4/kg CO_2]. Stacks were sampled either by flying directly above the stack or by circling the stack at a distance of <200 m. CO_2 peaks were used to determine the start and end points of the stack emission and a linear fit was applied between these points to define background, which was subtracted to give ΔCO_2 and ΔCH_4 (SI Figure S1). A standard linear regression was performed for ΔCH_4 (ppm) versus ΔCO_2 (ppm) using daily stack sampling data from each site and the regressions were forced through zero. The slope of the line (CH_4 ppm/CO_2 ppm) was converted to mass units (e.g., kg/kg) by multiplying by the ratio of the molecular weights of CH_4 to CO_2 (16/44) to yield EF_{stack} [kg CH_4/kg CO_2]. Stack emission factors were also calculated by an alternative method to verify results, as described in the SI. Where available, we compare $EF_{facility}$ to EF_{stack} to better understand the source of facility CH_4 emissions.

Throughput-Based Loss Rates and EFs at NGPPs. To calculate throughput-based loss rates for the NGPPs, hourly CO_2 emission and heat input data for P1−3 was downloaded from the EPA's AMPD Web site (https://ampd.epa.gov/ampd/).[13] Note that heat input is the energy content of fuel given in million British thermal units (mmBtu) where 1 Btu equals 1055 J. Using the start and end times of each flight, the hourly reported data from the AMPD was adjusted based on the true sampling times

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Table 1. Facility Emission Rates and CH_4:CO_2 Emission Factors[a]

		CH_4 ER (kg/h)		CO_2 ER (×10,000 kg/h)			CH_4:CO_2 EF (kg CH_4/kg CO_2)		
site	date	MBE ($\pm 2\sigma$)	RP	MBE ($\pm 2\sigma$)	RP	AMPD hourly	$EF_{facility}$ (EF $\pm 1\sigma$)	EF_{Stack} (EF $\pm 1\sigma$)	EF_{stack}/ $EF_{facility}$
P1	9/19	N.U.		N.U.		12	–	$1.5(\pm 0.5) \times 10^{-5}$	–
	9/20	120 ± 90	2	11 ± 9	12	12	$1.1(\pm 0.6) \times 10^{-3}$	$3.2(\pm 0.6) \times 10^{-4}$	0.29
	9/21	75 ± 30		17 ± 6		12	$4.4(\pm 1.2) \times 10^{-4}$	$3.3(\pm 0.4) \times 10^{-5}$	0.08
	avg.						$7.7(\pm 4.4) \times 10^{-4}$	$1.2(\pm 0.3) \times 10^{-4}$	0.16
P2	9/19	N.U.		N.U.		20	–	$7.0(\pm 1.5) \times 10^{-5}$	–
	9/20	N.U.	4	N.U.	20	20	–	$9.4(\pm 2.5) \times 10^{-5}$	–
	9/21	84 ± 76		24 ± 22		23	$3.5(\pm 2.3) \times 10^{-4}$	$2.6(\pm 1.1) \times 10^{-5}$	0.07
	avg.							$6.3(\pm 1.8) \times 10^{-5}$	–
P3	9/25	240 ± 70	2	18 ± 5	10	34	$1.3(\pm 0.3) \times 10^{-3}$	$7.8(\pm 0.9) \times 10^{-5}$	0.06
R1	7/31	360 ± 200	4	180 ± 110	2.3		$2.0(\pm 0.8) \times 10^{-4}$		
	10/1	N.U.		N.U.			–		
R2	7/31	540 ± 210	51	100 ± 60	19		$5.4(\pm 1.9) \times 10^{-4}$		
R3	9/25	830 ± 240	27	46 ± 13	28		$1.8(\pm 0.4) \times 10^{-3}$		

[a]Abbreviations: ER, emission rate; EF, emission factor; RP, EPA's 2014 Greenhouse Gas Reporting Program data;[14] AMPD, EPA's Air Markets Program Data;[13] fac., facility; N.U., mass balance flight data was not usable due to poor meteorological conditions or due to partial capture of facility emissions.

(see SI). Hourly throughput estimates [kg CH_4/h] were determined using these time-adjusted heat inputs [mmBtu], the conversion factor 1.02 [mmBtu/MCF] (where MCF is 1000 ft^3), the density of CH_4 at 15 °C and 1 atm (19.2 [kg CH_4/MCF]), and the assumption that 95% of natural gas is CH_4, using eq 3.

Using the calculated $EF_{facility}$ and EF_{stack}, projected annual CH_4 emission rates based on annual CO_2 emissions reported to the AMPD and GHGRP were calculated by multiplying EF_{stack} [kg CH_4/kg CO_2] and $EF_{facility}$ [kg CH_4/kg CO_2] by annual CO_2 emissions [kg CO_2/year] and then converting the resulting CH_4 emission rate [kg CH_4/year] to kg CH_4/h.

Annual throughput-based CH_4 loss rates for the NGPPs were then determined (eq 4) for both stack-only emissions and total facility emissions using the calculated projected annual CH_4 emission rates [kg CH_4/h] based on annual CO_2 emissions reported to the AMPD only, since the AMPD is based on real-world measured data from continuous emissions monitoring systems, whereas the GHGRP data is based on engineering calculations using outdated emissions factors.

The EPA's GHGRP currently requires NGPP operators to calculate annual combustion-related CH_4 emissions using a default heat input-based EF of 1.0×10^{-3} kg CH_4/mmBtu. For comparative purposes, a heat input-based EF was calculated based on both stack-only emissions (should be comparable to the GHGRP default EF) and facility-scale emissions from this study (eq 5). Again, the projected annual CH_4 emissions estimates derived from the AMPD CO_2 data was used.

Throughput-Based CH_4 EFs at Refineries. To calculate a throughput-based EF for the three refineries, projected annual CH_4 emission rates based on annual CO_2 emissions reported to the GHGRP were calculated by multiplying $EF_{facility}$ [kg CH_4/kg CO_2] by annual CO_2 emissions [kg CO_2/year] and then converting the resulting CH_4 emission rate [kg CH_4/year] to kg CH_4/h. This hourly emission rate was divided by the hourly throughput [barrels/h] of the specific refinery, determined from 2015 annual throughput data from www.eia.gov, to give a throughput-based EF [kg CH_4/barrel].

$$\text{throughput}\left[\frac{\text{kg CH}_4}{\text{h}}\right] = \text{heat input}[\text{mmBtu}] \times \frac{1\text{MCF}}{1.02\text{mmBtu}} \times \frac{19.2\text{ kg CH}_4}{1\text{MCF}} \times 0.95 \tag{3}$$

$$\text{loss rate}[\%] = \frac{\text{projected annual CH}_4\text{ emissions from AMPD CO}_2\text{ data [kg CH}_4\text{/h]}}{\text{throughput[kg CH}_4\text{/h]}} \tag{4}$$

$$EF_{\text{throughput-based}}\left[\frac{\text{kg CH}_4}{\text{mmBtu}}\right] = \frac{\text{projected annual CH}_4\text{ emissions from AMPD CO}_2\text{ data [kg CH}_4\text{/h]}}{\text{average hourly heat input [mmBtu/h]}} \tag{5}$$

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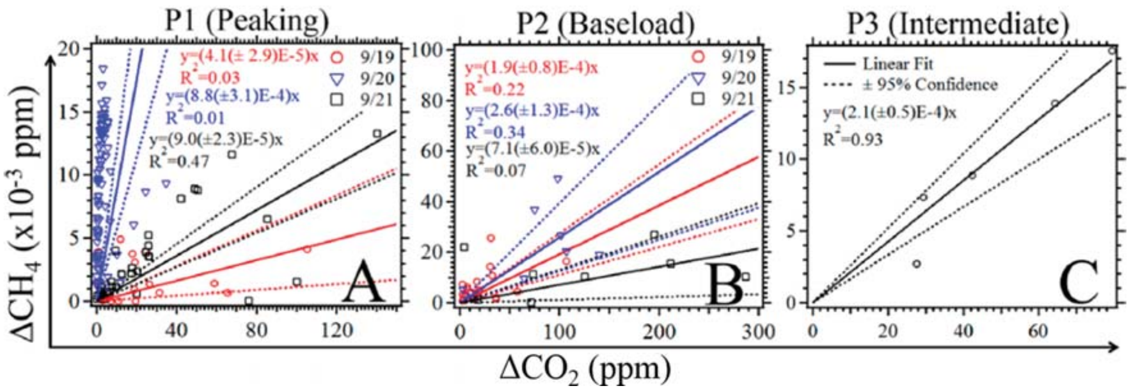

Figure 1. Determination of Stack-Based Emission Factors at the NGPPs for (A) P1 on 9/19, 9/20, and 9/21, (B) P2 on 9/19, 9/20, and 9/21, and (C) P3 on 9/25. Regressions were performed separately for each day of measurement according to the provided figure legends. Solid lines indicate the best fit line and dotted lines represent the 95% confidence bounds.

RESULTS AND DISCUSSION

Mass Balance Quantification of Facility CH_4 and CO_2 Emissions. To understand the magnitude of CH_4 emissions from NGPPs and refineries, six flights were performed at three NGPPs and three refineries, resulting in seven usable mass balance flight experiments at the six sites, and stack emission sampling on seven occasions at the three NGPPs (SI Table S1). Meteorological conditions for each flight (SI Table S1) and individual flight paths (SI Figure S2) are provided. The three NGPPs were selected to represent different power plant classifications, (SI Table S2), including peaking (P1), baseload (P2), and intermediate (P3), because the magnitude of emissions from NGPPs may relate to differences in natural gas throughput, and the operational costs of different electric generating units (EGUs) are a driving factor in understanding which power plants are dispatched to satisfy the temporally changing demand for electricity (www.eia.gov). For instance, baseload power plants operate continuously year-round, and generate the required amount, or "baseload", of electricity to match the average load. During periods when energy loads increase, for example, during heat waves or mid-day in summer, peaking facilities are invoked to generate the additional power needed. Alternatively, intermediate or "load-following" plants supplement the power generated by baseload facilities while adjusting their output to correlate with the hourly demand for electricity. Therefore, understanding the differences in emissions from these three power plant classifications will encourage improvements in mitigation strategies as they relate to specific operational conditions. For the peaking facility, P1, we performed mass-balance measurements on Sunday (9/20) and Monday (9/21) during peak hours of electrical demand. Two mass-balance experiments were attempted at the baseload facility, P2 (9/20 and 9/21), however, only the 9/21 experiment was successful due to poor winds on 9/20. Emissions from the intermediate facility, P3, were measured once, during a period of high energy demand.

The three refineries were successfully sampled once each and were selected based on both their proximity to Purdue University and their representation of small- (R1) to large- (R3) scale refineries based on processing capacity (SI Table S3). A second measurement was performed at R1 (10/1), however, interfering emissions from a nearby unknown source prevented determination of an emission rate. Final calculated CH_4 and CO_2 emission rates (kg/h) for each facility are shown in Table 1 to 95% confidence ($\pm 2\sigma$). It is important to note that variable winds during the P1 (9/20) and P2 (9/21) experiments contributed to high uncertainties in the emission estimates.

To increase understanding of the sources and magnitudes of U.S. greenhouse gas emissions, the Environmental Protection Agency (EPA) implemented the Greenhouse Gas Reporting Program (GHGRP) in 2009 with the goal of collecting and organizing self-reported emissions data from NGPPs and refineries emitting greater than 25 000 t of CO_2 equivalent per year (i.e., ~3000 kg CO_2/h, or ~110 kg CH_4/h, etc.). Additionally, the EPA also requires NGPPs to install continuous emissions monitoring systems (CEMS) that measure gas concentrations (e.g., CO_2) continuously from combustion exhaust stacks and report hourly emissions to the Air Markets Program Data (AMPD).[13] For comparative purposes, the 2014 GHGRP annual facility-specific CO_2 and CH_4 emission estimates and the 2015 AMPD[13] CO_2 emissions during the time of our actual measurements are also provided in Table 1. CH_4 emissions data are not available from the AMPD.

For the NGPPs, quantified CO_2 emission rates at P1 and P2 were not statistically different from their emissions reported to the 2014 GHGRP and the AMPD, and calculated CO_2 emissions at P3 were a factor of 1.8 larger than the GHGRP and a factor of 1.9 smaller than the AMPD. However, for all NGPPs, measured CH_4 emission rates were significantly larger than their respective 2014 GHGRP estimates, by factors of 60 (P1, 9/20), 38 (P1, 9/21), 21 (P2), and 120 (P3). Notably, there was a correlation between power plant operating capacity during the time of measurement and CH_4 emission rate (R^2 = 0.85) and CO_2 emission rate (R^2 = 0.65) (SI Figure S3). Significantly larger CH_4 emission rates were also observed at all three refineries when compared to their respective 2014 GHGRP emission estimates, by factors of 90 (R1), 11 (R2), and 31 (R3). Furthermore, measured CO_2 emissions were also larger at all three refineries compared to the 2014 GHGRP, by factors of 78 (R1), 5 (R2), and 2 (R3), although to a lesser extent than for CH_4. Refinery throughput (SI Table S3) was strongly correlated with CO_2 emissions (R^2 = 0.95) and CH_4 emissions (R^2 = 0.73) (SI Figure S3).

Facility-scale CH_4:CO_2 emission factors ($EF_{facility}$) for the NGPPs and refineries, and stack-based CH_4:CO_2 emission factors (EF_{stack}) for the NGPPs were calculated as described and are provided in Table 1. Markedly, in all cases for the NGPPs, the value of EF_{stack} was 6–29% that of $EF_{facility}$, indicating that emissions sampling from only stacks will likely fail to account for the full scale of emissions from a facility. Furthermore, stack

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Table 2. Projected Annual CH_4 Emission Rates using CH_4:CO_2 EF_{fac} and EF_{stack} and Reported CO_2 Emissions to the GHGRP and AMPD

site	date	2014 GHGRP CO_2[a] ER (kg/h)	2014 GHGRP projected CH_4 ER (kg/h) ± 1σ		2015 AMPD CO_2[b] ER (kg/h)	2015 AMPD projected CH_4 ER (kg/h) ± 1σ	
			$EF_{facility}$-derived	EF_{Stack}-derived		$EF_{facility}$-derived	EF_{Stack}-derived
P1	9/19		[c]	2 ± 1		[c]	2 ± 1
	9/20	115 491	130 ± 70	37 ± 7	104 531	110 ± 64	33 ± 6
	9/21		51 ± 13	4 ± 0		46 ± 12	3 ± 0
P2	9/19		[c]	14 ± 3		[c]	14 ± 3
	9/20	196 919	[c]	19 ± 5	199 758	[c]	19 ± 5
	9/21		69 ± 45	5 ± 2		70 ± 46	5 ± 2
P3	9/25	104 613	140 ± 28	8 ± 1	285 001	380 ± 77	22 ± 2
R1	7/31	23 034	5 ± 2	–	–	–	–
R2	7/31	188 628	100 ± 36	–	–	–	–
R3	9/25	282 959	510 ± 110	–	–	–	–

[a]EPA's 2014 Greenhouse Gas Reporting Program (GHGRP) annual facility-specific CO_2 emission rate estimate, [b]EPA's 2015 Annual Air Markets Program Data (AMPD) facility-specific CO_2 emission rate estimate, [c]Sampled stack emissions only

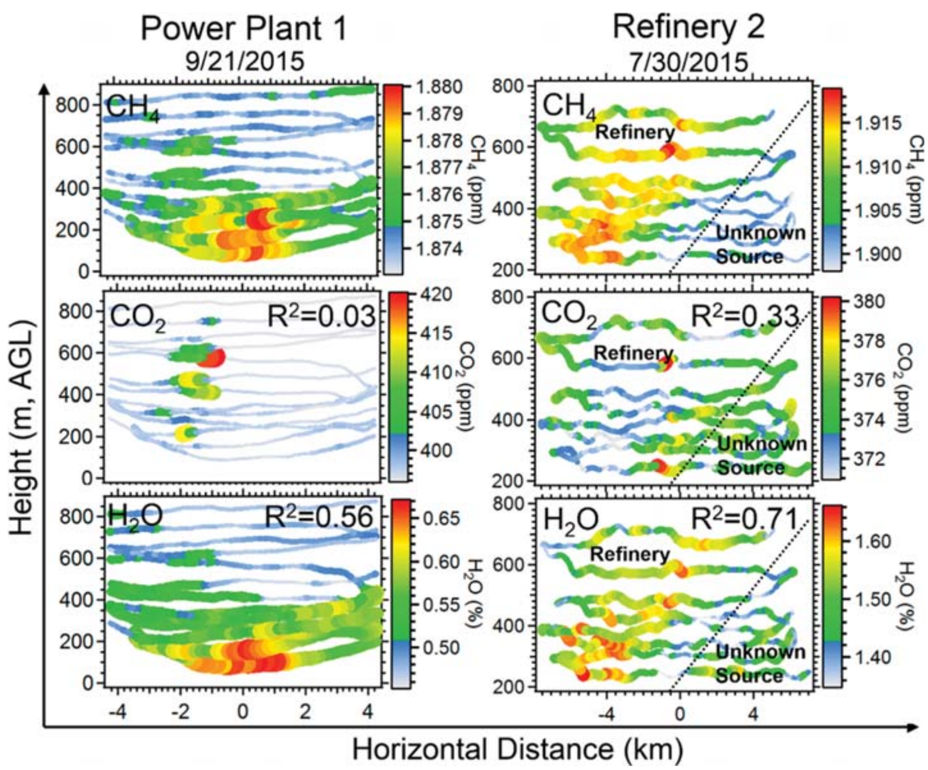

Figure 2. Co-location of CH_4, CO_2, and H_2O Emissions at Power Plants and Refineries. Using Power Plant 1 (P1) and Refinery 2 (R2) as examples, horizontal distributions of raw CH_4 (ppm), CO_2 (ppm), and H_2O (%) concentrations are shown versus height (m, above ground level). Analyte concentration is depicted by color (see color scales) and line width, with warmer colors and thicker line width corresponding to larger analyte concentration. The black dashed lines shown in the Refinery 2 (R2) graphs mark the separation of emissions from different facilities. R^2 values obtained from linear regressions of ΔCO_2:ΔCH_4 and ΔH_2O:ΔCH_4 (SI Figure S5) are displayed in the CO_2 and H_2O concentration panels, respectively.

emissions were sampled three times at both P1 and P2, permitting assessment of temporal variability in the magnitude of EF_{stack}. Regression analysis of total stack plume points for all days of measurement at P1–P3 are shown in Figure 1, organized by day of measurement according to the figure legend. Solid lines represent the best fit and dashed lines represent the 95% confidence bounds, with the slope equaling EF_{stack} (Table 1). At both P1 and P2, daily changes in EF_{stack} did occur by up to a factor of 21 and 4, respectively, despite there being no start-ups or shut-downs of electric generating units between measurements. Stacks at P1 and P2 were sampled at roughly the same time each day (P1: ~12:00 PM; P2: ~5:00 PM), on Saturday (9/19), Sunday (9/20) and Monday (9/21). Notably, despite P1 exhibiting the largest EF_{stack} on Sunday 9/20 compared to other measurements, electrical demand was higher during the measurement made on Monday 9/21, according to heat input data reported by the AMPD.

Due to temporal fluctuations in facility emissions caused by variations in facility operations, the hourly emission rates calculated here cannot be directly extrapolated to estimate annual facility emissions. However, this variability can be accounted for indirectly by applying the calculated CH_4:CO_2 EFs to annual CO_2 emissions reported to inventories to estimate a proportional CH_4 emission rate based on the known quantity of

CO_2 emitted. Therefore, annual average hourly CH_4 emission rates [kg/h] per facility were estimated by multiplying $EF_{facility}$ (NGPPs, refineries) and EF_{stack} (NGPPs) by annual reported CO_2 emission data, first converted from annual to hourly emission rates, from the GHGRP (NGPPs, refineries) and the AMPD (NGPPs), and are shown ($\pm 1\ \sigma$) in Table 2. These GHGRP-derived hourly CH_4 emission rates can then be extrapolated to estimate the annual atmospheric CH_4 emissions for each facility, which are 800 ± 400 Mg (P1), 600 ± 400 Mg (P2), 1200 ± 200 Mg (P3), 40 ± 20 Mg (R1), 900 ± 300 Mg (R2), and 4500 ± 1000 Mg (R3). Also, for cases at the NGPPs where both EF_{stack} and $EF_{facility}$ were available to estimate annual emissions, $EF_{facility}$-derived emissions were larger than EF_{stack}-derived emissions by factors of 3−22 for both the GHGRP- and AMPD-based projections, again indicating that emissions monitoring methods that only sample stack emissions may significantly underestimate facility emissions.

Assessment of Combustion- and Non-Combustion-Related CH_4 Emissions. To further investigate if CH_4 emissions were related to combustion or noncombustion processes, the correlation of enhanced concentrations of CH_4 with CO_2 and H_2O was assessed along the flight transects. If natural gas undergoes incomplete combustion, uncombusted CH_4 will exist in the presence of combustion products, for example, CO_2 and H_2O. Therefore, CH_4 concentration enhancements along the flight path that are colocated with CO_2 are likely to be uncombusted CH_4 from combustion processes, and at NGPPs, these emissions would originate from exhaust stacks. Alternatively, if CH_4 concentration enhancements are not colocated with CO_2, they likely originate from noncombustion-related equipment on the facility. Figure 2 shows greenhouse gas concentration data along the flight transects for two representative facilities, presented as a horizontal distribution of analyte (CH_4, CO_2, H_2O) concentrations versus altitude. Similar figures for all flights are provided in SI Figure S4. Subsequently, the concentration enhancement above background for all analytes (CH_4, CO_2, H_2O) was calculated along each transect by subtracting an altitude-dependent background and back trajectory analysis was used to spatially segregate emissions from the facility of interest from nearby unknown sources. Linear regressions of these facility-specific concentration enhancements, ΔCO_2:ΔCH_4 and ΔH_2O:ΔCH_4, along the flight paths were then performed, with two representative examples shown in Figure 3 and regressions for all flights provided in SI Figure S5.

For all three NGPPs, CH_4 enhancements were more strongly correlated with H_2O enhancements (R^2_{avg} = 0.60), than with CO_2 enhancements (R^2_{avg} = 0.15) downwind of the facilities. The separation of CH_4 emissions from each facility that was observed across the flight path is likely due to variation in the temperature of emissions from different sources within the facilities, which could result in differences in buoyancy of emissions. For instance, if NGPP CH_4 emissions were primarily from high temperature, combustion-related sources (e.g., stacks), then colocation of CH_4 and CO_2 would be expected (this was observed at all three NGPPs in the stack emissions, although to a lesser extent than for colocation of CH_4 and H_2O). Supporting this observation, for all three NGPPs, EF_{stack} was significantly lower than $EF_{facility}$ on all days, further indicating that the majority of NGPP CH_4 emissions are not emitted from stacks. The NGPPs in this study operate on highly efficient combined-cycle systems, which use both natural gas and steam turbines to generate 50−60% more energy than a gas turbine alone by capture and reuse of exhaust heat from the gas turbine into a heat recovery steam generator. Therefore, potential sources of CH_4 emissions at NGPPs include uncombusted CH_4 from stack exhaust (e.g., colocated with CO_2 and H_2O), and fugitive leaks from the facility equipment, including compressors, steam turbines, steam boilers, and condensers (e.g., colocated with H_2O). We can rule out the possibility that our results are caused by dispersion differences of stack-emitted H_2O, CH_4, and CO_2 due to our observation that combustion products (CO_2 and H_2O) and uncombusted CH_4 were colocated at all three NGPPs, with a separate, distinct grouping of CH_4 and H_2O emissions also present, that were not correlated with CO_2. Therefore, it is important to consider nonstack-related emissions at NGPPs when developing facility-scale CH_4 emissions monitoring methods.

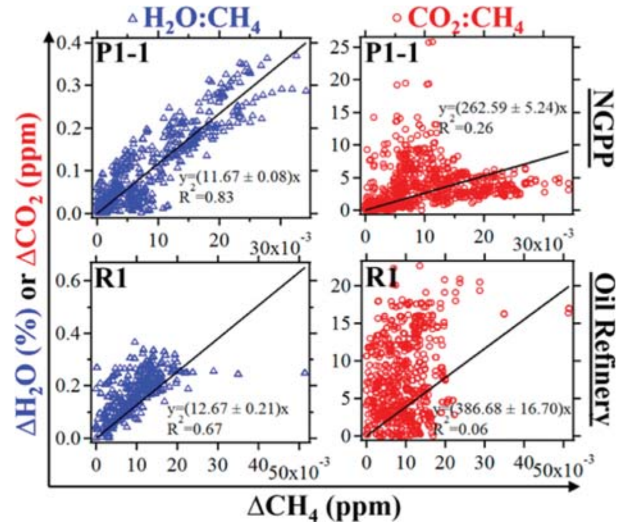

Figure 3. Regression Analysis of H_2O and CO_2 Enhancements versus CH_4 Enhancements Along the Flight Transects. Linear regressions were performed using transect concentration enhancement data from Figure 2 and SI Figure S4, with measurements made at P1 on 9/20 and R1 on 7/31 shown here as examples. Blue triangles (*y*-axis: ΔH_2O; *x*-axis: ΔCH_4) and red circles (*y*-axis: ΔCO_2; *x*-axis: ΔCH_4) show individual data points. Best fit line (black line) with equation and R^2 values are shown. Units of the slopes are [ppm/ppm] for the CO_2:CH_4 curves and [%/ppm] for the H_2O:CH_4 curves.

The three refineries demonstrated similar results, with CH_4 enhancements being more strongly correlated with H_2O enhancements (R^2_{avg} = 0.71) than with CO_2 enhancements (R^2_{avg} = 0.29), indicating that noncombustion-related CH_4 emissions may be a significant source of total CH_4 emissions at refineries. The equipment involved in petroleum refining, including furnace heaters, hydrogen generation units, gas turbines, and condensers, can be powered by various fuel types, including natural gas. Potentially, refineries may be a source of CH_4 emissions due to increased use of natural gas to power their utilities. Additionally, CH_4 is a minor component of crude oil, and therefore, a product of fractional distillation, and is a product of catalytic cracking. Possible sources of CH_4 and H_2O at refineries therefore include steam boilers, compressor engines, storage vessels, process heaters, process furnaces, and distillation towers. Therefore, inclusion of noncombustion-related CH_4 losses in EF calculations would help encompass a broader range of potential emission sources at these facilities and improve annual emissions estimates in U.S. inventories.

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Table 3. Facility Throughput Estimates, NGPP CH_4 Loss Rates, and Throughput-Based EFs

			CH_4 loss rate (%)		throughput-based EF[b]	
site	date	throughput estimate[a]	facility	stack	facility	stack
P1	9/19	41 000	–	0.01 (±0.00)	–	8.7 (±4.3) × 10^{-4}
	9/20	26 000	0.42 (±0.24)	0.13 (±0.02)	5.0 (±2.9) × 10^{-2}	1.5 (±0.3) × 10^{-2}
	9/21	30 000	0.15 (±0.04)	0.01 (±0.00)	2.0 (±0.6) × 10^{-2}	1.3 (±0.1) × 10^{-3}
	avg.		0.29 (±0.14)	0.05 (±0.01)	3.5 (±1.7) × 10^{-2}	5.7 (±1.6) × 10^{-3}
P2	9/19	10 000	–	0.14 (±0.03)	–	3.7 (±0.8) × 10^{-3}
	9/20	56 000	–	0.03 (±0.01)	–	5.1 (±1.3) × 10^{-3}
	9/21	70 000	0.10 (±0.01)	0.01 (±0.00)	1.6 (±1.1) × 10^{-2}	1.2 (±0.5) × 10^{-3}
	avg.			0.06 (±0.02)		3.3 (±0.9) × 10^{-3}
P3	9/25	150 000	0.25 (±0.05)	0.02 (±0.00)	6.1 (±1.2) × 10^{-2}	3.5 (±0.3) × 10^{-3}
R1	7/31	1130	–	–	4.4 (±1.8) × 10^{-3}	–
R2	7/31	8830	–	–	1.1 (±0.4) × 10^{-2}	–
R3	9/25	9940	–	–	5.1 (±1.1) × 10^{-2}	–

[a]P1−3 [kg CH_4/h]; R1−3 [barrels/h]. [b]P1−3 [kg CH_4/mmBtu]; R1−3 [kg CH_4/barrel].

It is estimated that the climate benefit of NGPPs over coal-fired power plants is contingent on total system CH_4 loss rates being less than 3% of throughput, with climate benefits observed immediately.[4] However, life cycle analyses indicate that CH_4 losses that occur from production to distribution and use must also be considered, which are estimated to equal 1.7% of production.[6] In this study, calculated facility-scale loss rates were less than 0.5% in all cases (Table 3), and so the climate benefit of using natural gas for electricity generation is not compromised given the magnitude of losses at the point of use of the NGPPs in this study, if we assume a supply chain leak rate of 1.7% of production. The percentage of unburned CH_4 emitted from stacks at the three NGPPs (0.01−0.14%) was lower than respective facility-scale losses (0.10−0.42%) in all cases, by factors of 3 (P1, 9/20) 15 (P1, 9/21), 10 (P2, 9/21), and 13 (P3, 9/25), again suggesting that more CH_4 is lost from non-combustion-related equipment than from combustion processes (Table 3). Furthermore, the observation that the majority of NGPP and refinery emissions are from noncombustion-related equipment would support the significant discrepancies between our calculated CH_4 emission rates and those reported to the 2014 GHGRP, which only requires reporting of combustion-related CH_4 emissions. The 2014 GHGRP CH_4 emission rates reported for the three NGPPs were all <4 kg/h.

In 2014, the GHGRP required power plants to calculate and report emissions related to general stationary fuel combustion (GHGRP, subpart C) and electricity generation (subpart D), and combustion-related CH_4 emissions were calculated by operators using a required heat input-based emission factor (EF_{GHGRP}) of 1.0×10^{-3} kg CH_4/mmBtu.[23] To examine the accuracy of EF_{GHGRP}, we used our measured data to derive EF_{stack} [kg/mmBtu] for the NGPP stacks (P1, $N = 3$ days of measurements; P2, $N = 3$; P3, $N = 1$) and $EF_{facility}$ [kg/mmBtu] for the complete facilities (P1, $N = 2$ days of measurements; P2, $N = 1$; P3, $N = 1$) (Table 3). For all measurement days, all three NGPPs' EF_{stack} [kg/mmBtu] values were larger than EF_{GHGRP} by an average factor of 4.4, ranging from 0.9 to 15 times larger. More notably, however, is the difference between the complete facility-derived $EF_{facility}$ [kg/mmBtu] and EF_{GHGRP}, which were factors of 50 (P1, 9/20), 20 (P1, 9/21), 16 (P2), and 61 (P3) times larger than the industry-used EF_{GHGRP}. Therefore, updating heat input-based CH_4 EFs at NGPPs may improve the accuracy of GHGRP data, which policymakers rely on to best understand U.S. CH_4 emission rates and the contributions of individual sources.

In addition to reporting general stationary fuel combustion-related emissions (GHGRP, subpart C), refineries are also required to report CH_4 emissions from asphalt blowing operations, uncontrolled blowdown systems, catalytic cracking and reforming units, delayed coking units, flares, process vents, storage tanks, and equipment leaks (subpart Y).[23] Similar to subpart C, emission estimates reported under subpart Y also are calculated using default EFs which may be outdated and could cause inaccurate estimation of annual emissions. Additionally, emissions may also originate from other types of process equipment, including boilers, process heaters, furnaces, incinerators, and thermal oxidizers. Our results suggest that both CH_4 and CO_2 emissions may be underestimated for these three refineries by the GHGRP. To determine if these results are representative of the full range of operating conditions will require further observations.

NGPPs and Refineries as Contributors to U.S. CH_4 Emissions. We estimate that NGPPs in the U.S. emit 0.46 ± 0.17 Tg CH_4 annually (SI Table S4) by using the average of the heat input-based EFs calculated from this study for the NGPPs [kg CH_4/mmBtu] and annual total heat-inputs for all NGPPs nationwide in 2015 as the activity factor (downloaded from the AMPD). Additionally, using the average of the throughput-based EFs calculated from this study for the refineries [kg CH_4/barrel] and the hourly throughput for all refineries in the U.S. in 2015 based on data from www.eia.gov [barrels/h], we estimate that U.S. refineries emit 0.15 ± 0.05 Tg CH_4 annually (SI Table S4). Combined, NGPPs and refineries are therefore estimated to contribute ~0.61 Tg CH_4 annually to U.S. emissions. By comparison, the EPA estimated that oil and gas operations emitted 9.8 Tg CH_4 in 2014, of which CH_4 emissions from NGPPs (0.01 Tg CH_4) and refineries (0.02 Tg CH_4) were estimated to be negligible by comparison.[6] For comparison, U.S. landfill operations and enteric fermentation processes were estimated to emit 5.9 and 6.6 Tg CH_4, respectively, in 2014. Therefore, consideration of improved emissions monitoring and reporting procedures for NGPPs and refineries would significantly improve U.S. inventory emissions estimates. Note that this is a preliminary estimate and that additional sampling is needed to improve robustness of the estimate. However, total emissions from NGPPs is likely to increase in the future as our reliance on NGPPs increases.

Results from this study indicate that NGPPs and crude oil refineries may be significant contributors to annual CH_4 emissions in the U.S., despite lack of facility emission reporting

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in U.S. inventories. Furthermore, results suggest that the primary source of CH_4 emissions at these facilities may be from noncombustion sources, partially explaining why inventory estimates appear biased low as EFs only consider combustion-related emissions. Future studies should aim to identify the specific emission sources at a larger sampling of these facilities, potentially by use of infrared cameras, and subsequently recalculate more robust EFs that consider these sources. Knowledge of common equipment sources would also help inform improvements in emissions mitigation strategies at these facilities, for example, by replacement of aging and faulty equipment, installation of carbon capture devices, and upgrades to improved control technologies related to the specific emission sources. Furthermore, updating CEMS to include hourly CH_4 emission monitoring would help account for combustion-related CH_4 emissions, which were also underestimated in this study.

While measurements in this study were performed during peak operating hours, emissions during periods of start-up and shut-down may be different. Therefore, future measurements at both NGPPs and refineries should be conducted during the full range of operations to develop more robust EFs for each operating condition. Top-down approaches, such as the aircraft-based mass balance technique described in this study, offer the ability to measure total facility emissions to calculate more comprehensive EFs that account for CH_4 emissions from both combustion- and noncombustion-related processes.

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

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Four tables and six figures (PDF)

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Notes

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■ ABBREVIATIONS

ALAR	Airborne Laboratory for Atmospheric Research
AMPD	Air Markets Program Data
CH_4	methane
CO_2	carbon dioxide
CRDS	cavity ring-down spectrometer
EF	emission factor
EPA	Environmental Protection Agency
GHGRP	GreenHouse Gas Reporting Program
LT	local time
mmBtu	million British thermal units
NGPP	natural gas-fired power plants
NOAA	National Oceanic and Atmospheric Administration
SI	supporting information

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NOTE ADDED AFTER ASAP PUBLICATION

The throughputs for refinery R1 and R3 were inadvertently switched, and this affected the downstream calculated values in the version of this paper published March 6, 2017. The values were corrected in the article and Table S4 of the Supporting Information and reposted on March 10, 2017.

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nature climate change

Untangling the confusion around land carbon science and climate change mitigation policy

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Depletion of ecosystem carbon stocks is a significant source of atmospheric CO_2 and reducing land-based emissions and maintaining land carbon stocks contributes to climate change mitigation. We summarize current understanding about human perturbation of the global carbon cycle, examine three scientific issues and consider implications for the interpretation of international climate change policy decisions, concluding that considering carbon storage on land as a means to 'offset' CO_2 emissions from burning fossil fuels (an idea with wide currency) is scientifically flawed. The capacity of terrestrial ecosystems to store carbon is finite and the current sequestration potential primarily reflects depletion due to past land use. Avoiding emissions from land carbon stocks and refilling depleted stocks reduces atmospheric CO_2 concentration, but the maximum amount of this reduction is equivalent to only a small fraction of potential fossil fuel emissions.

Despite the current level of mitigation effort, global CO_2 emissions continue to increase[1]. In addition to reducing emissions from fossil-fuel burning, the largest CO_2 source globally, mitigation efforts now include reducing what is in aggregate the second largest net source of CO_2 to the atmosphere: namely, carbon emissions from land-use change. Land carbon emissions accounted for about 36% of the anthropogenic CO_2 emitted into the atmosphere from 1850-2000[2], and about 12% of annual global CO_2 emissions from 2000 to 2010[1]. Avoiding and reducing land carbon emissions is therefore an integral part of any comprehensive approach to solving the climate change problem.

Globally, forests store around 300 Pg C (reported range 240–500 Pg C) in living biomass[2,3], equivalent to ~140 ppm of atmospheric CO_2 (atmCO_2; used to denote the concentration of CO_2 in the atmosphere, and although the SI unit for atmCO_2 is μmol mol^{-1}, we have adopted the more familiar unit of ppm). Forests are distributed in both developed and developing countries (Table 1). About half of the world's forests have already been cleared, with 40 million km^2 remaining and around 0.16 million km^2 of forest cleared annually[3]. Only 36% (~14.4 million km^2) of the world's forest is now primary forest[3]. In addition to deforestation, forests have been degraded by land-use activities such as logging and soil disturbance that deplete their organic carbon stocks and emit CO_2. Emissions from forest degradation are poorly quantified globally, but estimates indicate that they increase regional carbon emissions by nearly 50% over deforestation alone[4]. Conserving the world's remaining primary forests would avoid substantial emissions of CO_2. Afforestation and reforestation, moreover, can directly remove CO_2 from the atmosphere — but only up to a point, as we discuss later.

Nations are engaged in negotiations to reduce emissions of CO_2 and other greenhouse gases (GHGs) under the United Nations Framework Convention on Climate Change (UNFCCC). Developed countries that are signatories to the Kyoto Protocol (ratified by 37 countries and the European Union) committed themselves to a target of reducing their emissions of GHGs from 2008–2012, relative to 1990 levels. The target reduction was based on emissions from fossil fuels and industry, but removals by the land sector could be counted towards meeting the target. The Clean Development Mechanism under the Kyoto Protocol allowed for developed countries to offset fossil fuel emissions through, among other things, planting trees in developing countries. Similar kinds of offset project are allowed through the Joint Implementation mechanisms between developed countries. The extension or successor to the Kyoto Protocol is now being negotiated. There are parallel negotiations underway on the development of policies for Reducing Emissions from Deforestation and Degradation (REDD) — a voluntary scheme to mitigate land carbon emissions from developing countries.

Negotiated policy decisions involve political compromises to accommodate national interests. So far these decisions have fallen short of what will be necessary if atmCO_2 is to be stabilized at a level that avoids major climate change[5]. Furthermore, there is the potential for perverse outcomes whereby mitigation efforts not only fail to reduce atmCO_2, but even have negative impacts — either causing atmCO_2 to increase or adversely affecting other landscape values, such as biodiversity. Perverse outcomes can result from a gap between land carbon policy decisions and scientific understanding of what is required for successful mitigation: that is, from confusion around land carbon science.

In this Perspective we clarify some well-established fundamentals of the global carbon cycle that are frequently either misunderstood, or seemingly overlooked. This information provides the scientific context for considering the potential of land-based mitigation and to what extent it can be legitimately considered an 'offset' for fossil fuel CO_2 emissions. We do not advocate any particular policy, but we do draw attention to some proposed approaches that are likely to be ineffective, or worse.

Human perturbation of the global carbon cycle

The global carbon cycle is the subject of considerable confusion among non-specialists. A clear understanding of how humans have

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perturbed the cycle's natural stocks and flows of carbon is essential background to clarifying key scientific issues and ensuring effective policies.

Figure 1 illustrates changes in the primary stocks of the global carbon cycle as the result of human activity in three stylized time periods: the pre-agricultural era (>8 000 yr BP; Fig. 1a); the pre-industrial era (8 000 yr BP–1850, Fig. 1b); and the contemporary era (>1850–present day; Fig. 1c). These correspond with major phase shifts in the magnitude of the human environmental footprint in terms of land clearing and use of fossil fuels. Figure 1a shows that in the pre-agricultural era there was no human use of fossil fuel and relatively minimal depletion of land carbon due to human land use. Figure 1b and 1c show the impact of human activity on the primary stocks. The sources and calculations for the values in Fig. 1 are provided in Table 2.

During the pre-industrial era, land carbon began to be depleted (white segment of land carbon stocks) leading to an increase in the atmospheric carbon stock, with some of this carbon dissolving into the ocean stock (as indicated by the green segments). In the contemporary era humans began mining fossil fuel and burning it as a source of energy, as well as engaging in accelerated land clearance. Both activities have resulted in CO_2 emissions and a rapid and significant increase in the atmospheric carbon stock. A portion of the anthropogenic emissions added to the atmosphere is concurrently taken up by plants, and a fraction is dissolved into the ocean stock. This effect is illustrated in Fig. 1c by the segments of black carbon in the land and ocean stocks.

Figure 1d illustrates the hypothetical case of cleared land being largely returned to its pre-agricultural carbon stock levels. The amount of atmospheric carbon that potentially can be stored in the land buffer is, to first order, limited to the amount of carbon that was depleted from previous land use. The black segment signifies that an extra, modest amount of fossil fuel emissions could be stored as the result of the so-called CO_2 fertilization effect discussed below.

The lifetime of CO_2 in the atmosphere

A recurrent, serious misunderstanding is that the residence time in the atmosphere of a unit of CO_2 emitted from fossil fuel burning is quite short, on the order of a century. The First Assessment Report[6] of the Intergovernmental Panel on Climate Change (IPCC) incorrectly stated the 'lifetime' of CO_2 to be ~120 years. Many commentators since have assumed it to be about 100 years. They have probably been encouraged in this view by the use of a 100-year timeframe for the calculation of 'global warming potentials' (GWP, expressed relative to CO_2) for greenhouse gases with different lifetimes. However, it has long been recognized that any single number for the CO_2 lifetime conceals more than it reveals. CO_2 is taken up from the atmosphere by several distinct processes that have hugely different time constants[7,8]. Part of it is taken up by the land, and part dissolves in the ocean surface and mixes to the deep ocean. About 60% is removed from the atmosphere on a time scale of 100 years but it takes a very long time to remove the remaining fraction. A 'pulse' or unit of CO_2 emitted to the atmosphere is only fully removed from the atmosphere so that it no longer interacts with the climate system when it has completely dissolved in the deep ocean — a process requiring the concurrent dissolution of carbonate from ocean sediments (about 5,000 to 10,000 years) and enhanced weathering of silicate rocks (around 100,000 years). Modelling by Archer and colleagues indicated that 20–35% of the CO_2 emitted will still be in the atmosphere after 2–20 millennia. Tracing the history of the misunderstanding of CO_2 lifetimes, they commented that "…the result has been an erroneous conclusion, throughout much of the popular treatment of the issue of climate change, that global warming will be a century-timescale phenomenon"[9].

The reality is that for all practical purposes, fossil fuel CO_2 emission is irreversible[10]. Any eventual stable atmCO_2 will be dictated by total accumulated emissions over the preceding centuries[11] and not by the contemporaneous balance of emissions and removals. In this respect CO_2 behaves quite differently from the other major so-called long-lived GHGs — methane and nitrous oxide — which have atmospheric lifetimes in the order of 10 years and 100 years, respectively. This difference implies an important caveat for the use of GWP. Reduced emissions of nitrous oxide or methane might be substituted for reduced emissions of an 'equivalent' amount of CO_2. But the effects of the emitted CO_2 will continue to be felt for thousands of years, long after the effects of the reduced emissions of the other gases have disappeared.

Table 1 | Top 10 countries for total area of forest and other wooded land (see Annex Table 3, ref. 3).

Rank	Country	Forest (1,000 ha)	Country	Other wooded land (1,000 ha)
1	Russian Federation	809,090	Australia	135,367
2	Brazil	519,522	China	102,012
3	Canada	310,134	Canada	91,951
4	USA	304,022	Russian Federation	73,220
5	China	206,861	Argentina	61,471
6	DRC	154,135	Sudan	50,224
7	Australia	149,300	Ethiopia	44,650
8	Indonesia	94,432	Brazil	43,772
9	Sudan	69,949	Botswana	34,791
10	India	68,434	Afghanistan	29,471

The limited capacity of land carbon stocks

Land carbon plays an important role in the stocks and flows of the global carbon cycle, but the magnitude is limited and it has particular characteristics which contrast with the different qualities of the other main categories of carbon stocks (fossil fuel, atmosphere and ocean). The fossil fuel carbon stock was built up very slowly over millions of years and does not de-gas into the atmosphere at any significant rate. Emissions from this stock in the contemporary era constitute a one-way flow, which is a direct result of human activity. Carbon is stored in the other three major categories of stocks in different forms (on land as biomass and soil organic carbon, in the atmosphere as CO_2 gas and in the ocean primarily as dissolved inorganic carbon) and both the land and ocean carbon stocks naturally exchange with the atmospheric stock.

The potential size of the land carbon stock is determined chiefly by climate, and modified locally by substrate and topography, and the effects these have on plant growth[12]. The capacity of the land to remove atmospheric carbon and store it in vegetation and soil is limited to the amount previously depleted by land use. It has been estimated that if all the carbon so far released by land-use changes (mainly deforestation) could be restored through reforestation this would reduce atmCO_2 at the end of the century by 40–70 ppm. Conversely, complete global deforestation over the same time frame would increase atmospheric concentrations by about 130–290 ppm[13]. In comparison, the projected range of atmCO_2 in 2100, under a range of fossil fuel emissions scenarios developed for the IPCC, is 170–600 ppm above 2000 levels[14].These estimates highlight the very modest scope for reforestation to reduce atmCO_2 compared with both the magnitude of fossil fuel CO_2 emissions and emissions from derorestation and degradation. Moreover, complete reforestation of previously cleared land is an implausible scenario due to competing land uses — especially from food production

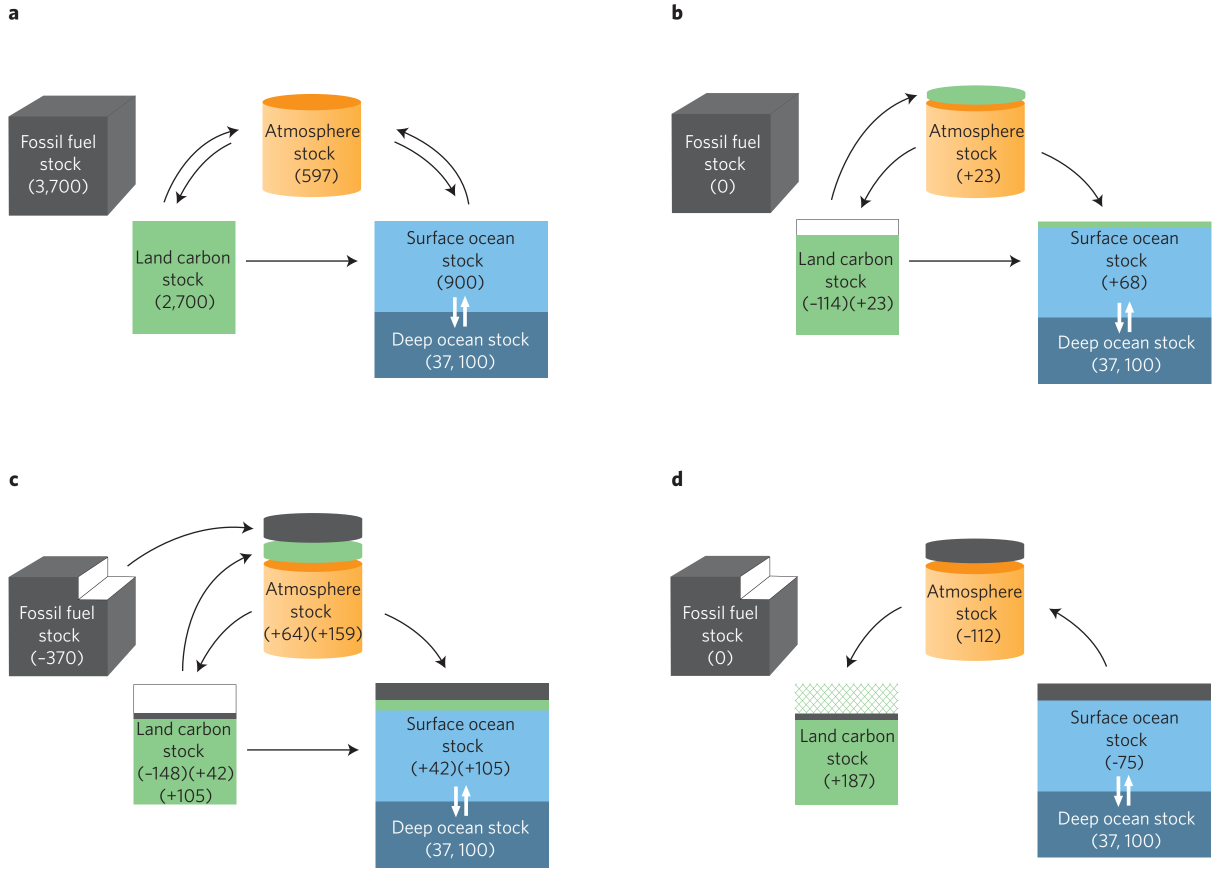

Figure 1 | Changes in the primary stocks of the global carbon cycle. a–c, A stylized illustration of the impact of human activity on the primary stocks over three time periods: the pre-agricultural era (>8,000 yr BP; **a**); pre-industrial era (8,000 yr BP to 1850; **b**); and contemporary era (1850 to the present day; **c**). The objects (cylinders and rectangles) represent the primary stocks of carbon in the major reservoirs of the global carbon cycle (fossil fuel, atmosphere, land, surface ocean and deep ocean) but are not drawn to scale. **d**, The hypothetical and unachievable case of "refilling" the land stock, that is, if all previously cleared land being returned to its pre-agricultural carbon stock with zero continuing fossil fuel emissions. Numbers in parentheses (Pg C) are indicative estimates of the carbon stocks (**a**) and changes in carbon stocks (**b**–**d**). The arrows represent the direction of carbon flows (fluxes) between stocks over the era, with arrows in panel **a** representing the natural background carbon cycle, and arrows in the other panels indicating the impact of anthropogenic change. Natural processes (as shown in panel **a**) involve two-way exchanges of carbon between the atmosphere–ocean (on the order of 70 Pg C yr^{-1}) and atmosphere–land (around 120 Pg C yr^{-1}) with a small natural hydrological flux of carbon discharged from rivers into oceans of 0.8 Pg C yr^{-1} (not shown are the very small sources due to volcanic activity and sinks due to weathering)[42]. The anthropogenic changes due to land use change and burning fossil fuels are also illustrated using colour coded slices (also not drawn to scale). These changes reflect processes that can be considered in general terms as operating over two timescales: on the order of a one to a few 1,000 years about 20% of the emitted CO_2 stays in the atmosphere[44,9], 60% is taken up by the ocean and 20% by land[44]; on the time scale of 100 years 43% of emissions remain in the atmosphere[45,46] with the rest taken up roughly equally between the land and ocean[46]. We use the simplified assumption that as atmospheric CO_2 is reduced, the ocean would 'outgas' CO_2, and the land would also outgas the carbon uptake due to the CO_2 fertilization effect, based on processes operating over the 100-yr timescale. The land retains the C uptake from fossil fuel emissions. Even if the unachievable was accomplished, after 100 years, there would still be an extra 134 Pg C in the atmosphere compared with the pre-agricultural era due to fossil fuel emissions. The estimates are based on sources and calculations in Table 2.

and the need to feed a human population predicted to surpass nine billion by 2050[15] — along with projected demand for land to produce transport biofuel of 0.3–0.5 million km^2 by 2030[16]. And even under this impossible scenario, land degradation means that some of the land carbon stock cannot be re-filled.

The difference between stocks and sinks

Land carbon scientists are clear on the difference between land carbon stocks and sinks, however policymakers and the interested citizen can be excused for not understanding (or sometimes forgetting) the distinction. As used in carbon cycle science, the term 'sink' always implies a net removal of carbon from the atmosphere — in other words, a net flux of carbon into the ecosystem. There is a persistent risk of confusion between a stock (in units of mass, g C) and a flux (in units of mass/time, g C yr^{-1}). Both the ocean and the land are indeed taking up part of the CO_2 that is emitted by human activities, so they do constitute sinks. But this uptake is a transient effect as discussed below.

The land carbon stock can be described as a 'buffer' by analogy with the term used in computer science to describe a device which temporarily stores data. The impact of land use activity is appropriately reported or accounted for as a change in stock over a given time period, that is, a depletion or re-filling of the buffer. When a forest is re-planted, at first it functions as a sink — with the net uptake of CO_2 due to photosynthesis being greater than respiration — and carbon is accumulating in woody biomass and soil. Over time, the net sink rate declines as the growth rate decreases relative to respiration rates. If the forest is allowed to develop into an ecologically mature state, the carbon stock approaches a dynamic equilibrium with prevailing environmental conditions, where

Table 2 | The sources and calculations for estimated changes of carbon stocks in Fig. 1

Stock	Pg C	Sources and calculations (references given in parentheses)
(a) Pre-agricultural		
Fossil fuel	3,700	Fig. 7.3 (42)
Land	2,700	Fig. 7.3 (42)
Atmosphere	597	Fig. 7.3 (42)
Shallow ocean	900	Fig. 7.3 (42)
Deep ocean	37,100	Fig. 7.3 (42)
(b) Pre-industrial (change from pre-agricultural)		
Fossil fuel	0	
Land	−114	Emissions from land clearance (43)
	23	20% taken up by land due to CO_2 fertilization effect, 1,000-year timescale (44)
Atmosphere	23	20% of emissions remain in atmosphere, 1,000-year timescale (9,44)
Ocean	68	60% taken up by ocean, 1,000-year timescale (44)
(c) Contemporary (change from pre-industrial)		
Fossil fuel	−370	IPCC Fossil fuel emissions (42)
Land	−148	Emissions from land clearance (43)
	42	28.5% of land carbon emissions taken up by land due to CO_2 fertilization effect, 100-year timescale (46)
	105	28.5% of land carbon emissions taken up by land due to CO_2 fertilization effect, 100-year timescale (46)
Atmosphere	64	43% of land carbon emissions remain in atmosphere 100-year timescale (45,46)
	159	43% of fossil fuel carbon emissions remain in atmosphere 100-year timescale (45,46)
Ocean	42	28.5% of land carbon emissions taken up by ocean, 100-year timescale (46)
	105	28.5% of land carbon emissions taken up by ocean, 100-year timescale (46)
(d) Hypothetical restoration of the land carbon buffer (change from contemporary)		
Fossil fuel	0	
Land	187	+262 restored to the land (114+148), minus 28.5% reduced CO_2 fertilization effect (−75), 100-year timescale
Atmosphere	−112	−262 removed by land restoration, +75 out-gassed from ocean, +75 reduced CO_2 fertilization effect on land
Ocean	−75	Response of ocean to lowered atmospheric CO_2 is out-gassing of 28.5% of 262, 100-yr timescale

respiration approximately balances photosynthesis. At this point, the depleted land carbon stock has been refilled and the sink function has gone. The mitigation value of the ecosystem resides in maintenance of the stored carbon stock.

At present some forests have carbon sequestration potential due to depletion of carbon stocks from past land use[17]. Reforestation of previously cleared or logged land (especially in Europe, the USA and China), together with deforestation and degradation (especially, but not exclusively, in tropical developing countries), are all included in the calculation of net emissions noted above from land use change.

The land and ocean are sinks, and globally they removed an estimated 56% of all CO_2 emitted from human activities during the period 1958–2010, each sink in roughly equal proportion[18]. Although land-use change is a source of emissions, the land as a whole is functioning as a sink at present. This land sink reflects the natural response of ecosystems to the influence of environmental change, which is now leading to a net uptake of CO_2 due to several factors. Rising atmCO_2 leads to a boost in plant productivity (the CO_2 fertilization effect), whereby the increase in net primary production outpaces the increase in respiration of soil carbon stocks[19,20]. Experimental evidence has shown that net primary productivity of temperate forests increases by around 23% in response to a 200 ppm increase in CO_2 (that is, when grown in atmCO_2 of 550 ppm)[21]. However, the effect varies geographically[22], is constrained (to an uncertain degree) by nitrogen availability[23] and depends on CO_2 continuing to increase. If CO_2 were stabilized, this effect would disappear probably after a lag of a few decades. The practical effect of an increase in atmospheric CO_2 on potential ecosystem carbon stocks is a modest increase in the size of the buffer that could be refilled.

Ecologically mature (>200 years) and old-growth forests aged up to 800 years can continue to function as sinks. Old-growth tropical forests accumulate around 5 Mg C km^{-2} yr^{-1} in living biomass, which could be yielding a carbon sink of 1.3 Pg C yr^{-1} (0.8–1.6 Pg C yr^{-1}) across all tropical forests[24,25]. We reiterate, however, that the mitigation value of tropical forests — and old-growth forests in general — does not lie in their present, transient function as carbon sinks. In terms of carbon mitigation policy, the primary reason to conserve forests is the carbon stocks they contain. The idea that replacing primary forests by plantations will 'create sinks' and thereby be positive for climate mitigation is incorrect, as it fails to account for the loss of carbon stock from the primary forest[26]. Furthermore, plantation forests store less carbon than the pre-existing natural primary forest, secondary (regenerating) natural forests or a primary forest under the same environmental conditions[27–30].

Climate change may increase potential carbon stocks in some regions: for example, through increased rainfall and/or decreased potential evaporation where plant growth is limited by water availability, and through enhancement of the growing season in northern temperate regions due to increases in temperature. But conversely, increasing aridity in other regions is likely to reduce plant growth through drying or heat stress[31], and to increase the likelihood that forest areas are subject to wildfire, which can reduce the long-term carbon carrying capacity of landscapes[32]. Hence, there are competing processes resulting in changes in the potential land carbon stock. An analysis based on 13 coupled climate–carbon cycle models pointed to future climate change reducing the efficiency of the Earth system in absorbing anthropogenic carbon emissions, leading to a larger fraction of anthropogenic CO_2 staying airborne and therefore some amplification of global warming. Despite large uncertainties, all models simulated a relative weakening of both the land and ocean carbon sinks in the future, warmer climate[33].

Policy implications

UNFCCC negotiations are characterized by an extraordinary effort to make use of the best available science as reviewed by the IPCC. However, the negotiations are a complex political process with many interests operating, and as policies are implemented, scientific and/or unintended shortcomings in some decisions are revealed. These are inevitable given that the attempt to mitigate human-forced climate change is a new kind of problem. As inconsistencies in policies are revealed they should be seen as part of an ongoing process for scientists and negotiators to learn and make the necessary improvements.

Parties (that is, countries; including developed and developing) that are signatories to the UNFCCC report on emissions of CO_2 due to change (depletion) in carbon stock from different land cover types. For this purpose they only report on areas of forests identified as 'managed' (but in practice these may include areas considered largely 'natural' — with native species and little or no timber removals, for instance). Under the Kyoto Protocol, Annex I (developed) countries account for changes in stock between the first commitment period (2008 to 2012) and 1990. Under Article 3.3, parties have to report all afforestation, reforestation and deforestation (that is, where there is a change of land use to or from forest land to another land class, such as grassland or cropland). Under Article 3.4 parties can elect to report changes in stocks on areas identified as 'Forest Management'; that is, it is not mandatory. Some countries, Australia for example, opted not to report on these emissions.

The implementation of the Kyoto Protocol for forests is problematic[35] as it does not apply a distinction between natural forest ecosystems and plantations, nor between primary forest and semi-natural forests logged for industrial wood production as there is technically no change in land cover[36]. As noted above, clear-felling of natural forest for even-aged natural regeneration or plantation[37] results in depletion of the land buffer and significant CO_2 emissions[38]. If forest management is elected, these emissions will be captured as change in stock in managed forests between 1990 and the commitment period. If it is not elected, the interpretation of the rules is that the land remains forest land, and no deforestation is deemed to have occurred.

The Durban accounting rules negotiated in 2010 for the second commitment period of the Kyoto Protocol (2013 to 2020) are a significant improvement and address two key concerns[39]. Accounting for emissions from forest management will be mandatory. Accounting for conversion of natural forests to plantation forests will be required (although it is not yet clear if this will be reported under deforestation or forest management). Furthermore, Parties will have to report on how harvesting or forest disturbance that is followed by the re-establishment of a forest is distinguished from deforestation.

Although future accounting approaches thus represent an improvement, there remain concerns that need attention by governments when formulating national policies and programs, and among business and civil society in promoting voluntary and market-based mitigation schemes. If carbon is to be usefully stored (on land, in the ocean or in geological repositories), it must remain stored not just for 100 years, but for more than 10,000 years. This issue of 'permanence' is widely recognized in the UNFCCC negotiations, but not necessarily on the long timescales involved. Indeed it is accepted de facto in many policy contexts that it is sufficient to maintain stores for 100 years. For example, Article 87 of the Australian Government's Carbon Credits (Carbon Farming Initiative) Act 2011 defines the maximum potential relinquishment period for an eligible offsets project as 100 years (that is, the time period the person holding the carbon credit is responsible for the sequestered carbon stock)[40].

Voluntary carbon offset markets in operation that are used by business including airlines, industrial and energy companies[41] tend to have similar misconceptions of the science. It helps to have clarity about the meaning and intention of an 'offset'. It must be recognized that forest conservation can avoid or reduce future carbon emissions, but does not in any meaningful sense offset continuing emissions from other sources. It must also be recognized that the capacity of the land buffer to remove and store CO_2 from the atmosphere is strictly limited. However vigorous the measures taken to increase land carbon stocks, their total potential for carbon storage is minuscule compared with the stock of fossil fuels that could yet be burnt.

Conclusions

On the basis of our review of key scientific issues related to the global carbon cycle, the following insights should be considered when climate change mitigation polices are being negotiated, regulatory frameworks formulated and programmes and projects implemented.

As long as the right kinds of land management responses are implemented, the land carbon buffer can provide a valuable, cost-effective, short-term service in helping to reduce atmCO_2, and slow the rate of anthropogenic climate change, bringing co-benefits for biodiversity and sustainable livelihoods, and giving us some time to develop a low carbon economy.

There are strict, environmentally determined limits on the maximum amount of carbon that can be restored to land carbon stocks, and good reasons why this maximum will not be achieved. Sequestering carbon into depleted ecosystem stocks removes CO_2 from the atmosphere and is thus usefully considered as partially refilling the buffer that was depleted by human activities. Avoiding emissions by protecting high-carbon ecosystems from land-use change that depletes their carbon stocks is an important part of a comprehensive approach to greenhouse gas mitigation. The mitigation value of forests lies not in their present net uptake of CO_2, but in the longevity of their accumulated carbon stocks.

Consistent with our understanding of the lifetime of the airborne fraction of a pulse of CO_2, the most effective form of climate change mitigation is to avoid carbon emissions from all sources. This means that there is no option but to cut fossil fuel emissions deeply, and not to continue these emissions under the erroneous assumption that they can be offset in the long term by the uptake of CO_2 in land systems.

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Additional information

Supplementary information is available in the online version of the paper. Reprints and permissions information is available online at www.nature.com/reprints. Correspondence should be addressed to B.M.

Competing financial interests

The authors declare no competing financial interests.

Forest Bioenergy or Forest Carbon? Assessing Trade-Offs in Greenhouse Gas Mitigation with Wood-Based Fuels

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The potential of forest-based bioenergy to reduce greenhouse gas (GHG) emissions when displacing fossil-based energy must be balanced with forest carbon implications related to biomass harvest. We integrate life cycle assessment (LCA) and forest carbon analysis to assess total GHG emissions of forest bioenergy over time. Application of the method to case studies of wood pellet and ethanol production from forest biomass reveals a substantial reduction in forest carbon due to bioenergy production. For all cases, harvest-related forest carbon reductions and associated GHG emissions initially exceed avoided fossil fuel-related emissions, temporarily increasing overall emissions. In the long term, electricity generation from pellets reduces overall emissions relative to coal, although forest carbon losses delay net GHG mitigation by 16–38 years, depending on biomass source (harvest residues/standing trees). Ethanol produced from standing trees increases overall emissions throughout 100 years of continuous production: ethanol from residues achieves reductions after a 74 year delay. Forest carbon more significantly affects bioenergy emissions when biomass is sourced from standing trees compared to residues and when less GHG-intensive fuels are displaced. In all cases, forest carbon dynamics are significant. Although study results are not generalizable to all forests, we suggest the integrated LCA/forest carbon approach be undertaken for bioenergy studies.

Introduction

Forests can contribute to greenhouse gas (GHG) mitigation strategies through capturing and storing atmospheric CO_2 in live biomass, dead organic matter, and soil pools, supplying a source for wood products that both stores carbon and can displace more GHG-intensive alternatives, and providing a feedstock for bioenergy to displace fossil fuel use. While the merit of each of these options has been individually investigated, trade-offs associated with forest resource utilization decisions must also be considered. Of particular interest is the relationship between harvest and forest carbon storage and how this impacts the GHG mitigation performance of forest products, including bioenergy. Existing tools employed to evaluate emissions associated with different forest resource use decisions are not individually well suited to considering such interactions.

Life cycle assessment (LCA) has been applied to bioenergy options, including electricity generation and transportation fuels. The GHG mitigation potential of bioenergy products depends on activities throughout the entire life cycle (LC), making such a perspective necessary for a comprehensive evaluation. Numerous LCAs have focused on agricultural biomass as feedstock for bioenergy, e.g., reviewed in ref (*1*). Comparatively few LCAs have evaluated bioenergy from forest biomass; those that have examined electricity generation (e.g., ref (*2*)), heating (e.g., ref (*3*)), and transportation (e.g., ref (*4*)). Bioenergy LCAs have generally found that the substitution of fossil fuel-derived energy with biomass-derived alternatives reduces GHG emissions, owing in part to the assumption that biomass-based CO_2 emissions do not increase atmospheric CO_2.

Conventional wisdom has generally accepted this assumption of biomass 'carbon neutrality', and thus, most of the LC GHG emissions associated with bioenergy production are attributed to fossil carbon inputs into the system (*5*). In practice, however, the assumption of carbon neutrality may not accurately represent carbon cycling related to biomass growth (e.g., ref (*6*)). The practice of annual or semiannual harvest in agriculture means that carbon uptake by biomass may reasonably match carbon release in bioenergy systems within a short time frame, although land use change impacts resulting from biomass production can upset this balance (*7*). In temperate forests, the harvest cycle can range from 60 to 100 or more years due to the relatively slow growth of forest species. It could therefore take a century for carbon stocks to be replaced, particularly under a clearcutting regime (harvest of all merchantable trees). Harvest patterns and associated implications for forest carbon stocks vary extensively, ranging from clearcuts to variable retention patterns, including shelterwood and selection cuts. Some variable retention approaches may actually increase forest regeneration, increasing the potential to recover carbon (*8*). Bioenergy production from harvest residues (tree tops and branches) also impacts forest carbon stocks; left uncollected, residues continue to store carbon until released by decomposition or treatment for forest regeneration. While sustainable forest management should ensure that harvest does not impair the long-term productivity of forests, harvest and other forest management activities clearly impact present and future forest carbon stocks. LCA, in its current form, is not well suited to consider the complexities of forest carbon dynamics.

Forest carbon studies have weighed the carbon balance of harvest with the GHG mitigation potential of forest products (e.g., refs *9*–*11*). Some studies have utilized sophisticated forest carbon models to track changes in carbon stored in living biomass (above ground and below ground), dead organic matter, and soil pools (e.g., refs *12, 13*). These studies, however, generally employ simplified assumptions regarding the GHG emissions of forest products (including bioenergy) and have not incorporated a full LC approach. Given the dependence of emissions on specific system

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characteristics (e.g., biomass source, bioenergy production process, fuel displaced), generalized assumptions regarding the GHG mitigation potential of bioenergy are inadequate for informing decision making and public policies.

State-of-the-art tools are available for independently evaluating both the LC emissions of bioenergy systems and forest carbon dynamics. Using these methods in isolation, as has been general practice, stops short of the comprehensive evaluation needed to properly assess the GHG emissions of forest products. In an assessment of GHG mitigation performance of structural wood products, ref (*14*) incorporated LCA with an analysis of forest carbon dynamics. While the study did not consider bioenergy as a product, the results illustrate the importance of considering forest carbon and LC emissions simultaneously when evaluating forest products. Applied to bioenergy, integrating LCA with forest carbon modeling would improve understanding of potential contributions to climate change mitigation.

Bioenergy has been treated inconsistently across energy and climate change policy initiatives in terms of how (or if) GHG emissions are quantified. Forest bioenergy policies that ignore carbon flows in the forest may prove ineffective at achieving actual emissions reductions (*15*). Exclusion of forest carbon from current initiatives is in part due to data issues, although emerging guidelines may ameliorate this situation (*16*). Tools that are able to synthesize forest carbon data and LCA and evaluate trade-offs between bioenergy and forest carbon remain to be developed.

Forest bioenergy has the potential to significantly reduce GHG emissions compared with fossil fuel alternatives. However, interactions between biomass harvest and forest carbon and the resulting effect on the GHG mitigation performance of bioenergy systems are inadequately understood. The objectives of this study are to demonstrate the integration of LCA and forest carbon modeling to assess the total GHG emissions (referred to as "emissions") of forest-based bioenergy options and to determine how emissions reductions associated with bioenergy are impacted when forest carbon is taken into account. We demonstrate this approach through a case study investigating two bioenergy products (wood pellets, referred to as pellets, and ethanol) from two biomass sources (standing trees and harvest residues, referred to as residues) within the Great Lakes−St. Lawrence (GLSL) forest region of Ontario, Canada.

Methods

We develop a framework integrating two analysis tools: life cycle inventory (LCI) analysis and forest carbon modeling. See Supporting Information for additional detail on all methods. LCI analysis quantifies emissions related to the production and use of forest biomass-derived energy. The LCI is based on the assumption of immediate biomass carbon neutrality, as is common practice, and is therefore employed to quantify the impact of all emissions on atmospheric GHGs with the exception of biomass-based CO_2.

Forest carbon modeling quantifies the impact of biomass harvest on forest carbon dynamics, permitting an evaluation of the validity of the immediate carbon neutrality assumption. If biomass-based CO_2 is fully compensated for by forest regrowth, biomass harvest will have no impact on forest carbon stocks. Reduced forest carbon indicates that a portion of biomass-based CO_2 emissions contributes to increased atmospheric GHGs and should be attributed to the bioenergy pathway. The total emissions associated with a bioenergy system are the sum of the two sets of GHG flows (those resulting from the LCI and those from the forest carbon analysis)

$$\mathrm{GHG_{Tot}}(t) = \Delta\mathrm{FC}(t) + \mathrm{GHG_{Bio}}(t) \qquad (1)$$

where $\mathrm{GHG_{Tot}}(t)$ is the total emissions associated with bioenergy, $\Delta\mathrm{FC}(t)$ is the change in forest carbon due to biomass harvest for bioenergy, and $\mathrm{GHG_{Bio}}(t)$ is the GHG emissions associated with bioenergy substitution for a fossil fuel alternative [all reported in metric tonne CO_2 equivlent (tCO_2equiv)] at time *t*.

The change in forest carbon, $\Delta\mathrm{FC}(t)$, is the difference in forest carbon stocks between harvest scenarios: those 'with' and 'without' bioenergy production. While we present this as a single parameter in eq 1, in reality forest carbon models consider the complexity of carbon fluxes between pools within the forest and between the forest and atmosphere. Carbon in biomass harvested for bioenergy is assumed to be immediately released to the atmosphere. However, forest regrowth will capture and store atmospheric CO_2 over time. There is therefore a time dependency to the carbon impact of forest harvest for bioenergy. Assessing the change in forest carbon requires consideration of the forest response following harvest and the fate of the biomass source if it is not harvested for bioenergy (standing trees could be harvested for other uses or never harvested; residues could decompose on site, be burned as part of site preparation, or be collected for other uses). Local conditions influence such factors and must inform specific applications of this method. Information relevant to the current case study is provided in the following methods subsection.

LCI quantifies emissions associated with all activities from initial resource extraction and fuel production through to the use of fuels, inclusive of transportation and distribution stages. Emissions related to the production of inputs are included based on their cradle-to-grave activities. Comparing emissions of a bioenergy product with the relevant reference fossil fuel alternative(s) determines the bioenergy GHG mitigation performance. The output of the bioenergy LCI models, emissions per functional unit, is not directly compatible with the output of forest carbon models, which quantify carbon stocks over relatively long time periods (e.g., 100 years) in order to fully capture the impact of management decisions. To integrate the assessment tools, we quantify the cumulative emissions associated with bioenergy production within the time period investigated with the forest carbon model (e.g., 100 years), considering GHG mitigation from fossil fuel displacement to be permanent. LCI results are converted to a quantity of emissions by

$$\mathrm{GHG_{Bio}}(t) = \int_0^t Q_i(t) \times \mathrm{GHG}_i \,\mathrm{d}t \qquad (2)$$

where $\mathrm{GHG_{Bio}}(t)$ represents emissions associated with bioenergy substitution for fossil fuel alternative(s) at time *t* (tCO_2equiv), $Q_i(t)$ is the quantity of biomass used to produce bioenergy product *i* at time *t* (e.g., oven dry tonne (odt) biomass/year), and GHG_i is the emissions associated with bioenergy product *i* per unit biomass (tCO_2equiv/odt). Summing the bioenergy emissions (based on the LCI results) and the forest carbon emissions gives the total emissions of bioenergy utilization over time as shown in eq 1.

Considering emissions over a long time period is relevant to the carbon dynamics of a forest; however, this introduces uncertainty regarding future forest conditions, markets, and the performance of the energy systems investigated. The LCI and forest carbon analysis in this research consider that these conditions remain static throughout the time frame due to the difficulty of deriving reasonable estimates for these parameters. These issues are further examined in the Results and Discussion.

Application of LCI/Forest Carbon Model framework. We apply the above framework to investigate the impact of forest carbon dynamics on the total emissions associated with several forest-based bioenergy pathways. Forest biomass is assumed to be procured for the production of fuels for electricity generation and light-duty vehicle (LDV) transportation. Reference models are also developed for conventional fuel sources to which the bioenergy pathways are compared. We examine emissions of selected GHGs (CO_2, CH_4, N_2O), reported as CO_2equiv based on 100 year global warming potentials (*17*). See the Supporting Information for additional case study details and data.

The pathways considered are as follows. (1) Electricity generation: (a) Reference coal: production of electricity from coal at an existing generating station (GS) in Ontario; (b) Pellet cofiring, harvest residue: production of electricity at 20% cofiring rate (energy input basis) at retrofit coal GS, pellets produced from residues; (c) Pellet cofiring, standing tree: production of electricity at 20% cofiring rate (energy input basis) at a retrofit coal GS, pellets produced from standing trees. (2) Transportation: (a) Reference gasoline: gasoline use in LDV; (b) E85, harvest residue: ethanol/gasoline blended fuel use in LDV, ethanol produced from residues (biomass is not pelletized); (c) E85, standing tree: ethanol/gasoline blended fuel (85% ethanol by volume) use in LDV, ethanol produced from standing trees (biomass is not pelletized).

Biomass Sources. Biomass is supplied from standing trees and residues from 5.25 million hectares within the GLSL forest region in Ontario. This area represents 19% of provincially owned forest managed for timber production. Trees allocated for harvest that are not currently utilized for traditional products could serve as a source of biomass for bioenergy applications without impacting markets for conventional wood products. Residues do not have a useful purpose in the region's conventional forest products industry and are left to decompose in the forest. Competition for limited wood resources can result in diversion from current uses (e.g., pulp) to bioenergy (*18*) with potential indirect emissions consequences (*7*). By limiting the present study to biomass sources unutilized for conventional products, we avoid such market interactions.

Standing tree harvest and related forest operations (regeneration, road construction/maintenance, and transport to the pellet/ethanol facility) are assessed using a model developed in our previous work (*6*). Emissions related to residue collection are calculated by treating the residues as a byproduct of forest harvest. Only additional fuel use required for collection beyond that of current harvest operations is allocated to the residues; other forest operations are allocated to the primary forest product and are therefore not included in the present study. Residue collection consists of roadside chipping and loading.

Electricity Pathways. LCI models representing electricity generation from coal and cofiring of pellets from standing trees were developed in our prior work (*6*). The models consider emissions associated with the full fuel LCs from initial resource extraction through to combustion as well as upstream emissions related to process inputs. One kWh is selected as the functional unit for the analysis. We assume that pellet production from residues and their use for cofiring is similar to that of pellets from standing trees but modify the pelletization process to reflect that residues are chipped in the forest (standing trees are delivered as logs). For both sources, 15% of input biomass is assumed to be consumed during pellet production to dry the biomass. Avoiding fossil fuel use reduces emissions during the pelletization process but increases biomass input to pellet production and associated forest carbon impacts. Implications of this assumption are considered in Results and Discussion.

Transportation Pathways. Ethanol production, transportation, distribution, and use as E85 fuel in LDV are modeled based on the wood-to-ethanol biochemical conversion pathway in the Government of Canada's "well-to-wheel" model, GHGenius 3.17 (*4*). The gasoline portion of E85 fuel and the reference gasoline pathway are also taken from GHGenius. The functional unit for the transportation pathways is 1 km driven. Significant uncertainty exists in evaluating ethanol production from cellulosic feedstock as technological development and optimization is ongoing and production not yet at commercial scale (*19*).

Forest Carbon. The forest carbon dynamics related to biomass harvest are evaluated using FORCARB-ON, an Ontario-specific adaptation of the FORCARB2 model (*12*). FORCARB-ON quantifies carbon stocks (in living trees, soil, standing dead trees, down dead wood, forest floor, and understory vegetation pools) based on harvest schedules and inventories that producers are required to report to the Province. Harvest schedules take into account species and age composition of the forest, age classes eligible for harvest, natural disturbance frequency, growth rates, and forest succession. The model estimates forest carbon stocks over 100 years, a time frame relevant to the long-term perspective of forest management planning.

We evaluate forest carbon stocks for three potential harvest scenarios: (1) "current harvest" baseline, where biomass (standing trees, residues) is not collected for bioenergy production and therefore timber is removed solely to satisfy the current demand for traditional wood products; (2) "current + residue" harvest, with residue removal for bioenergy production; and (3) "maximum allowable" harvest, with additional standing tree harvest (compared to the baseline) for bioenergy production (residues are not collected). The difference in forest carbon stocks between the bioenergy production scenarios and "current harvest" baseline scenario is allocated to the bioenergy products. Additional standing tree harvest for bioenergy occurs as scheduled under forest management plans; following harvest, stands are regenerated by planting or natural regeneration, varying by site. If not harvested for bioenergy, standing trees eventually undergo natural succession and are subject to a small likelihood of natural disturbance. Residue collection is assumed to not impact soil carbon stocks; uncollected residues are assumed to decompose on site, either at the roadside or near where trees were felled. The consequence of collecting residues for bioenergy production is that this temporary carbon store is 'liquidated' immediately (combusted during bioenergy production and use) rather than decomposing slowly in the forest. Therefore, the associated change in forest carbon is the difference between immediate release (bioenergy) and decomposition over time if not collected. As noted previously, these factors could vary by location with a potentially significant impact on the assessed forest carbon emissions. We do not consider emissions related to the current harvest for traditional wood products or their use. Under the assumptions in this study, this is not affected by the decision to undertake additional harvest or collect residues for bioenergy production.

Results and Discussion

Life Cycle Inventory Results, Excluding Forest Carbon. LCI results for the pathways are shown in Table 1, using the assumption of immediate biomass carbon neutrality. LCI emissions for biomass are greater when sourced from standing trees than from residues. Upstream (fuel production) stages, however, are minor contributors to LC emissions of either pellets or ethanol. The majority of emissions arise from the combustion of fossil fuels, both as the fossil portion during bioenergy use and in the reference fossil pathways. Excluding changes in forest carbon, 20% pellet cofiring reduces LC emissions by 18% compared to coal-only operation (kWh basis) whether standing trees or residues are utilized, whereas an E85-fueled LDV reduces LC emissions by 57% compared to a gasoline LDV (km-driven basis). The greater emission reduction of E85 relative to pellet cofiring gives the appear-

TABLE 1. Life Cycle GHG Emissions Associated with Bioenergy Product (wood pellets, ethanol) Blended for Use and Substitution for Fossil Reference Pathway[a]

	electricity generation pathways			transportation pathways		
life cycle stage	coal[c,d] (g CO_2equiv/kWh)	20% pellet cofiring, residue (g CO_2equiv/kWh)	20% pellet cofiring, standing tree[c] (g CO_2equiv/kWh)	gasoline[f] (g CO_2equiv/km)	E85, residue (g CO_2equiv/km)	E85, standing tree (g CO_2equiv/km)
forest operations		1.9	4.3		5.1	11.7
bioenergy production, distribution [b]		9.5	9.6		46	46
upstream fossil energy component	62	50	50	77	16	16
fuel use (combustion)[e]	939	760	760	211	48	48
total life cycle emissions	1001	821	824	288	116	123

[a] Values assume immediate carbon neutrality and do not take into consideration forest carbon implications. [b] Includes transport of biomass to the production facility, bioenergy production, electricity coproduct credit from biochemical production of ethanol, and bioenergy transportation/distribution stages. [c] Reference (*6*). [d] Surface coal mining removes biomass and disturbs soil, which results in GHG emissions due to direct land use change. These emissions along with other mining process emissions are considered in our analysis. [e] Fuel use consists of GHG emissions from the fossil component of fuel (coal, gasoline) and non-CO_2 GHG emissions associated with bioenergy (pellet, ethanol) combustion. [f] Reference (*4*).

TABLE 2. Forest Carbon Impacts of Continuous Biomass Harvest

	forest carbon stock change ($MtCO_2$equiv)										
	year										
biomass source	0	10	20	30	40	50	60	70	80	90	100
residues	0[a,b]	−8.2	−11.8	−13.0	−13.5	−13.9	−14.3	−14.7	−15.0	−15.2	−15.2
standing trees	0	−43.6	−80.9	−106.3	−112.5	−113.4	−112.7	−132.8	−143.6	−150.8	−150.7

[a] Negative values indicate a GHG emission source (forest carbon stocks are reduced due to biomass harvest) that is attributable to bioenergy production. [b] Reported values are the total stock change due to continuous harvest. For example, 50 years of continuous standing tree harvest reduces total forest carbon stocks by 113.4 $MtCO_2$equiv.

ance that this pathway represents a preferred use of biomass for reducing emissions, but this results primarily from the cofiring scenario utilizing a lower proportion of biomass fuel (20%, energy basis) than E85 (79%, energy basis).

We convert the LC emissions from their initial functional units (kWh, km driven) to a basis of one odt of biomass removed from the forest for bioenergy production ($odt_{biomass}$). This makes the LCI and forest carbon model results compatible and facilitates a comparison of the two bioenergy pathways (electricity, ethanol) in terms of their effectiveness of biomass utilization in reducing emissions (see Supporting Information, equation S-3). Over their respective LCs, the production and use of pellets from standing trees displaces 1.49 tCO_2equiv/$odt_{biomass}$, while ethanol production and use displaces 0.51 tCO_2equiv/$odt_{biomass}$, exclusive of forest carbon impacts. Utilizing residues as a feedstock for pellets and ethanol displaces 1.50 and 0.53 tCO_2equiv/$odt_{biomass}$, respectively. Substitution of coal with pellets provides a greater mitigation benefit than substitution of gasoline with ethanol, primarily due to the higher GHG intensity of coal. To put these values into perspective, the constituent carbon in biomass is equivalent to 1.83 tCO_2equiv/odt. The significance of releasing this biomass-based CO_2 is considered subsequently.

Forest Carbon Analysis Results: Impact of Biomass Harvest. Sustainable biomass sources in the study area could provide, on average, 1.8 million odt/year from standing trees and 0.38 million odt/year from residues. Combined, these sources could provide 2.2% of annual electricity generation in the province or reduce gasoline consumption by 3.3% (see Supporting Information). Forest carbon loss due to undertaking biomass harvest in the study area over a 100 year period is shown in Table 2. For both sources (residues, standing trees), harvest reduces forest carbon asymptotically toward a "steady state". For standing trees, as more stands are harvested for bioenergy over time, the rate of carbon accumulation in regrowing stands increases toward a point where, under ideal conditions, carbon accumulation balances removals associated with continued harvest. For residues, a similar steady state is eventually achieved when the rate of carbon removals at harvest is matched by the expected rate of residue decomposition if harvest is not undertaken. Continuing biomass harvest once a steady state has been reached would not impact forest carbon stocks; however, initiating biomass harvest beyond current removals has significant emissions consequences in the near to medium term. Forest carbon loss due to harvest residue collection approaches a maximum of ~15$MtCO_2$equiv, whereas standing tree harvest for bioenergy results in a carbon loss exceeding 150 $MtCO_2$equiv after 100 years. Proportional to the quantity of biomass provided, standing tree harvest results in a greater impact on forest carbon than harvest residue collection because live trees would generally continue to sequester carbon if not harvested, whereas carbon in uncollected residues declines over time.

Total GHG Emissions: Combined LCI and Forest Carbon Analysis Results. Summing the cumulative emissions of the bioenergy options (LCI results Figure 1, dashed lines) and the forest carbon emissions (Figure 1, dotted lines) results in the total emissions of bioenergy production and use (Figure 1, solid lines). When reductions in forest carbon are included, emission mitigation is delayed and reduced compared to the case where immediate biomass carbon neutrality is assumed. For all scenarios investigated, total emissions from the bioenergy pathways initially exceed those of the reference fossil fuel pathways, indicating an initial increase in emissions resulting from bioenergy use. Emissions associated with forest carbon loss due to biomass harvest exceed the reduction of fossil fuel-based emissions provided by bioenergy substitution. The emissions increase associated with bioenergy, however, is temporary: the rate of forest carbon loss decreases with time, whereas the emissions reduction associated with utilizing bioenergy in place of fossil alternatives continues to increase throughout the 100 year period, proportional to the cumulative quantity of pellets or ethanol produced. A

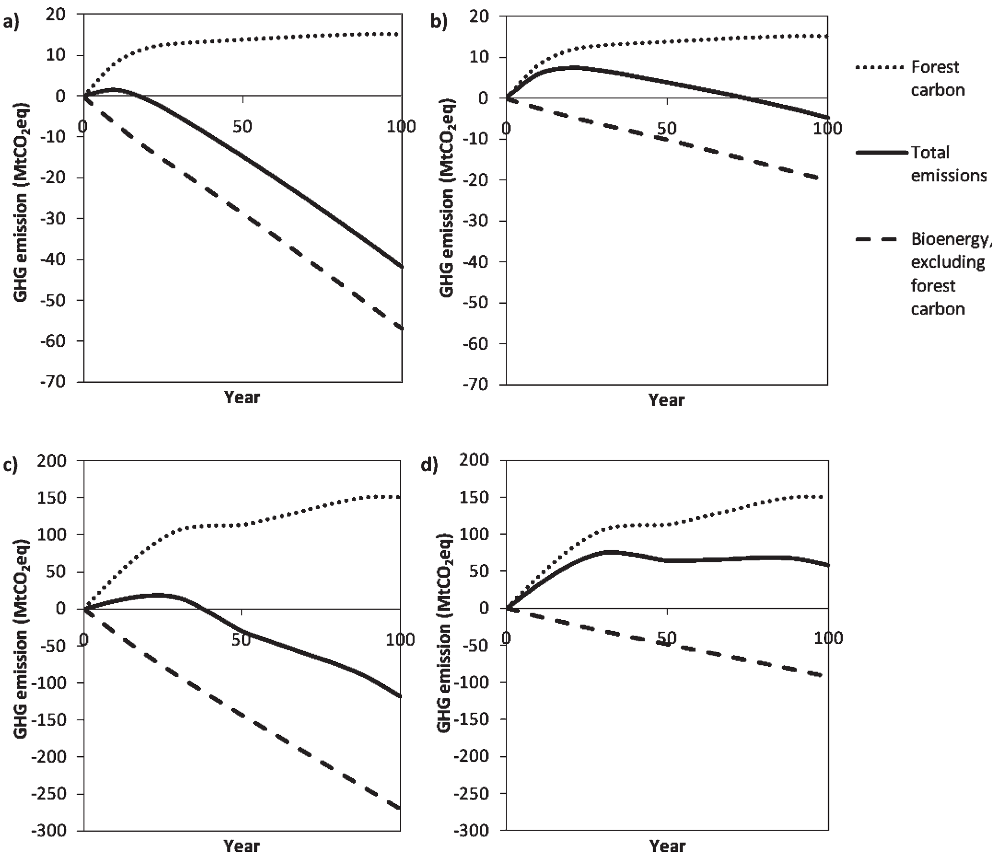

FIGURE 1. Cumulative GHG emissions from continuous biomass harvest for bioenergy production: (a) pellets produced from residues, displacing coal (20% cofiring), (b) ethanol produced from residues, displacing gasoline (E85 fuel), (c) pellets produced from standing trees, displacing coal (20% cofiring), and (d) ethanol produced from standing trees, displacing gasoline (E85 fuel). Positive values indicate an increase in GHG emissions to the atmosphere.

time delay therefore exists before bioenergy systems reach a "break-even" point where total emissions for the bioenergy and reference fossil pathways are equal. Only after the break-even point are net emissions reductions achieved.

Figure 1a and 1b shows the total emissions resulting from continuous use of residues for pellet and ethanol production, respectively, over a 100 year period. Excluding forest carbon, the emissions reduction associated with utilizing bioenergy in place of fossil alternatives increases steadily over time. The reduction of forest carbon stocks due to residue collection slows toward a steady state. Co-firing with pellets produced from residues reduces cumulative emissions relative to coal only after an initial period of increased emissions lasting 16 years. Forest carbon impacts of residue removal reduce the total emission mitigation at year 100 from 57 $MtCO_2$equiv (expected assuming immediate biomass carbon neutrality) to 42 $MtCO_2$equiv.

Compared to the electricity pathway results, utilization of residues for ethanol production is more greatly impacted by changes in forest carbon, due to the lower GHG intensity of the displaced fuel (gasoline compared to coal). An overall emission reduction occurs only after 74 years of continuous production of ethanol; total GHG reductions by year 100 are reduced by 76% from expected performance assuming immediate biomass carbon neutrality.

Due to the greater forest carbon impact of standing tree harvest compared to residue collection, bioenergy production from standing trees performs worse in terms of reducing emissions (Figure 1c and 1d). Pellet production from standing trees results in a greater initial emissions increase, reaching a break-even point only after 38 years of continuous production and use when displacing coal for electricity generation. The total emissions reductions from utilizing wood pellets from standing trees over a 100 year period, expected under the assumption of biomass carbon neutrality, is reduced by 56% when forest carbon impacts are considered.

As in the residue cases, for the standing tree cases forest carbon more significantly impacts total emissions of ethanol than those associated with pellets for electricity generation. Ethanol production from standing trees (Figure 1d) does not reduce emissions at any point within the 100 year period; instead, overall emissions to the atmosphere increase relative to the gasoline reference pathway. Disregarding biobased CO_2 emissions, as is common to most LCAs, would return an opposite, and erroneous, result. This contradiction, also identified elsewhere (*15*), illustrates the misleading consequence of assuming immediate biomass carbon neutrality when quantifying emissions of some bioenergy pathways.

Simply adding biobased CO_2 emissions associated with bioenergy production and use to the LCI totals presented in Table 1 would increase emissions associated with bioenergy. Pellet cofiring (at 20%) would result in (all in gCO_2equiv/kWh) 1039 (residue) and 1042 (standing tree) compared to 1001 for coal only. E85 would emit (all in gCO_2equiv/km) 711 (residue) and 718 (standing tree) compared to 288 for gasoline. This approach, however, would not accurately assess the impact of bioenergy production and use on the atmosphere. By only considering carbon in harvested biomass, near-term emissions would be underestimated (decomposition of uncollected biomass, for example, below ground biomass, is omitted). Mid- to long-term emissions would be overestimated as compensation for biobased CO_2 emissions within the forest (e.g., regrowth) is not considered.

Sensitivity Analysis. A sensitivity analysis is performed to assess the impact of key sources of uncertainty/variability in the LCI and forest carbon model parameters on the study results (see Supporting Information). The results are not sensitive to most parameters, and the general trends of the impacts of biomass harvest on carbon stocks and their contribution to overall emissions were not found to be impacted by uncertainty in the parameters. The pellet pathway results were found to be most sensitive to assumptions related to the quantity of biomass used for drying during pelletization (15% of input biomass in base case) (see Supporting Information Figure S-3). Reducing the consumption of biomass during the drying stage increases pellet output and fossil fuel displacement per unit of input biomass. Colocation of pelletization facilities with processes generating waste heat could reduce the drying energy requirement. If no input biomass is required for drying, there are larger emissions reductions associated with pellet use and the time before reaching break even with the fossil energy system is reduced from 16 to 11 years (residues) and from 38 to 29 years (standing trees). When forest carbon is excluded from the analysis, biomass utilization for drying energy has a minimal impact on LC emissions (*6*).

Study Implications. The simplified assumption of immediate biomass carbon neutrality has been commonly employed in bioenergy studies, owing in part to emissions from the energy and forest sectors being reported separately in national inventories (*17*). This study, however, shows that increasing biomass removals from the forest significantly reduces carbon stocks and delays and lessens the GHG mitigation potential of the bioenergy pathways studied. Ignoring the complex relationship between forest carbon stocks and biomass harvest by employing the carbon neutrality assumption overstates the GHG mitigation performance of forest bioenergy and fails to report delays in achieving overall emissions reductions.

Combining LCI analysis and forest carbon modeling as an analytical approach provides a more accurate representation of the role of forest bioenergy in GHG mitigation. When forest carbon dynamics are included in the case study, the use of forest-based bioenergy increases overall emissions for many years and, in the worst-performing scenario (standing tree harvest for ethanol production), does not yield any net climate mitigation benefit over the 100 year period. Carbon implications of bioenergy production are not limited to forests, and these results should not be taken to suggest that agricultural biomass is inherently preferable. Land use impacts associated with agriculture-sourced bioenergy can greatly increase LC emissions (*7*). Nonbioenergy systems can also impact carbon stocks (e.g., overburden removal in coal mining). While the contribution to total emissions may not be significant in all situations, a comprehensive evaluation of any fossil or renewable system should consider impacts of life cycle activities on terrestrial carbon stocks.

Do our results support continued reliance on fossil fuels for electricity generation and transportation? Fossil fuel use transfers carbon from the Earth's crust to the atmosphere; moving beyond reliance on these energy sources is imperative to address climate change and nonrenewable resource concerns. Bioenergy offers advantages over other renewable options that are limited by supply intermittency and/or high cost. However, effective deployment of bioenergy requires the thoughtful selection of appropriate pathways to achieve overall emissions reductions. Harvesting standing trees for structural wood products has been reported to reduce overall emissions: storing carbon in wood products and displacing GHG-intensive materials (steel, concrete) exceeds associated forest carbon impacts (*14*). In comparison, using standing trees for bioenergy immediately transfers carbon to the atmosphere and provides a relatively smaller GHG benefit from displacing coal or gasoline, increasing overall emissions for several decades. Identifying biomass supply scenarios that minimize forest carbon loss will improve the emission mitigation performance of forest bioenergy. Residues employed for bioenergy reduce emissions from coal after a much smaller delay than standing trees, while other forest biomass sources (e.g., processing residuals) could offer near-term emission reductions if used to replace GHG-intensive fossil fuels. Industrial ecology approaches (e.g., utilizing end-of-life wood products as a biomass source; integrating bioenergy production with other wood products to utilize waste heat for processing) could reduce forest carbon implications of bioenergy production and are deserving of further consideration.

Utilizing bioenergy to displace the most GHG-intensive fossil fuels minimizes initial emissions increases and reduces the time required before net GHG benefits are achieved. Ethanol production for gasoline displacement, under the modeled conditions, is not an effective use of forest biomass for GHG reductions. Displacing coal in electricity generation, in comparison, is superior in reducing emissions. However, this does not indicate that electricity applications are always preferable. The mitigation performance of biomass-derived electricity depends on the displaced generation source. Further, these results represent the expected near-term state of energy system technologies and do not consider changes in either the reference or the bioenergy pathways over the time frame studied. Performance improvements are inevitable with technological maturation and commercialization. Technological developments regarding thermal electricity generation (e.g., efficiency improvements; viable carbon capture and storage) would be applicable to both biomass and coal, while improvements in pellet production would not greatly influence total emissions. Emissions from producing ethanol, regarding both the ethanol production process and the appropriate reference pathway in the future given the limited petroleum supply and associated price volatility, is uncertain and in the future could prove a more effective means of emissions reductions than reported here. Ethanol can also play an important role in addressing economic and energy security concerns related to petroleum dependency.

Although the method demonstrated in this research is generalizable, site-specific characteristics of forests prevent the generalization of specific results from this study. Numerous factors would influence forest carbon dynamics and must be considered in specific analyses. Intensifying silvicultural practices (e.g., planting instead of natural regeneration, utilization of fast-growing species) could shorten, but not eliminate, the period of net emission increase found in our results. In some jurisdictions, residues are burned during site preparation for forest regrowth. Using such residues for bioenergy would not significantly impact forest carbon stocks.

While GHG mitigation is an important consideration of forest resource utilization, numerous other factors must be considered in the decision-making process. In particular, declines in Ontario's forest sector have negatively impacted communities that would welcome the investment and employment opportunities associated with bioenergy. Other environmental factors and technical constraints must be considered before implementing bioenergy production.

The potential of forest-based bioenergy to reduce emissions from fossil fuels must be balanced with forest carbon impacts of biomass procurement. This perspective is of particular importance as policies related to climate change mitigation, deployment of renewable energy, and the forest bioeconomy are developed and implemented. Considering bioenergy in isolation of its impact on forest carbon could inadvertently encourage the transfer of emissions from the energy sector to the forest sector rather than achieve real reductions. Accounting methods must be designed to measure the complete impact of mitigation options on the atmosphere. By considering the broader impacts of bioenergy production on the forest, particularly forest carbon pools, policy can lend support to effective uses of forest resources for climate change mitigation.

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Supporting Information Available

Additional detail on biomass sources, life cycle inventory of bioenergy systems, forest carbon analysis, and additional results and discussion. This material is available free of charge via the Internet at http://pubs.acs.org.

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Carbon debt and carbon sequestration parity in forest bioenergy production

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Abstract

The capacity for forests to aid in climate change mitigation efforts is substantial but will ultimately depend on their management. If forests remain unharvested, they can further mitigate the increases in atmospheric CO_2 that result from fossil fuel combustion and deforestation. Alternatively, they can be harvested for bioenergy production and serve as a substitute for fossil fuels, though such a practice could reduce terrestrial C storage and thereby increase atmospheric CO_2 concentrations in the near-term. Here, we used an ecosystem simulation model to ascertain the effectiveness of using forest bioenergy as a substitute for fossil fuels, drawing from a broad range of land-use histories, harvesting regimes, ecosystem characteristics, and bioenergy conversion efficiencies. Results demonstrate that the times required for bioenergy substitutions to repay the C Debt incurred from biomass harvest are usually much shorter (< 100 years) than the time required for bioenergy production to substitute the amount of C that would be stored if the forest were left unharvested entirely, a point we refer to as C Sequestration Parity. The effectiveness of substituting woody bioenergy for fossil fuels is highly dependent on the factors that determine bioenergy conversion efficiency, such as the C emissions released during the harvest, transport, and firing of woody biomass. Consideration of the frequency and intensity of biomass harvests should also be given; performing total harvests (clear-cutting) at high-frequency may produce more bioenergy than less intensive harvesting regimes but may decrease C storage and thereby prolong the time required to achieve C Sequestration Parity.

Keywords: bioenergy, biofuel, C cycle, C sequestration, forest management

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Introduction

The search for alternatives to fossil fuel energy has yielded several possibilities, many of which are derived from biomass. Bioenergy has been viewed as a promising alternative to fossil fuels because of its capacity to increase the energy security in regions that lack petroleum reserves and because their production and combustion does not require a net transfer of C from Earth's lithosphere to its atmosphere. While bioenergy is understandably among the most heavily promoted and generously subsidized sources of renewable energy, recent research has brought greater attention to the environmental costs of broad-scale bioenergy production (Fargione *et al*., 2008; Searchinger *et al*., 2008, 2009) as well as the limits of how much energy it can actually produce (Field *et al*., 2008).

One alternative to crop-based biofuels is woody biomass harvested directly from forests, an avenue thought to be more promising than harvesting non-woody species for a variety of reasons. First, woody biomass stores more potential energy per unit mass than non-woody biomass (Boundy *et al*., 2011). Second, many forms of non-woody biomass are often utilized following a lengthy conversion process to ethanol or biodiesel, a process which results in a significant loss of potential energy of the harvested biomass (Field *et al*., 2008) as well as additional energy that may be expended in the conversion process itself (Walker *et al*., 2010). By contrast, woody biomass is more readily utilized for energy production without any further modifications (Richter *et al*., 2009). Third, landscapes managed for bioenergy production using woody biomass are able to store more C per unit of land area than crop-based biofuels.

Woody biomass is already a primary source of energy for 2 billion people; the FAO estimates that over half of the world's total round wood removals from forests and trees outside forests are intended for bioenergy production (FAO; Parikka, 2004). Many of these harvests are specifically intended to provide a C-neutral energy source to substitute for fossil fuels (Parikka, 2004; Richter *et al*., 2009; Buford & Neary, 2010), yet such harvests can arrest the C sequestration of many forests far short of their full potential (Harmon *et al*., 1990; Canadell & Raupach, 2008; Pan *et al*., 2011). Much of the world's

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forested land area stores far less C than it potentially could (House *et al.*, 2002; Canadell & Raupach, 2008), and foregoing future harvest/s could provide a more rapid amelioration of atmospheric CO_2 then bioenergy production. A recent study conducted in US West Coast forests examined the C storage/bioenergy production trade-offs of many ecosystems and found that the current C sink for most ecosystems is so strong that it cannot be matched or exceeded through substitution of fossil fuels by forest bioenergy over the next 20 years. However, due to its reliance on existing field data instead of simulation models, it could not extrapolate these results beyond the 20-year period (Hudiburg *et al.*, 2011). Another recent study that addressed these trade-offs is the so-called 'Manomet' study, which modeled bioenergy production systems for different forest types in Massachusetts and found that utilizing forests for bioenergy production reduces C storage without providing an equitable substitution in the near-term (Walker *et al.*, 2010). However, the approach taken by the 'Manomet' study dealt short-term repayment in C Debts at the stand level, while our approach focuses on the C Debt that is incurred as a result harvesting forests for bioenergy production over the long-term at the landscape level. We provide further description of our concept of C Debt *sensu* Fargione *et al.* (2008) by contrasting it with what we refer to as the C Sequestration Parity, which we outline in the discussion below.

Carbon debt

Compared to fossil fuels, woody biomass yields a lower amount of energy per unit mass of C emitted. Since biomass harvesting reduces C storage but does not produce the same amount of energy that would be obtained from an equal amount of C emissions from fossil fuel combustion, recouping losses in C storage through bioenergy production may require many years. We refer to this recoupment as the *C Debt Repayment*, calculated as the change in C storage resulting from bioenergy harvests and associated C substitution, demonstrated in Fig. 1. A mathematical representation is given below in Eqn (1), where $C^{m}_{\text{storage}(t)}$ is the amount of C stored in a managed forest at time t, $C^{m}_{\text{storage}(0)}$ is the amount of C stored in a managed forest at $t = 0$ (before bioenergy harvests have begun), and $C^{m}_{\text{harvest}(t)}$ is the amount of C biomass harvested from a managed forest at time t, which is multiplied by the bioenergy conversion factor $\eta_{biomass}$:

$$C^{m}_{\text{debt}(t)} = C^{m}_{\text{storage}(t)} - C^{m}_{\text{storage}(0)} - \sum_{t=1}^{n} C^{m}_{\text{harvest}(t)} \times \eta_{\text{biomass}} \tag{1}$$

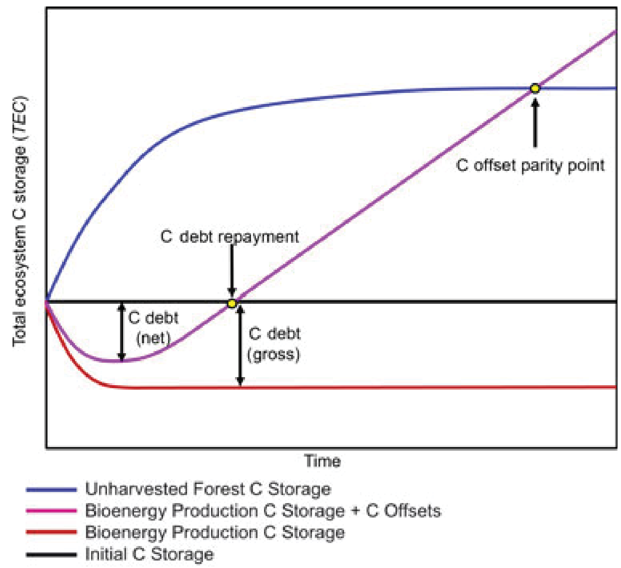

Fig. 1 Conceptual representation of C Debt Repayment vs. the C Sequestration Parity Point. C Debt (Gross) is the difference between the initial C Storage and the C storage of a stand (or landscape) managed for bioenergy production. C Debt (Net) is C Debt (Gross) + C substitutions resulting from bioenergy production.

Carbon sequestration parity

A repayment of the C Debt does not necessarily imply that the forest has been managed for maximal amelioration of atmospheric CO_2. If a forest is managed for the production of bioenergy to substitute for traditional fossil fuel energy as part of an effort to ameliorate atmospheric CO_2 concentrations, such a strategy should be gauged by the climate change mitigation benefits that would accrue by simply leaving the forest unharvested. Ascertaining the point at which a given strategy provides the maximal amount of climate change mitigation benefits requires accounting for the amount of biomass harvested from a forest under a given management regime, the amount of C stored under a given management regime, and the amount of C that would be stored if the forest were to remain unharvested (Schlamadinger & Marland, 1996a,b,c; Marland & Schlamadinger, 1997; Marland *et al.*, 2007). It is expected that a forest that is continuously managed for bioenergy production will eventually produce enough bioenergy to 'recoup' the associated loss in C storage (the so-called carbon debt) through the substitution of bioenergy for fossil fuel energy. However, the ultimate effectiveness of this strategy should be determined by the amount of time required for the sum of the total ecosystem C storage and bioenergy C substitution to exceed the amount of C that would be stored if that same forest were to remain unharvested (Fig. 1). We refer to this difference as the C sequestration differential ($C^{m}_{differential(t)}$), illustrated in Eqn (2) below:

$$C^{m}_{differential(t)} = C^{u}_{storage(t)} - C^{m}_{storage(t)} - \sum_{t=1}^{n} C^{m}_{harvest(t)} \times \eta_{biomass} \quad (2)$$

where $C^{u}_{storage(t)}$ is the amount of C stored in an *unmanaged* forest at time *t*. We refer to the crossing of this threshold as the point of *C Sequestration Parity*. Thus, we make a distinction between the amount of time required for the bioenergy production system to recoup any reductions in C storage resulting from bioenergy production (C Debt repayment) and the amount of time required for the bioenergy production system to equal the C than would be stored if the forest were to remain unharvested (C Sequestration Parity Point), as the latter represents a more ambitious climate change mitigation strategy (Fig. 1).

Materials and methods

We simulated the growth and harvest of woody biomass using a significantly updated version of the ecosystem simulation model LANDCARB (Harmon, 2012). LANDCARB is a landscape-level ecosystem process model that can simulate a full spectrum of potential harvesting regimes while tracking the amount of material harvested, allowing one to simulate ecosystem C storage while tracking the amount of fossil fuel C that could be substituted by using harvested materials as biomass fuels. LANDCARB integrates climate-driven growth and decomposition processes with species-specific rates of senescence and mortality while incorporating the dynamics of inter- and intra-specific competition that characterize forest gap dynamics. Inter- and intra-specific competition dynamics are accounted for by modeling species-specific responses to solar radiation as a function of each species' light compensation point and assuming light is delineated through foliage following a Beer-Lambert function. By incorporating these dynamics the model simulates successional changes as one life-form replaces another, thereby representing the associated changes in ecosystem processes that result from species-specific rates of growth, senescence, mortality, and decomposition.

LANDCARB represents stands on a cell-by-cell basis, with the aggregated matrix of stand cells representing an entire landscape. Each cell in LANDCARB simulates a number of cohorts that represent different episodes of disturbance and colonization within a stand. Each cohort contains up to four layers of vegetation (upper tree layer, lower tree layer, shrub, and herb) that each have up to seven live pools, eight dead pools, and three stable pools. For example, the upper and lower tree layers are comprised of seven live pools: foliage, fine-roots, branches, sapwood, heartwood, coarse-roots, and heart-rot, all of which are transferred to the appropriate dead pool following mortality. Dead sapwood and dead heartwood can be either standing or downed to account for the different microclimates of these positions. Dead pools in a cell can potentially contribute material to three, relatively decay-resistant, stable C pools: stable foliage, stable wood, and stable soil. There are also two pools representing charcoal (surface and buried).

Our modeling approach with LANDCARB was designed to account for a broad range of ecosystem characteristics and initial landscape conditions of a forest, both of which are influential in determining rate of C debt repayment and the time required for C sequestration parity. Forests with high productivity can generate fossil fuel substitutions more rapidly than forests with low productivity. Conversely, forests with high-longevity biomass raise the C storage of the ecosystem (Olson, 1963), which has implications for C debt and C sequestration parity. Furthermore, forests can contain a wide range of C stores even within a fixed range of productivity and C longevity (i.e., lower rates of mortality and decomposition; Smithwick *et al.*, 2007), yet we know of no study to date that has examined the impact of forest productivity and biomass longevity on C Debt repayment or C Sequestration Parity. Furthermore, we know of no previous study that examines a sufficiently large range of forest management strategies and land-use histories to ascertain exactly what sort of situation/s might provide for an efficient utilization of forest biomass for bioenergy production.

To provide a more comprehensive evaluation of the effectiveness of utilizing forest bioenergy as a substitute for fossil fuels, we performed our analysis across a wide range of ecosystem properties by simulating three levels of forest growth and three levels of biomass longevity, resulting in nine distinct ecosystems (Table 1). Levels of longevity were drawn from published rates of bole growth efficiency, mortality, and decomposition (growth and biomass Harmon *et al.*, 2005). The upper and lower bounds of these parameters were intended to cover the range of these processes for most of the world's temperate forests. Our parameters are largely drawn from forests of the US Pacific Northwest, but the extreme values of bole growth efficiency, mortality, and decomposition could be considered extreme values of other forests as well, thereby giving our results maximal applicability.

We ran each of our nine simulated ecosystems under four sets of initial landscape conditions: afforesting post-agricultural land (age = 0), forest recovering from a severe disturbance (age = 0), old-growth forest (age > 200 years), and a forest harvested on a 50-year rotation (mean age ~25 years). Each combination of ecosystem characteristics and land-use history was simulated with seven different management strategies (Table 2), which included one unharvested control group as well as three biomass harvest frequencies (25, 50, 100 years) applied at two different harvest intensities (50% harvest of live stems, 100% harvest of live stems). We assumed that our post-agricultural landscape did not have any legacy C storage apart from a small amount of soil C, thus our post-agricultural simulation did not have any spin-up simulation. However, simulations of the other land-use histories all had a 500-year spin-up simulation were run to establish initial live, dead, and soil C stores. Additionally, for the two simulations that were recovering from harvests and prior disturbance (recently disturbed and rotation forest) we tracked the respective C stores from these events. To simulate a landscape that had previously been harvested on a 50-year rotation, we simulated an annual clearcut on 2% of the landscape throughout the 50 years prior to the

Table 1 Table of selected growth, mortality, and decomposition characterstics for each of our nine ecosystems. Classifications G1, G2, and G3 represent increasing growth rates, represented by the Site Index. L1, L2, and L3 represent increasing biomass longevities. The group with the lowest potential C storage had the lowest growth rate (G1) combined with the highest rates of mortality and decomposition that yielded the lowest rates of biomass longevity (L1). The upper and lower bounds of our rates of growth and longevity were intended to cover the range of these processes for most of the world's forests, thereby giving our results maximal applicability. Thus, the group referred to as G1-L1 is the group with the lowest potential C storage, while the group referred to as G3-L3 has the highest potential C storage. Also note that L1 and L3 values represent extreme values of mortality and decomposition, whereas L2 represents a median value, rather than a midpoint between L1 and L3. $\mathrm{Mortality}_{\mathrm{MAX}}$ is the maximum rate of mortality, while k_{Foliage} and $k_{\mathrm{Heartwood}}$ are decomposition constants for foliage and heartwood. Potential C Storage is the mean amount of C storage of an old-growth stand under these characteristics, as measured over a 500 year interval

Group	Bole growth efficiency +ΔMg Stem C/+ΔMg Leaf C)	$\mathrm{Mortality}_{\mathrm{MAX}}$ (yr^{-1})	k_{Foliage} (yr^{-1})	$k_{\mathrm{Heartwood}}$ (yr^{-1})	Potential C storage (Mg C ha^{-1})
G1-L1	0.35	0.03	0.25	0.1	212
G1-L2	0.35	0.02	0.2	0.02	230
G1-L3	0.35	0.01	0.15	0.01	296
G2-L1	0.54	0.03	0.25	0.1	359
G2-L2	0.54	0.02	0.2	0.02	492
G2-L3	0.54	0.01	0.15	0.01	621
G3-L1	0.84	0.03	0.25	0.1	645
G3-L2	0.84	0.02	0.2	0.02	757
G3-L3	0.84	0.01	0.15	0.01	954

Table 2 List of all bioenergy production system characteristics simulated. We incorporated four land-use histories, three levels of biomass accumulation, three levels of biomass longevity, three different harvest frequencies and two levels of harvest intensity

Land-use histories	Growth rates	Biomass longevities	Harvest frequencies	Harvest intensities
Post-agricultural (age = 0)	G1*	L1*	100 (100Y)	50% (050H)
Recently disturbed (age = 0)	G2*	L2*	50 (50Y)	100% (100H)
Rotation forest (age ~25)	G3*	L3*	25 (25Y)	
Old-growth (age > 200)				

*See Table 1 for details.

completion of the spin-up. In accordance with a prior framework for harvested C decomposition, we assumed that 60% of the harvested C would go directly into long-term C storage mediums (i.e., houses, buildings) that decayed at the rate of 1% per year (Harmon & Marks, 2002). The remaining 40% of the harvested C was assumed to be lost to the atmosphere during manufacturing (Harmon & Marks, 2002). Landscapes were first harvested for bioenergy production in the year following the completion of the spin-up.

Initial conditions of our disturbed forest were analogous to those of a severe pine beetle outbreak. To simulate this condition, we initiated a total mortality of all trees at the end of the spin-up, prior to the biomass harvests. We then simulated an annual salvage logging on 5% of the landscape for each of the 5 years following the simulated pine-beetle disturbance (25% of the landscape was salvage logged). We assumed that 75% of all salvageable biomass was removed in each salvage logging. Salvageable materials harvested in the first 5 years following disturbance were assumed to be stored in wood products and subject to the same decomposition scheme outlined above for the 50-year Rotation Harvest. Such conditions are fairly similar to those in a landscape subject to a high-severity, stand-replacing wildfire, though a landscape subject to a pine beetle infestation will initially have more C storage than one experiencing a high-severity wildfire. However, this difference is temporary and would have a minimal effect on the long-term effects of biomass harvesting, thus this set of initial conditions could also be considered as a proxy for the initial conditions that would follow a high-severity wildfire.

Wildfire

Our analysis also incorporates wildfires in all simulations, not only because they are naturally occurring phenomena in many forest ecosystems, but also because amount of harvestable biomass in an ecosystem can be altered by the event of wildfire, which needs to be accounted for. In the LANDCARB model, fire severity controls the amount of live vegetation killed and the amount of combustion from the various C pools, and is influenced by the amount and type of fuel present. Fires can increase (or decrease) in severity depending on how much the weighted fuel index a given cell exceeds (or falls short of) the fuel level thresholds for each fire severity class (T_{light}, T_{medium}, T_{high}, and T_{max}) and the probability values for the increase or decrease in fire severity (P_i and P_d). For example, a low-severity fire may increase to a medium-severity fire if the fuel index sufficiently exceeds the threshold for a medium-severity fire. Fuel level thresholds were set by monitoring fuel levels in a large series of simulation runs where fires were set at very short intervals to see how low fuel levels needed to be to create a significant decrease in expected fire severity.

The fire regime for low-growth forests (G1) is characterized by a low-severity, high frequency fire regime, with a mean fire return interval (MFRI) of 16 years (Bork, 1985), similar to the fire regime in a Ponderosa pine forest, also a low-growth rate forest. Fire regimes for the medium and high-growth forests (G2, G3) consisted of high-severity, low frequency (MFRI = 250 years) fire regimes, similar to that of a Douglas-fir or Sitka spruce forest (Cissel *et al*., 1999). We generated exponential random variables to assign the years of fire occurrence (Van Wagner, 1978) based on literature estimates (Bork, 1985) for mean fire return intervals (MFRI) for each ecosystem. The cumulative distribution for our negative exponential function is given in Eqn (1) where X is a continuous random variable defined for all possible numbers x in the probability function P and λ represents the inverse of the expected time for a fire return interval given in Eqn (2).

$$P\{X \leq x\} = \int_0^x \lambda e^{-\lambda x} \mathrm{d}x \tag{1}$$

where

$$E[X] = \frac{1}{\lambda} \tag{2}$$

Fire severities in each year generated by this function are cell-specific, as each cell is assigned a weighted fuel index calculated from fuel accumulation within that cell and the respective flammability of each fuel component, the latter of which is derived from estimates of wildfire-caused biomass consumption.

Bioenergy conversion factors

Previous studies on the mitigation potential of bioenergy have yielded conflicting conclusions about the potential for bioenergy production from woody biomass (Schlamadinger & Marland, 1996a,b,c; Marland & Schlamadinger, 1997; Marland *et al*., 2007; Walker *et al*., 2010). Differences in these conclusions are due, in part, to the different assumptions regarding the efficiency of bioenergy utilization. Energy is required for transporting biomass and powering bioenergy conversion facilities, and some is lost due to inefficiencies in the conversion process (Hamelinck *et al*., 2005; Walker *et al*., 2010). Thus, it is difficult to provide a one-size-fits-all estimate of bioenergy conversion efficiency. Rather than using one value, we will evaluate a range of bioenergy conversion efficiencies, ranging from 0.2 to 0.8, to ascertain the sensitivity of C offsetting schemes to the range in variability in the energy conversion process. We estimate the *average* bioenergy conversion factor for woody biomass ($\eta_{biomass}$) to be 0.51, meaning that harvesting 1 Mg of biomass C for bioenergy production will substitute for 0.51 Mg fossil fuel C since less energy per unit C emissions is obtainable from biomass compared to fossil fuel. Calculations for this conversion factor ($\eta_{biomass}$) are in the Supporting Information. A conversion factor of 0.8 represents a highly efficient utilization of bioenergy, though such a conversion efficiency is likely not realistic. Conversely, a conversion factor of 0.2 represents a highly inefficient method of energy utilization, though some bioenergy facilities and conversion processes do operate at this low level of efficiency (Walker *et al*., 2010).

We ran our analysis across 252 distinct scenarios, as we had nine distinct ecosystems (based on three levels of forest growth for three levels of biomass longevity), four initial types of initial landscape conditions, and seven treatment groups (one control, plus three treatment frequencies applied at two levels of intensity). Output from the 252 distinct modeling scenarios was analyzed using seven different bioenergy conversion factors, meaning that our analysis had 1764 combinations of ecosystem properties, initial landscape conditions, harvest frequencies, and bioenergy conversion factors. Our analysis quantifies the degree to which the harvesting and utilization of forest-derived bioenergy alters the landscape-level C storage and bioenergy production in order to calculate (1) the time required for the C mitigation benefits accrued by forests managed for bioenergy production to repay the C Debt incurred from the harvest, and (2) the time required for the C mitigation benefits accrued by forests managed for bioenergy production to achieve C Sequestration Parity, the point at which the sum of forest C storage and bioenergy C substitution equals or exceeds the C mitigation benefits of a comparable forest that remained unharvested.

Results

Times required for repayment of the carbon debts

Most Post-Agricultural landscapes repaid their C debts within 1 year because their initial live C storages were low to begin with and did not require any waiting period for the repayment of their C Debt (Fig. 2). Thus, by undergoing a conversion from a Post-Agricultural landscape to a bioenergy production landscape, there was a repayment of the C Debt as well as an increase in landscape C storage. Similarly, Rotation Harvest landscapes harvested for bioenergy production every 100 years increased their C storage, as they were previously harvested at a frequency of 50 years. Most of the Rotation Harvest landscapes repaid their C Debt in a year due to their initially low live C storage, as their average stand age is ~25 years. However, some of these landscapes that were clear-cut every 50 or 25 years required much longer to repay their C Debt. Harvesting with greater frequency and intensity lowers C storage and prolongs the time needed for repayment of the C Debt; clear-cut harvests performed on Rotation Harvest landscapes every 25 years required 100 to over 1000 years to repay their C Debt. Once a landscape requires several years to repay its C Debt, it may then exhibit sensitivity to the bioenergy conversion efficiencies used to calculate rate at which it can substitute for C emissions from fossil

 GCB Bioenergy, doi: 10.1111/j.1757-1707.2012.01173.x

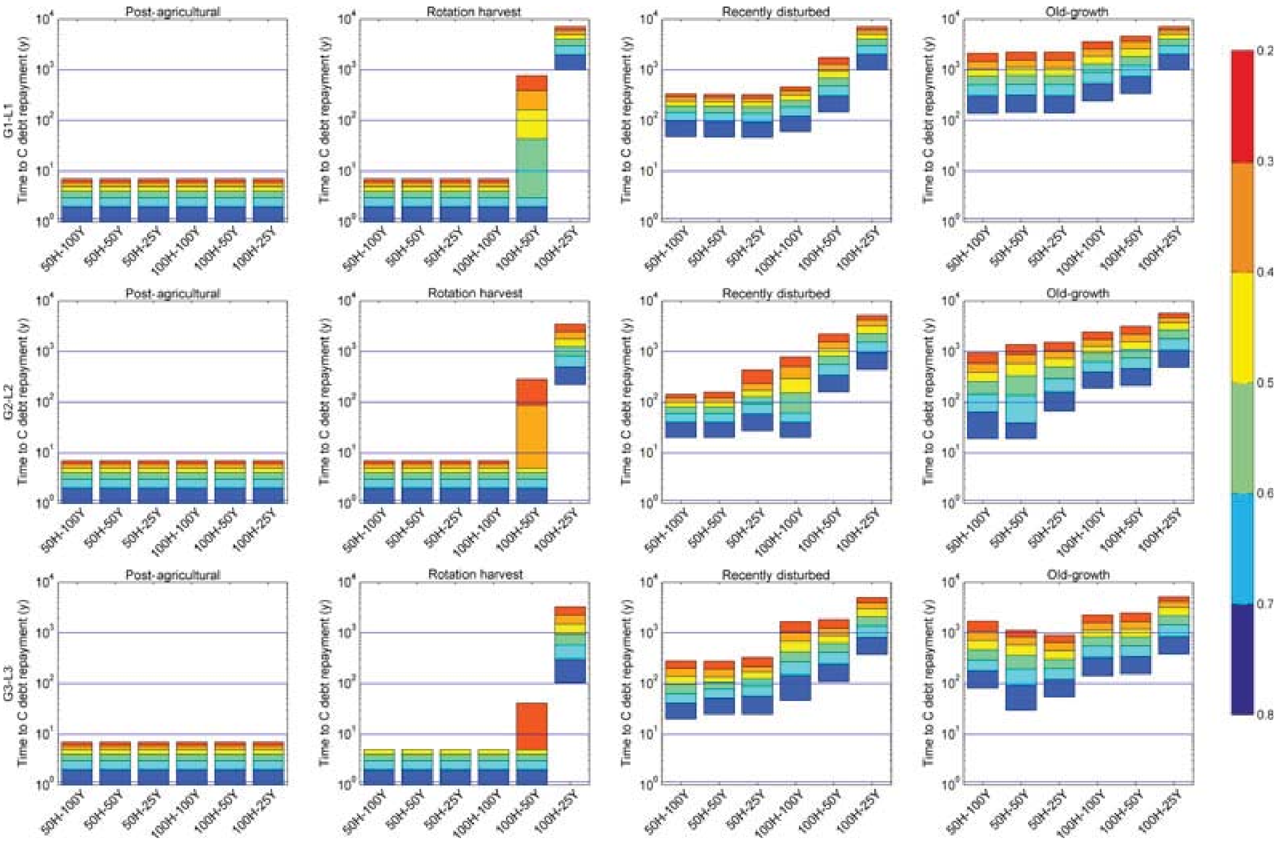

Fig. 2 Comparisons of the time required for a repayment of the C Debt Repayment among three of our nine ecosystem types, each with six biomass harvesting regimes and four land-use histories. Note that times are represented on a log scale. Different harvesting regimes are indicated on the *x*-axis, with 50% and 100% harvesting intensity represented as 50H and 100H, respectively. Harvest frequencies of 25, 50, and 100 years are represented as 25Y, 50Y, and 100Y.

fuels. Recently Disturbed landscapes required more time for a repayment of the C Debt and were much more sensitive to harvest frequency, harvest intensity, and bioenergy conversion efficiencies (Fig. 2). Following disturbance, these landscapes can store high amounts of dead C that can persist for decades. Due to low net primary production following disturbance, recovery to pre-disturbance levels of C storage can take many years, ranging from 20 to over 1000 years. Old-growth landscapes usually took the longest amount of time to repay their C debts because their initial C storages were so high, ranging from 19 to over 1000 years.

Times required to reach carbon sequestration parity

The amounts of time required for C Sequestration Parity were usually longer than the amounts of time required for a repayment of the C debt. In general, Old-Growth landscapes achieved C Sequestration Parity at a faster rate than other categories of land-use history since they have more initial biomass available for bioenergy production (Fig. 3). Recently Disturbed landscapes were the second fastest, followed by Rotation Harvest landscapes, though differences between these two categories of land-use history are relatively minor. Post-Agricultural landscapes took longer than the other categories of land-use history, due to of a lack of initial biomass available to harvest for bioenergy production.

Times required to reach C Sequestration Parity were longest for the low-productivity ecosystems and shortest for the high-productivity ecosystems (Fig. 3), indicating that high productivity ecosystems were able to more quickly recoup their substantial reductions in C storage compared to the rates at which low-productivity ecosystems were able to recoup their considerably smaller reductions in C storage. Within each respective grouping of ecosystem productivity (G1, G2, G3), there were significant effects of different biomass longevities (L1, L2, L3) on the amount of time required for C Sequestration Parity. Increased biomass longevity (i.e., lower rates of mortality and decomposition) increased

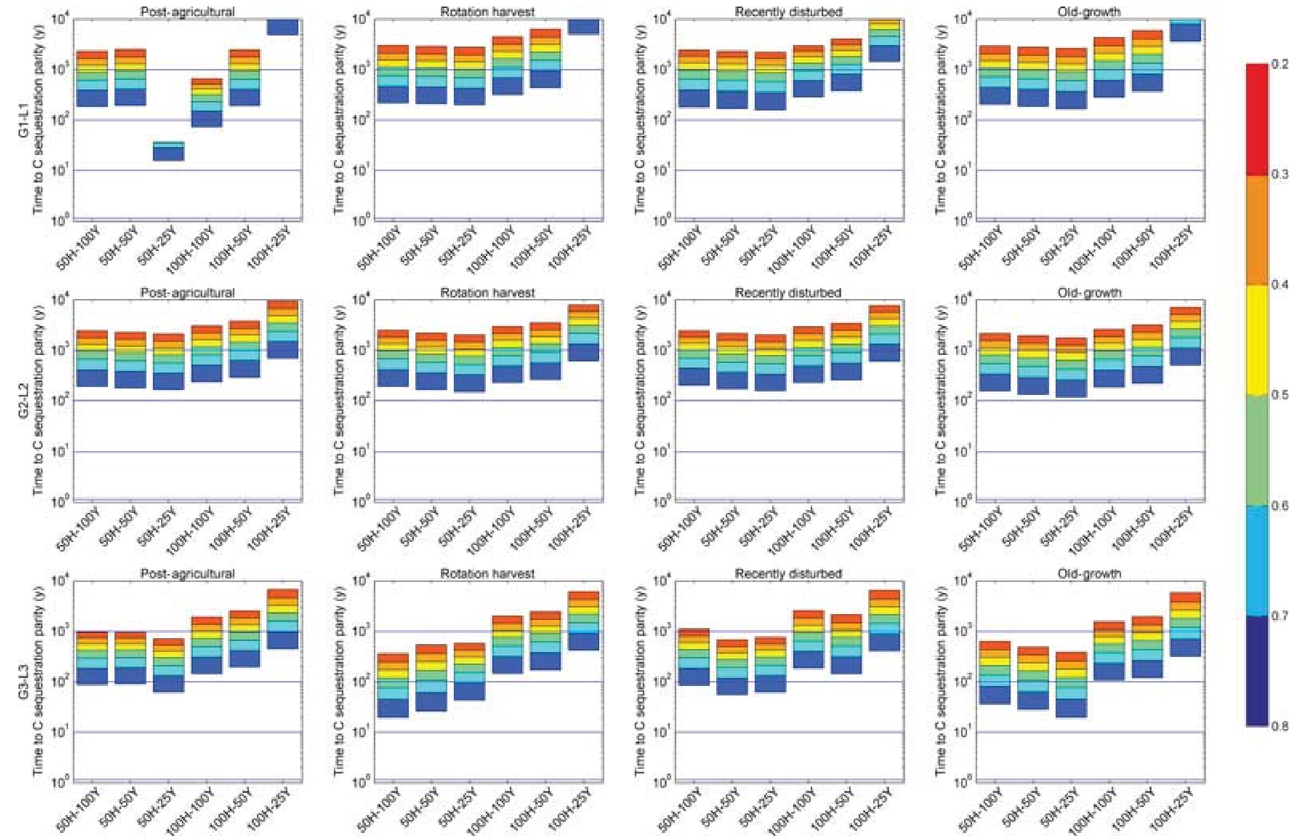

Fig. 3 Comparisons of the time required for a repayment of the C Sequestration Parity among three of our nine ecosystem types, each with six biomass harvesting regimes and four land-use histories. Note that times are represented on a log scale. Different harvesting regimes are indicated on the *x*-axis, with 50% and 100% harvesting intensity represented as 50H and 100H, respectively. Harvest frequencies of 25, 50, and 100 years are represented as 25Y, 50Y, and 100Y.

the times required to reach C Sequestration Parity, a trend which was consistent across all three rates of ecosystem productivity.

Regardless of land-use history and ecosystem characteristics, most scenarios required well over 100 years to reach C Sequestration Parity. Simulations with total harvests performed every 25 years often required more than 1000 years for C Sequestration Parity. Some scenarios achieved C Sequestration Parity in < 50 years, but most of these were scenarios with relatively high bioenergy conversion efficiencies. Harvests performed at lower frequency (50, 100 years) and intensity (50% harvest) required less time; partial harvests (50% harvest) performed every 25 years appeared to reach C Sequestration Parity more rapidly than any other management regime. Harvesting frequency and intensity appeared to affect all ecosystems similarly. Without exception, performing a clear-cut every 25 years resulted in the greatest reduction in C storage and required the longest periods to achieve C Sequestration Parity, suggesting that attempts to generate bioenergy from forests would be most effective in substituting for fossil fuels when managed for moderate amounts of production over a long time scale.

Discussion

Delays in the time required for a net benefit of a substitution of bioenergy for fossil fuels are caused by two factors. First, harvesting materials for bioenergy increases the C losses from the forest over the losses caused by mortality and decomposition, thus, increasing the amount of biomass harvest for bioenergy production will increase the C Debt. Second, since there is less potential energy per unit of C emissions in biomass energy compared to fossil fuels, substituting biomass for fossil fuels does not result in a 1 : 1 substitution of energy per unit of C emission. Consequently, ecosystems that are capable of quickly repaying their C Debts were those that had little C storage to begin with.

Our simulations demonstrated that initial landscape conditions and land-use history were fundamental in determining the amount of time required for forests to repay the C Debt incurred from bioenergy production.

While Recently Disturbed and Old-Growth landscapes required considerable time to repay their C Debts, Post-Agricultural and Rotation Harvest landscapes were capable of repaying their C Debt in relatively short time periods, often within 1 year. However, a quick repayment of the C Debt and an increase in C storage does not imply a high degree of bioenergy production; it merely indicates that more C is being stored in a bioenergy production system. Post-Agricultural landscapes undergoing afforestation have minimal initial C storage, and managing them for an appreciable yield of bioenergy production would require a considerable waiting period. Furthermore, the conversion of an agricultural field to a forest could have short-term climatic warming effects while the afforesting landscape is in the early stages of succession, since a decrease in landscape albedo resulting from afforestation could yield climatic warming effects that would overshadow any climatic cooling effects associated with an uptake of atmospheric CO_2 (Jackson *et al.*, 2008; Anderson *et al.*, 2011), as the latter would be relatively small during the early stages of forest succession. By contrast, a Rotation Harvest system would not undergo a significant change in albedo during a transition to a landscape managed for bioenergy production. However, Rotation Harvests have a much different legacy than a Post-Agricultural landscape, since a history of harvesting on the landscape implies that there is additional wood being stored in wood products which are slowly decomposing (see Methods). Consequently, the ongoing decomposition of previously harvested materials lowers terrestrial C storage.

The times required for Old-Growth landscapes to repay C Debt were similar to the times required for them to achieve C Sequestration Parity, since the initial C storage of an old-growth landscape is at or near the level of C that could be stored in the landscape if it were to remain unharvested. Consequently, Old-Growth landscapes required long periods of bioenergy production to achieve C Debt Repayment and C Sequestration Parity. For the three other land-use histories, reaching the point of C Sequestration Parity requires much more time than a repayment of C Debt. Trends were quite consistent among the Recently Disturbed, Rotation Harvest, and Old-Growth landscapes and most simulations required at least 100 years to reach C Sequestration Parity (Fig. 3).

Times required for C Sequestration Parity were longest for the low-productivity ecosystems and shortest for the high-productivity ecosystems. Similarly, the effects of biomass longevity were quite consistent among the Recently Disturbed, Rotation Harvest, and Old-Growth landscapes (Fig. 3). Within each respective grouping of ecosystem productivity (G1, G2, G3), there were significant effects of different biomass longevity rates (L1, L2, L3) on the amount of time required to reach a point of C Sequestration Parity. Higher rates of biomass longevity (i.e., lower rates of mortality and decomposition) resulted in longer times required for C Sequestration Parity, a trend which was consistent across all three rates of ecosystem productivity (Fig. 3). Such a result may seem counterintuitive at first, but the net effect of lowering mortality and decomposition rates is that potential C storage is increased. Since ecosystems with lower mortality and slower decomposition have higher potential C storage, more bioenergy substitutions must be produced to exceed the amount of C stored in a forest that is allowed to grow without harvest. Annual biomass harvest varied little among our different levels of longevity. Therefore, higher rates of biomass longevity raised the target for C Sequestration Parity without resulting in a comparable increase of bioenergy production. We note that biomass longevity is largely a function of the environmental factors that control rates of biomass decomposition, such as temperature and moisture, and is governed by catastrophic disturbances to a lesser degree. Our simulations reiterate previous findings (Mitchell *et al.*, 2009; Campbell *et al.*, 2012) about the limited impact that wildfires have on biomass longevity; wildfires may temporarily lower the C storage of the landscape but most of the losses that occur are among unharvestable components of the forest, such as leaf litter and fine woody debris. Most of the harvestable biomass remains unconsumed even by high-severity wildfires and can either be salvage harvested shortly thereafter or persist on the landscape for decades (Mitchell *et al.*, 2009; Campbell *et al.*, 2012).

However, C storage is not the only way that vegetation affects climate, as different levels of surface reflectance (albedo) and evapotranspiration result in different levels of heat absorbance in the terrestrial biosphere (Jackson *et al.*, 2008; Anderson *et al.*, 2011). Utilizing degraded agricultural lands for the production of bioenergy via non-woody plant species (i.e., switchcane, switchgrass, etc.) could both reduce heat absorbance in the terrestrial biosphere and produce bioenergy to serve as a substitute for fossil fuels. A recent study by Beringer *et al.* (2011) estimated that, by 2050, the cultivation of bioenergy crops on degraded agricultural land could produce 26–116 EJ yr^{-1}, 3–12% of projected global energy demand. Additional energy may be obtained from secondary sources, such as residues from agriculture and forestry, municipal solid waste, and animal manures, and the combined production potential could potentially be around 100 EJ yr^{-1} by then (Ifeu, 2007; Iea, 2009; Wbgu, 2009; Haberl *et al.*, 2010), thereby generating an additional 10% of projected global energy demand (13–22% total). However, it is unclear what proportion of degraded agricultural lands would be better utilized for climate change mitigation via reforestation, rather than by non-woody bioenergy production. Non-woody bioenergy crops would need a sufficiently high surface reflectance if their climate change mitigation benefits were to exceed the mitigation benefits of afforestation, but the studies conducted on this topic have yielded conflicting results. Some studies have suggested that land cover types with high albedos could yield a greater cooling to the atmosphere than temperate forests (Diffenbaugh & Sloan, 2002; Oleson *et al*., 2004; Bala *et al*., 2007) while other studies have shown the opposite (DeFries *et al*., 2002; Jackson *et al*., 2005; Juang *et al*., 2007), indicating that further research on these tradeoffs is needed.

Further research is also needed to ascertain the potential conversion efficiencies of woody biomass. Our findings indicate that an accounting of the C emissions that are necessary for the harvest, transport, and firing of woody biomass must be performed if forest bioenergy is to be utilized without adding to atmospheric CO_2 concentrations in the near-term. Many of our combinations of forest productivity, biomass longevity and harvesting regimes required more than 100 years to achieve C Sequestration Parity, even when the bioenergy conversion factor was set at near maximal level. A consideration of stand characteristics and land-use history may also prove to be imperative for any bioenergy production system to be effective. Competing land-use objectives make it highly unlikely that forests will be managed purely for C mitigation efforts, and many of the current management objectives within existing forests will undoubtedly prevent them from reaching their full C storage potential. Achieving the maximal C mitigation potential of what remains becomes all the more imperative, as mean global temperatures, sea-level rise, or the melting of ice sheets may continue long after any future stabilization of atmospheric CO_2 and other greenhouse gases (Jones *et al*., 2009). Managing forests for maximal C storage can yield appreciable, and highly predictable, C mitigation benefits within the coming century, while managing forests for bioenergy production will require careful consideration if they are to provide a C neutral source of energy without yielding a net release of C to the atmosphere in the process.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. Times for Carbon Debt Repayment for all Post-Agricultural landscapes.

Figure S2. Times for Carbon Sequestration Parity for all Post-Agricultural landscapes.

Figure S3. Times for Carbon Debt Repayment for all Rotation Harvest landscapes.

Figure S4. Times for Carbon Sequestration Parity for all Rotation Harvest landscapes.

Figure S5. Times for Carbon Debt Repayment for all Recently Disturbed landscapes.

Figure S6. Times for Carbon Sequestration Parity for all Recently Disturbed landscapes.

Figure S7. Times for Carbon Debt Repayment for all Old-Growth landscapes.

Figure S8. Times for Carbon Sequestration Parity for all Old-Growth landscapes.

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Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues

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Abstract

Forest harvest residues are important raw materials for bioenergy in regions practicing forestry. Removing these residues from a harvest site reduces the carbon stock of the forest compared with conventional stem-only harvest because less litter in left on the site. The indirect carbon dioxide (CO_2) emission from producing bioenergy occur when carbon in the logging residues is emitted into the atmosphere at once through combustion, instead of being released little by little as a result of decomposition at the harvest sites. In this study (1) we introduce an approach to calculate this indirect emission from using logging residues for bioenergy production, and (2) estimate this emission at a typical target of harvest residue removal, i.e. boreal Norway spruce forest in Finland. The removal of stumps caused a larger indirect emission per unit of energy produced than the removal of branches because of a lower decomposition rate of the stumps. The indirect emission per unit of energy produced decreased with time since starting to collect the harvest residues as a result of decomposition at older harvest sites. During the 100 years of conducting this practice, the indirect emission from average-sized branches (diameter 2 cm) decreased from 340 to 70 kg CO_2 eq. MWh^{-1} and that from stumps (diameter 26 cm) from 340 to 160 kg CO_2 eq. MWh^{-1}. These emissions are an order of magnitude larger than the other emissions (collecting, transporting, etc.) from the bioenergy production chain. When the bioenergy production was started, the total emissions were comparable to fossil fuels. The practice had to be carried out for 22 (stumps) or four (branches) years until the total emissions dropped below the emissions of natural gas. Our results emphasize the importance of accounting for land-use-related indirect emissions to correctly estimate the efficiency of bioenergy in reducing CO_2 emission into the atmosphere.

Keywords: bioenergy, forest harvest residue, indirect emissions, land use, soil carbon, Yasso07

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Introduction

Bioenergy, i.e. energy derived from renewable biomass, is used to replace fossil fuels in energy production in order to decrease greenhouse gas emissions into the atmosphere. The rationale behind this practice is that bioenergy does not cause any net carbon dioxide (CO_2) emissions since the amount of CO_2 released into the atmosphere in combustion is taken up again by the next generation of growing plants (Wihersaari, 2005; Stupak *et al*., 2007; Lattimore *et al*., 2009).

Following this idea, as a means to cut down greenhouse gas emissions, the Council of the European Union (EU) adopted a directive on the promotion of renewable energy, including bioenergy. This directive set targets to produce 20% of the final energy consumption using renewable energy sources in the EU by the year 2020. This target is higher for member states already producing a lot of renewable energy, for example as a by-product of pulping industry. Consequently, the national commitment is 38% for Finland and 49% for Sweden (Directive 2009/28/EC). During the reference year of the directive 2005, renewable sources represented already 28% of the total energy production in Finland and 39% in Sweden, while the EU-average was 11%.

These high targets for renewable energy are increasing the focus on biomass for energy production. Worldwide, this growing interest in bioenergy puts pressure

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on land use changes, including deforestation and consequent conversion of the forest land to energy crop cultivation (Melillo *et al*., 2009).

Recently, indirect CO_2 emissions from bioenergy production associated with these land use changes have caused concern (Searchinger *et al*., 2008, 2009; Melillo *et al*., 2009). These indirect emissions occur when bioenergy production reduces the carbon stocks of biomass or soil. These carbon losses may be remarkable, and it is even possible that replacing fossil fuels with bioenergy increases net greenhouse gas emissions into the atmosphere as a consequence of large indirect emissions. The assumed CO_2 neutrality of biofuels like ethanol and their actual potential to mitigate climate change have already been questioned because of large negative impacts on the carbon stock of soil (Fargione *et al*., 2008; Searchinger *et al*., 2008, 2009; Melillo *et al*., 2009).

It is important to realize that the indirect emissions of bioenergy production are not limited to the cases of land use change but may also be caused by new practices of ecosystem management within the same land use. In countries with extensive forest cover and an already high share of renewable energy, an appealing way to produce bioenergy is to intensify biomass removals from forests. Forested countries Finland and Sweden are pioneers in the field of using forest residues for energy production (Mälkki & Virtanen, 2003). Still, in order to meet the EU commitment of renewable energy, Finland plans to increase the use of logging residues for energy production from $3.6\,\mathrm{Mm^3\,yr^{-1}}$ in 2006 to $12\,\mathrm{Mm^3\,yr^{-1}}$ by 2020 (Ministry of Employment and the Economy of Finland, 2008).

Until now, research on the effects of logging residue removal has focused on nutrient balances (e.g. Wall, 2008; Luiro *et al*., 2009), socioeconomic impacts (e.g. Börjesson, 2000), profitability (e.g. Heikkilä *et al*., 2007), forest productivity (e.g. Peng *et al*., 2002) and properties of wood fuel (e.g. Alakangas, 2005), whereas the indirect emissions have received little attention. Lattimore *et al*. (2009) dealt with the indirect emissions to some extent in their recent review. They concluded that, in order to be sustainable, bioenergy production from forest residues must not have adverse effects on soil quality, hydrology and water quality, site productivity, or forest biodiversity but also not on greenhouse gas balances.

The indirect emissions from removing forest harvest residues, and using them for energy production, result from combusting the residues and releasing CO_2 into the atmosphere soon after harvesting instead of letting them decompose slowly at the harvested site. As a consequence of such practice, the amount of carbon stored at the forest site decreases, possibly to a remarkable degree.

There are some field studies (Johnson *et al*., 2002; Jones *et al*., 2008) and model-based calculations (Palosuo *et al*., 2001; Peng *et al*., 2002; Ågren & Hyvönen, 2003; Eriksson *et al*., 2007) regarding the effect of logging residue removal on the carbon stock of soil. Some of these studies show clearly that intensified removal of harvest residues reduces the soil carbon stock (Johnson & Curtis, 2001; Ågren & Hyvönen, 2003; Eriksson *et al*., 2007). Palosuo *et al*. (2001) estimated that the indirect emissions from decreasing carbon stock are an order of magnitude larger than the other emissions from an energy production chain utilizing forest harvest residues. Despite the significant contribution of indirect emissions to the estimate of total emissions per unit of produced energy, Palosuo *et al*. (2001) calculate emissions of approximately $50\,\mathrm{kg\,CO_2\,eq.\,MWh^{-1}}$, which is 80–90% less than the emissions from various fossil fuels.

The indirect emission of using logging residues for energy production depend critically on the decomposition rate of the residues if they were left at the site. Studies based on extensive sets of measurements have been published recently making it possible to estimate the decomposition rate of the harvest residues more reliably than before (e.g. Tarasov & Birdsey, 2001; Palviainen *et al*., 2004; Mäkinen *et al*., 2006; Vávřová *et al*., 2009). We have used these measurements plus other measurements related to decomposition and carbon cycling in soil and developed a new soil carbon model Yasso07 (Tuomi *et al*., 2008, 2009). The large datasets used and advanced mathematical methods applied make the Yasso07 model particularly suitable for estimating the decomposition rate of woody litter in boreal forests.

In this study, we used this model to estimate the indirect emissions from using logging residues for bioenergy production. The objectives of the current study were to (1) introduce an approach to estimate the indirect CO_2 emissions associated with bioenergy production from forest harvest residues, (2) estimate these emissions in a forested boreal landscape during the first 100 years after starting to produce bioenergy from harvest residues, and (3) compare the total CO_2 emissions per unit of bioenergy produced to the emissions caused by using other fuels.

Materials and methods

Modelling decomposition of forest harvest residues

To estimate the indirect CO_2 emissions from producing bioenergy from forest harvest residues in boreal conditions, we simulated the decomposition of logging residues using a user-interface of the dynamic soil carbon model Yasso07 (Tuomi *et al*., 2008, 2009, http://www.environment.fi/syke/yasso). The measurements of the decomposition of woody litter used to develop the model were taken in Finland and neighboring regions in Estonia and Russia (M. Tuomi, R. Laiho, A. Repo & J. Liski, unpublished results). These measurements used represent the majority of data on woody litter decomposition in this region. The data sets includes branches and stems ranging from 0.5 to 60 cm in diameter, and the mass loss of these woody biomass components has been followed for 1–70 years since the start of decomposition. In addition to these data, the Yasso07 model is based on an extensive data set on decomposition of nonwoody litter across Europe and North and Central America (Tuomi *et al*., 2009) plus data sets on the accumulation and stock of soil organic carbon (Liski & Westman, 1995, 1997; Liski *et al*., 1998). These additional measurements specially provide information on the cycling of recalcitrant organic carbon compounds in soil that are relevant for the long-term carbon balance of decomposing woody litter.

The parameter values of Yasso07 have been sampled using a Markov chain Monte Carlo method (Tuomi *et al*., 2009). This method has been used to make sure that, first, the model is not over-parameterized given the data and, second, there are unequivocal maximum likelihood values for each parameter combination. This mathematical approach and the data available in the development process of Yasso07 make this model suitable for this study because the uncertainty estimates of the model predictions are available and because the data covers the simulated scenarios well without extrapolation.

Model simulations

Using Yasso07, we simulated the decomposition of harvest residues at a typical site of harvest residue removal in Finland, namely an even-aged mature 81–100-year-old Norway spruce (*Picea abies* L.) forest stand located in the Pirkanmaa region of Southern Finland. Spruce stands cover some 40% of forest area in this region (Korhonen *et al*., 2000). Clear-cut spruce stands are favorable targets of harvest residue removals because there are more logging residues than in clear-cut Scots pine stands, which are also common in the region. Spruce stumps are also preferred over pine stumps because they are easier to extract from the soil because of a shallower root structure (Alakangas, 2005; Wihersaari, 2005).

To illustrate differences in decomposition rate between different harvest residues, we simulated decomposition of spruce branches varying from 1 to 5 cm in diameter and stumps varying from 10 to 35 cm in diameter for a 100-year period after the start of decomposition. The mean diameters of spruce branches and stems in the study region, 2 and 26 cm, respectively (Korhonen *et al*., 2000; Kantola *et al*., 2007), were chosen for more detailed analyses of the indirect emissions caused by using the logging residues for bioenergy production. The other input variables of the Yasso07 model used in the simulations are shown in Table 1. When estimating the indirect emissions we assumed that needles were left at the site and only branches or stumps were removed. We assumed also that there was little or no delay in combusting the harvest residues at a power plant and thus CO_2 was released to the atmosphere at once.

Energy production estimation

The indirect CO_2 emissions from using the harvest residues for bioenergy production were taken to be equal to amount of carbon remaining in the harvest residues if the residues were left to decompose at the site harvested. These emissions were also related to the amount of bioenergy produced. The cumulative indirect emissions caused by combusting the harvest residues until year i were calculated by summing up the amounts of carbon left in the harvest residues until this year (i) and relating these emissions to the cumulative amount of bioenergy produced. In other words, we

Table 1 The values of input variables used in the Yasso07 model

Climate variables	
Mean annual temperature	3.2 °C
Temperature amplitude	11.6 °C
Precipitation	681 mm
Chemical composition of woody litter	**Branch ± SD/stump ± SD (%)**
Acid hydrolysable compounds	59 ± 4.3/70 ± 5.0
Water soluble compounds	1 ± 0.3/1 ± 0.2
Ethanol soluble compounds	1 ± 0.3/1 ± 0.2
Klason lignin (neither hydrolysable nor soluble compounds)	37 ± 1.0/28 ± 0.8

The climate values represent averages for southern Finland between 1971 and 2000 (Drebs *et al*., 2002) and the chemical composition averages of several individual studies (Hakkila, 1989). The standard deviation (SD) values of the chemical composition are based on coefficients of variation calculated from a data base of foliage litter (Berg *et al*., 1991). The temperature amplitude means a half of the difference between the mean temperatures of the warmest and the coldest month of the year.

simulated a case where the practice of removing the harvest residues and using them for bioenergy production was started and continued on a harvest area of similar size year after year. We applied biomass-compartment-specific net calorific heating values to estimate the amount of energy obtained by combusting the harvest residues, i.e. 19.30 MJ kg^{-1} for dry Norway spruce branches and 19.18 MJ kg^{-1} for dry stumps (Nurmi, 1997). These values of dry logging residues range commonly from 18 to 20 MJ kg^{-1} (Alakangas, 2005). The carbon content of the harvest residues was assumed to be equal to 50% of dry wood (m/m).

In order to estimate the full fuel cycle emissions from using the logging residues for energy production, we added other emissions from a typical wood chip fuel production chain using harvest residues to the calculated indirect emissions. These emissions result from (1) collecting, chipping, and transporting the harvest residues, (2) emitting methane (CH_4) and nitrous oxide (N_2O) from combustion, (3) fertilizing the forest to compensate for nutrient loss, and (4) recycling ash, and they range typically from 5 to 18 kg CO_2 eq. MWh^{-1} produced (Palosuo *et al*., 2001; Wihersaari, 2005). We used a central value of this range, 12 kg CO_2 eq. MWh^{-1} produced, in our calculations. Other estimates for direct emissions from wood fuel chain range from 5 to 20 kg CO_2 eq. MWh^{-1} (Korpilahti, 1998; Mälkki & Virtanen, 2003; Berg, 2010) depending on background information and included operations in the estimates. Fossil fuel comparison values 280, 306, and 395 kg CO_2 eq. MWh^{-1} were estimates of entire fuel cycle emissions of natural gas, oil, and diesel, and coal (Statistics Finland, 2006; Ecoinvent Centre, 2007).

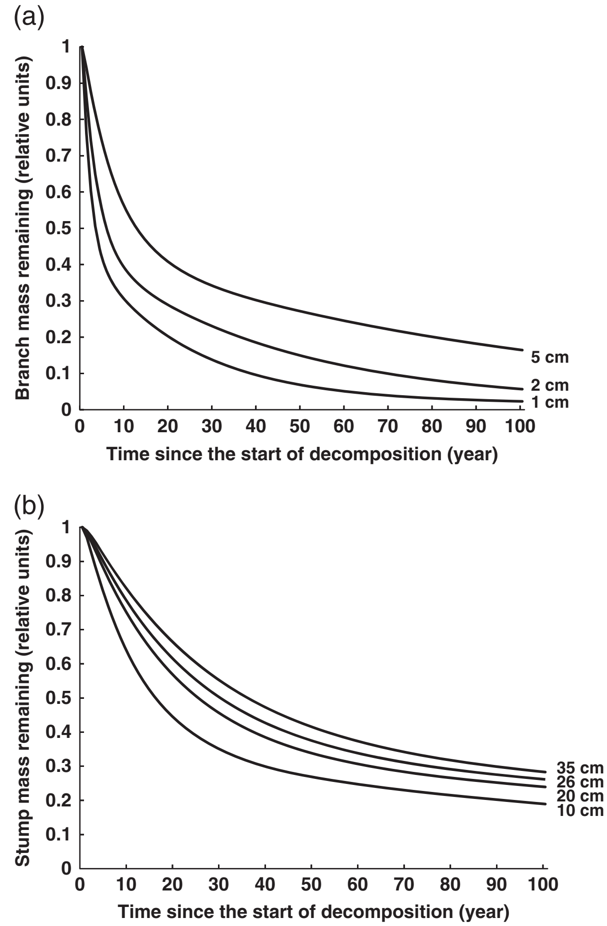

Fig. 1 Mass remaining of decomposing Norway spruce branches (diameter 1–5 cm) and stumps (diameter 10–35 cm) over a 100-year period after the start of decomposition in southern boreal conditions as simulated using the Yasso07 model (model input values in Table 1).

Results

Branches lost mass at a remarkably higher rate than stumps because the simulated decomposition rate of woody litter was dependent on the initial diameter (Fig. 1). For example, after 10 years of decomposition, the branches 1–5 cm in diameter had 30–55% of the initial mass still remaining (Fig. 1a) while stumps 10–35 cm in diameter had 63–81% (Fig. 1b). The simulated rate of mass loss decreased over time, and after 100 years of decomposition there was still 2–16% of the initial branch mass remaining and 19–28% of the initial stump mass remaining.

The indirect CO_2 emissions, caused by combusting logging residues after harvesting instead of letting them decompose at the harvested site, were equal to the CO_2 emissions from combustion, 340 kg CO_2 MWh^{-1}, when the practice was started but these emissions decreased over time as a result of decomposition of the harvest residues (Fig. 2). The indirect emissions of using branches for bioenergy decreased faster than the emissions of using stumps because the branches decomposed faster (Figs 1 and 2). After the first 10 years of conducting this practice, the indirect emissions from branches were equal to 200 kg CO_2 MWh^{-1} and those from stumps 310 kg CO_2 MWh^{-1} (Fig. 2). After the first 100 years, these emissions from the branches and stumps were equal to 70 and 160 kg CO_2 MWh^{-1}, respectively.

The estimated greenhouse gas emissions from the rest of the bioenergy production chain, i.e. the emissions from collecting, transporting, chipping, and combusting the harvest residues plus the emissions from fertilizing the forest and recycling the ash, were equal to 12 kg CO_2 eq. MWh^{-1} (Fig. 3). At the time of starting this practice, these direct emissions represented only 3% of the total emissions caused by using the harvest residues

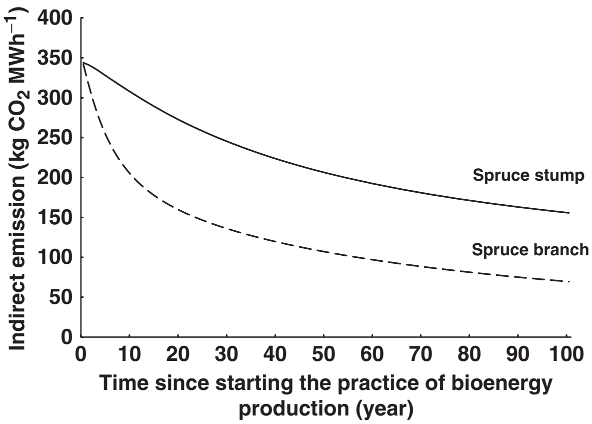

Fig. 2 Indirect CO_2 emission per unit of energy produced from using Norway spruce stumps (diameter 26 cm) or branches (diameter 2 cm) for bioenergy over a 100-year period after starting this practice. This indirect emission is equal to the carbon stock of woody litter lost in combustion per energy obtained until each year.

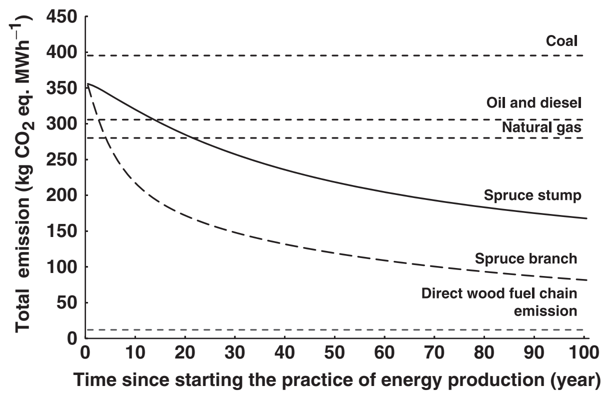

Fig. 3 The total greenhouse gas emission per unit of energy produced from using Norway spruce branches (diameter 2 cm) or stumps (diameter 26 cm) for bioenergy over a 100-year period after starting this practice and the total emissions from various fossil fuels. The emission estimates of bioenergy production include both an indirect (see Fig. 2) emission resulting from decreasing carbon stock and a direct wood fuel chain emission (equal to 12 kg CO_2 eq. MW h^{-1}) resulting from collecting, chipping and transporting the harvest residues, CH_4 and N_2O emissions from combusting the residues, fertilizing the forest to compensate for nutrient loss, and recycling of ash. The estimates of the fossil fuels represent entire fuel cycle emissions (Statistics Finland, 2006; Ecoinvent Centre, 2007).

for bioenergy. After the 100-year period of conducting this practice, this share increased to 15% if bioenergy was produced from branches and to 7% if bioenergy was produced from stumps (Fig. 3). The increased contribution of the direct emissions was a result of the decreased indirect emissions (Fig. 2).

At the time of starting to use the harvest residues for energy production, the total emissions were comparable to the emissions caused by using fossil fuels (Fig. 3). After 10 years of producing bioenergy from branches, the total emissions caused were 210 kg CO_2 eq. MWh^{-1}. These emissions are 24%, 30%, or 46% lower than the emissions caused by producing the energy from natural gas, oil, or coal, respectively. If bioenergy was produced from stumps, it took 22 years for the average emissions to decrease below the emissions of producing the energy from natural gas or 14 years for the emissions to decrease below the emissions of oil. After 100 years of producing energy by combusting branches, the emissions were lower by 71%, 74%, or 79% than the emissions caused by producing the energy from natural gas, oil, or coal, respectively. For bioenergy produced by combusting stumps, these percentages of emission reductions were 40%, 46%, or 58%, over 100 years, compared with the emissions of natural gas, oil, or coal, respectively.

Discussion

Using logging residues for energy production decreases the amount of carbon stored in forest and causes thus indirect CO_2 emissions into the atmosphere. These indirect emissions occur because combustion releases carbon of logging residues to the atmosphere at once, otherwise the residues would form a slowly decomposing carbon stock at the harvest site. The indirect emissions are an order of magnitude larger than the direct emissions from the rest of this bioenergy production chain (Fig. 3), not accounting for the CO_2 emissions of combusting the harvest residues. The indirect emissions depend on the decomposition rate of the harvest residues, with the decomposition rate being lower with an increasing size of woody litter (Fig. 1, Harmon *et al.*, 1986; Janisch *et al.*, 2009; M. Tuomi, R. Laiho, A. Repo & J. Liski, unpublished results). The use of bigger-sized stumps for energy production causes therefore larger indirect emissions than the use of smaller-sized branches (Fig. 2). The average indirect emissions per unit of energy produced decrease over time since the start of this form of bioenergy production (Fig. 2) but during the first few years, or a couple of decades in the case of stumps, the total emissions caused by energy production from harvest residues are comparable to the emissions of fossil fuels (Fig. 3).

The reliability of the current results depends critically on the decomposition estimates of the harvest residues. The Yasso07 decomposition and soil carbon model used in this study is based on a large collection of mass loss measurements taken on woody litter in boreal forests across Finland and neighboring countries plus large sets of other relevant measurements from across the world (see 'Materials and methods'). This model has been shown to give unbiased estimates for the decomposition of woody (M. Tuomi, R. Laiho, A. Repo & J. Liski, unpublished results) and nonwoody litter (Tuomi *et al*., 2009). Compared with other studies on the decomposition of woody litter carried out under comparable conditions, the estimates of this study are similar except for the end of the 100-year study period; for this late phase of decomposition, the current estimates of mass remaining are higher. Mäkinen *et al*. (2006) estimated that Norway spruce stems lost about 20% of their initial mass during the first 10 years of decomposition and the stems disappeared completely in some 60–80 years of time. On the other hand, according to the residuals reported by Mäkinen *et al*. (2006), their model seems to underestimate the mass remaining after 30 years of decomposition. Melin *et al*. (2009) compared several decomposition models developed for Norway spruce logs, snags, and stumps. They concluded that after 10 years of decomposition some 60–75% of the initial mass was still remaining whereas after 100 years of decomposition practically none or $<10\%$ of the initial mass was still left. Our higher estimates of mass remaining during the late phases of decomposition can be explained by a difference between the methods used. The two earlier studies were based on measurements of woody litter mass remaining. These measurements may not capture the formation and translocation of well-decayed soil organic matter originating from woody litter. The Yasso07 model, on the other hand, is additionally based on measurements of formation of soil organic matter (Tuomi *et al*., 2009). For this reason, we think that the estimates of Yasso07 model are probably more realistic for the late phases of woody litter decomposition.

In addition to the decomposition estimates, the reliability of the current results depends also on the other parameters of our calculations such as the reference energy systems or combustion techniques chosen as well as variation in the chemical composition of litter (affecting the decomposition rate estimates of the harvest residues) and calorific values. Furthermore, the simulations were done for climate conditions prevailing in Southern Finland today. The results do not thus account for the effects of climate change or those of increased atmospheric CO_2 concentration on forest growth. In addition, forest soil disruption associated with stump removal may release additional CO_2 into the atmosphere. Currently, empirical research on the magnitude of these emissions is few in number (Jandl *et al*., 2007; Walmsley & Godbold, 2010). Hope (2007) found that stump removal together with forest floor scarification reduced soil carbon stocks in the first year of stump harvesting and 9 years later. This conclusion is apparent only if during stump removal the forest soil is completely scarified by removal or mixing with mineral soil. Although the quantitative data on stump extraction and emissions associated with this practice is scarce, generally it is known that soil disturbance can change the microclimate and stimulate the decomposition of litter (Johansson, 1994). In a Finnish study site preparation after a clear-cut with mixing organic matter with mineral soil increased CO_2 efflux from soil but this effect leveled off rapidly (Pumpanen *et al*., 2004). Despite of these uncertainties, we think that the current estimates are reliable enough to demonstrate the magnitude of indirect emissions associated with producing bioenergy from harvest residues in boreal coniferous forest.

The indirect emissions, caused by the reduced carbon stock of decomposing harvest residues, represented 85–97% of the total emissions of this bioenergy production chain, not accounting for the CO_2 emissions from combusting the harvest residues. These emissions are thus highly significant for the full fuel cycle emissions of logging residues. When comparing the present results to earlier ones it is important to acknowledge differences in system boundaries, energy use-technology, reference energy system, conversion technology and type and management of raw material (Cherubini *et al*., 2009). Still, it is possible to conclude that a common outcome of the earlier studies is that the greenhouse gas emissions from logging, collecting, chipping, and transporting harvest residues are relatively low (Börjesson, 1996; Palosuo *et al*., 2001; Mälkki & Virtanen, 2003; Wihersaari, 2005). These estimates have ranged from 4 to $20\,kg\,CO_2\,MWh^{-1}$ depending on the details of the bioenergy production chains studied. The current study shows that if indirect CO_2 emissions are accounted for, the total CO_2 emissions are at least an order of magnitude higher than these emissions which have been considered to represent the total emissions (cf. Mälkki & Virtanen, 2003).

The decreasing effect of logging residue removal on soil carbon stock has been demonstrated earlier but this has not been considered to be problematic as long as this removal practice does not jeopardize the carbon sink of soil (Börjesson, 2000; Ågren & Hyvönen, 2003; Mälkki & Virtanen, 2003; Petersen Raymer, 2006; Eriksson *et al*., 2007; Sievänen *et al*., 2007; Eriksson & Gustavsson, 2008). Sievänen *et al*. (2007) calculated that increasing the removals of logging residues from 4 to $15\,Mm^3\,yr^{-1}$ in Finland will not turn the Finnish forests from net carbon sinks to net sources. However, the intensified removals of the logging residues would decrease the annual carbon sink of these forest soils by 3.1 million tons of CO_2 eq. (Sievänen *et al*., 2007). Assuming that $1\,m^3$ of harvest residues gives 2 MWh of energy (Alakangas, 2005), the indirect emission from the decreasing carbon stock of soil is equal to about 100 kg CO_2 eq. MWh^{-1}. This estimate is somewhat lower than the estimates of the present study because Sievänen *et al*. (2007) applied an earlier version of Yasso soil carbon and decomposition model in their study (Liski *et al*., 2005). This earlier model version gave less reliable, higher estimates for the decomposition rate of woody litter because it was based on a substantially smaller number of measurements. Nevertheless, these figures demonstrate that it is important to relate the indirect and direct emissions of bioenergy production to the amount of energy obtained in order to get a correct picture on the efficiency of using different energy sources in decreasing greenhouse gas emissions to the atmosphere. The current results demonstrate that if the indirect CO_2 emissions are counted in, the total fuel chain emissions from using spruce branches or stumps for energy production may cause even bigger CO_2 emissions during the first years or decades of starting this practice than producing energy from oil or natural gas.

The indirect greenhouse gas emissions resulting from using logging residue for bioenergy production are highest per unit of energy produced immediately when the practice is started. The average emissions per energy unit decrease, however, over time. As a result of this temporal pattern, this form of bioenergy production is not efficient in decreasing emissions to the atmosphere in the near future. Our results stress the importance of considering time perspective when assessing the potential of different bioenergy options to mitigate climate change (Schlamadinger & Marland, 1996; Petersen Raymer, 2006). The issue of which temporal approach is appropriate depends on the management and policy strategies and whether the selection of energy systems is made to meet long-term or short-term greenhouse gas reduction objectives (Schlamadinger *et al*., 1997).

It is possible to reduce the indirect emissions of logging residue removals by collecting quickly decomposing harvest residues for bioenergy production, for example branches instead of stumps. However, it may be still tempting to extract stumps from harvest sites because the gain of primary energy per hectare may be twice that compared with collecting branches (Eriksson & Gustavsson, 2008). Leaving the needles at the harvest site, which helps to avoid nutrient loss, has a marginal effect on the carbon balance at clear-cut sites, although needle and fine root litter produce more than two-thirds of the soil carbon stock in growing forests (Ågren & Hyvönen, 2003).

The indirect emissions have a remarkable effect on the total greenhouse gas emissions from some systems of bioenergy production, as demonstrated in this study for harvest residues of boreal forests, and emphasized for several other systems by Johnson (2009) and Searchinger *et al*. (2009). For this reason, to account for the actual greenhouse gas effect of various alternative bioenergy options, it would be essential to include the indirect emissions adequately in guidelines of greenhouse gas inventorying and reporting. Currently, the rules of carbon accounting applied under the Kyoto Protocol do not count all indirect emissions, which among other things distorts the accounting of net emissions (Johnson, 2009; Searchinger *et al*., 2009). A particular shortcoming of the accounting rules under the Kyoto Protocol is that a party to this protocol may choose not to account for changes in one or several of the agreed carbon pools (aboveground biomass, belowground biomass, litter, dead wood, and soil organic carbon) as long as it can reliably show that the pool is not decreasing. Owing to this threshold, some of the indirect emissions caused by using logging residues for bioenergy production may be excluded from the inventory figures. On the other hand, in inventory reports of greenhouse gases under the United Nations Framework Convention on Climate Change, the carbon release resulting from forest harvesting must be counted and reported as a land-use emission or as a energy emission but not both (IPCC, 2000). Today, the land-use-related greenhouse gas emissions from bioenergy production systems are recognized and investigated (IPCC, 2000; Melillo *et al*., 2009) but one of the practical problems is that measuring methods for the indirect effects and feasible means to bring these impacts to regulatory policies are still lacking (Mathews & Tan, 2009).

Conclusions

Using logging residues as a source of bioenergy causes net CO_2 emissions into the atmosphere and a great majority (85–97%) of these emissions are indirect emissions resulting from a decline in the carbon stock of harvest residues in forest. The amount of the indirect emissions increases with a decreasing decomposition rate of the harvest residues. Norway spruce stumps decay at slower rate than branches and consequently the energy use of the stumps causes 1.5–2 times larger indirect emissions than the use of branches. Production of bioenergy from forest harvest residues causes emissions that are comparable to the emissions of fossil fuel over the first few years (branches) or first few decades (stumps) of the practice. After 50 years, bioenergy produced of Norway spruce stumps decreases average emissions per unit of energy produced by some 20% and bioenergy from branches by some 60% compared with entire fuel cycle emissions of natural gas.

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INVITED EDITORIAL

Large-scale bioenergy from additional harvest of forest biomass is neither sustainable nor greenhouse gas neutral

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Abstract

Owing to the peculiarities of forest net primary production humans would appropriate ca. 60% of the global increment of woody biomass if forest biomass were to produce 20% of current global primary energy supply. We argue that such an increase in biomass harvest would result in younger forests, lower biomass pools, depleted soil nutrient stocks and a loss of other ecosystem functions. The proposed strategy is likely to miss its main objective, i.e. to reduce greenhouse gas (GHG) emissions, because it would result in a reduction of biomass pools that may take decades to centuries to be paid back by fossil fuel substitution, if paid back at all. Eventually, depleted soil fertility will make the production unsustainable and require fertilization, which in turn increases GHG emissions due to N_2O emissions. Hence, large-scale production of bioenergy from forest biomass is neither sustainable nor GHG neutral.

Keywords: bioenergy, biomass, ecosystem function, forestry, greenhouse gas emission, human appropriation of net primary production

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Climate change impacts resulting from fossil fuel combustion challenge humanity to find energy alternatives that would reduce greenhouse gas (GHG) emissions. One important option in this context is bioenergy. There is a wealth of literature on actual yields of different energy crops and production systems (WBGU, 2009; NRC, 2011). Beringer *et al.* (2011) estimate that 15–25% of global primary energy could come from bioenergy in the year 2050. A prominent recent assessment suggested that bioenergy provision could even be up to 500 EJ yr^{-1}, more than current global fossil energy use (Chum *et al.*, 2012) and that GHG mitigation could be sustained under future climate conditions (Liberloo *et al.*, 2010).

Western and developing countries are on a course to increase bioenergy production substantially. For example, the United States enacted the Renewable Fuels Standard as part of the 2005 Energy Policy Act and amended it in 2007, mandating the use of renewable fuels for transportation from 2008 to 2022 and beyond. In addition, 20% of all EU energy consumption is to come from renewable sources by 2020 with bioenergy as a focal point in this effort (COM, 2006a). In 2005, the European Commission adopted the Biomass Action Plan (COM, 2005) and in 2006 the Strategy for Biofuels (COM, 2006b), both of which aim to increase the supply and demand for biomass. Strategies that could substantially diminish our dependence on fossil fuels without competing with food production include substitution with bioenergy from forests (Tilman *et al.*, 2009), either by direct combustion near the source or by conversion to cellulosic ethanol. There are important questions about GHG reduction, economic viability, sustainability and environmental consequences of these actions.

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Greenhouse gas reduction

The general assumption that bioenergy combustion is carbon-neutral is not valid because it ignores emissions due to decreasing standing biomass and contribution to the land-based carbon sink. The notion of carbon-neutrality is based on the assumption that CO_2 emissions from bioenergy use are balanced by plant growth, but this reasoning makes a 'baseline error' by neglecting the plant growth and consequent C-sequestration that would occur in the absence of bioenergy production (Searchinger, 2010; Hudiburg *et al.*, 2011), and it ignores the fact that fossil fuels are needed for land management, harvest and bioenergy processing.

Recent life cycle assessments cast doubt on the existence of emission savings of bioenergy substitution from forests. In the Pacific Northwest United States, policies are being developed for broad-scale thinning of forests for bioenergy production, with the assumed added benefit of minimizing risk of crown fires. This includes forests of all ages and thus timeframes of biomass accumulation. However, a recent study suggests that more carbon would be harvested and emitted in fire risk reduction than would be emitted from fires (Hudiburg *et al.*, 2011). Furthermore, policies allow thinning of mesic forests with long fire return intervals, and removal of larger merchantable trees to make it economically feasible for industry to remove the smaller trees for bioenergy. These actions would lead to even larger GHG emissions beyond those of contemporary forest practices (Hudiburg *et al.*, 2011).

Increased GHG emissions from bioenergy use are mainly due to consumption of the current carbon pool and from a permanent reduction of the forest carbon stock resulting from increased biomass harvest (Holtsmark, 2011). When consumption exceeds growth, today's harvest is carbon that took decades to centuries to accumulate and results in a reduction of biomass compared to the current biomass pool (Holtsmark, 2011; Hudiburg *et al.*, 2011). Hence, it is another example of 'slow in and fast out' (Körner, 2003). Consequently, reduction in forest carbon stocks has been shown to at least cancel any GHG reductions from less use of fossil fuel over decadal time spans (Haberl *et al.*, 2003; Mc-kechnie *et al.*, 2011). Boreal forests with relatively low carbon sequestration potential may take centuries before permanent reduction of the carbon stocks resulting from increased bioenergy harvest is repaid by reduced emissions from fossil fuels (Holtsmark, 2011). For more productive temperate regions, an infinite payback time was found implying that lower GHG emissions are achieved through C-sequestration in forests rather than through bioenergy production (Hudiburg *et al.*, 2011).

Recent studies of the differences in timing of CO_2 emissions from bioenergy production and forest carbon uptake (Cherubini *et al.*, 2011a,b) suggest that the 'upfront' CO_2 emitted during biomass harvest and combustion stays in the atmosphere for decades before the CO_2 is removed by the growing forest. It results in a 'pulse' of warming in the first decades of bioenergy implementation. This contrasts calls for a rapid reduction of the growth rate of climate forcing (Friedlingstein *et al.*, 2011) required to achieve the policy of limiting warming to 2 °C.

The initially reported emission savings from forest bioenergy are based on erroneous assumptions in the accounting schemes. Studies that corrected these errors suggest that forest management that reduces the current biomass pool is unlikely to result in the envisioned emissions savings at all, and certainly not over the next decades.

Economic viability

Emerging technologies such as biofuel refineries and combined heat and power plants have to compete against established technologies applied in coal, gas and nuclear power plants. In the United States, a recent National Research Council report concluded that only in an economic environment characterized by high oil prices (e.g. >$191 per barrel), technological breakthroughs (cellulosic ethanol) and at a high implicit or actual carbon price would biofuels be cost-competitive with petroleum-based fuel (NRC, 2011). Hence, incentives favouring bioenergy (i.e. production quota, subsidies, tax cuts) will be needed to complement or even replace fossil fuel-based technologies (Schneider & Kaltschmitt, 2000; Ryan *et al.*, 2006; Ahtikoski *et al.*, 2008; NRC, 2011).

Schemes favouring the economics of one practice or technology over another often lead to unanticipated side-effects. For example, side-effects have been documented for the Common Agricultural Policy of the European Union (Macdonald *et al.*, 2000; Stoate *et al.*, 2001), and forest-based bioenergy production would seem to be similar. In Germany, where bioenergy is subsidized, the market price for woody biomass increased from 8 to 10 € m^{-3} in 2005 to 46 € m^{-3} for hardwood and 30–60 € m^{-3} for coniferous wood in 2010. Prices for woody biomass for bioenergy now reach 60–70% of saw log prices (Waldbesitzerverband, 2010; wood sales by one of the authors). Such prices discourage the production of quality timber and make root extraction and total tree use attractive options despite the documented unfavourable effects on soil carbon, soil water and nutrient management (Johnson & Todd, 1998; Johnson & Curtis, 2001; Burschel & Huss, 2009; Peckham & Gower, 2011).

For the German example, the price increase is driven by the installation of distributed bioenergy plants and the competitive market of other uses for biomass, such as wood for production of cellulose. Although the details will differ among regions and countries, increasing imports by developed nations is the most likely response to an increasing wood demand (Seintsch, 2010), because total wood harvest has not substantially changed in the developed world (i.e. ~1.4×10^9 m^3

between 1990 and 2010 in Europe and North America, FAO, 2010). Increased imports are likely to be met through land-use (intensity) change in other regions (lateral transfer of emissions). In the case of increased imports, these are most likely met by harvesting previously unmanaged forests or forest plantations. Thus, similar to crop-based production systems, forest-based bioenergy requires additional land, contrary to previous expectations (Tilman *et al*., 2009). Increased wood imports, thus, represent a global footprint of local energy policies and should be accounted for in life cycle assessment of wood-based bioenergy.

Reduced manufacturing residue losses and other technological advances such as glued wood-based elements initiated a trend towards shorter rotations and thus younger forests. However, the economics of bioenergy production supported by existing subsidy schemes is expected to reduce rotation length to its lowest limit and promote questionable management practices and increased dependency on wood imports. Further, high prices for biomass will discourage forest owners from investments in long rotations, resulting in a shortage of quality timber. Given the time required to produce high-quality timber, such shortage cannot be remedied by short-term (economic) incentives.

Environmental consequences

Homogeneous young stands with a low biomass resulting from bioenergy harvest are less likely to serve as habitat for species that depend on structural complexity. It is possible that succession following disturbance can lead to young stands that have functional complexity analogous to that of old forests; however, this successional pathway would likely occur only under natural succession (Donato *et al*., 2011). A lower structural complexity, and removal of understory species, is expected to result in a loss of forest biodiversity and function. It would reverse the trend towards higher biomass of dead wood (i.e. the Northwest Forest Plan in the United States) to maintain the diversity of xylobiontic species.

Cumulative impacts of bioenergy-related management activities that modify vegetation, soil and hydrologic conditions are likely to influence erosion rates and flooding and lead to increased annual runoff and fish habitat degradation of streams (Elliot *et al*., 2010). Young uniform stands with low compared to high standing biomass have less aesthetic value for recreation (Tahvanainen *et al*., 2001) and are less efficient in avalanche control and slope stabilization in mountains owing to larger and more frequent cutting (Brang, 2001). A potential advantage is that younger forests with shorter rotations offer opportunities for assisted migration, although there is great uncertainty in winners and losers (species, provenances, genotypes) in a future climate (Larsen, 1995; Millar *et al*., 2007; Pedlar *et al*., 2011). Plantations, however, largely contribute to pathogen spread, such as rust disease (Royle & Hubbes, 1992).

Forests offer several important ecosystem services in addition to biomass and some would be jeopardized by the bioenergy-associated transition from high to low standing biomass. Agriculture provides a visible example for abandoning most ecosystem services except biomass production (Foley *et al*., 2005); communities in intensive agricultural regions often rely on (nearby) forested water sheds for drinking water, recreation and offsetting GHG emissions from intensive agriculture (Schulze *et al*., 2009).

Sustainability

From a historical perspective, a transition from forest biomass burning to fossil fuels literally fuelled the industrial revolution, and consequently, caused rapid climate change. However, the collapse of biomass use enabled the recovery of largely degraded forest ecosystems (Gingrich *et al*., 2007). Partly due to recovery from previous (mis)use, C-sequestration is especially strong over Europe (Ciais *et al*., 2008; Luyssaert *et al*., 2010) and the United States (Williams *et al*., 2011). As such, C-sequestration can be considered a side-effect of the transition of energy sources from wood to fossil fuels (Erb *et al*., 2008). Industrial-scale use of forest biomass for energy production would likely reverse this trend or at least reduce the carbon sink strength of forests (Haberl *et al*., 2003; Holtsmark, 2011; Hudiburg *et al*., 2011). The historical forest resource use in Europe and the United States is the present day situation in Africa. For example, southern African *miombo* forests have been degraded into shrubland as a result of charcoal production, where charcoal is the main energy source for rural communities even at a very low level of total energy consumption (Kutsch *et al*., 2011).

A widespread misconception is that the most productive forests are necessarily the strongest carbon sinks. Actually, net primary productivity of forests is typically negatively correlated with the cumulative amount of carbon stored in biomass (Fig. 1). In reality, old forests show lower NPP but store the largest amount of carbon (Luyssaert *et al*., 2008; Hudiburg *et al*., 2009; Bugmann & Bigler, 2011) because slow growing forest live longer than fast growing forest (Schulman, 1954; Bigler & Veblen, 2009). Hence, on areas currently forested, any fast rotation management and use for fossil fuel substitution is reducing forest carbon sequestration. At regional scales, a permanent increase in annual wood harvest results in a permanent reduction in the amount of

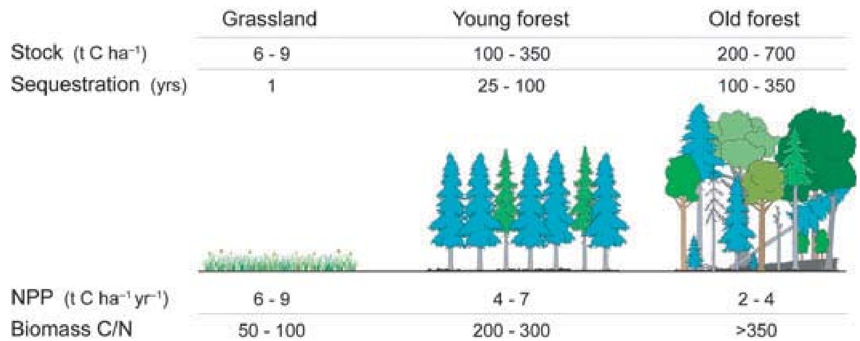

Fig. 1 Land management trade-off: maximizing productivity vs. carbon stocks. Given fixed resource availability, land managers can maintain highly productive ecosystems with a low standing biomass such as grasslands. The dominant tissues are leaves and roots with a low C/N ratio (~50). The same resources could be used to grow forest. With time forest accumulate considerable amounts of carbon in their biomass but forest that grow old have a lower net primary production than young forest and grasslands. Woody biomass has high C/N ratios (~400) and with an increasing share of woody biomass in the total biomass, the C/N ratio of the ecosystem decreases. Consequently, the time integral of productivity will be lower for an old forest compared with grassland, but at the same time, the time integral of nitrogen export will be lower for an old forest (closed nitrogen cycle) compared with a grassland (open nitrogen cycle). Hence, increasing the biomass pool size is the sustainable way of capitalizing from forests in the C-sequestration vs. C substitution debate. Ranges in the figure are for temperate ecosystems based on (Van Tuyl *et al.*, 2005; Luyssaert *et al.*, 2007, 2008; Schulze *et al.*, 2009; Keith *et al.*, 2009).

carbon stored in forests at the regional scale due to a lower average stand age (Körner, 2009; Holtsmark, 2011).

Globally, ~7% of global forest net primary production (NPP) outside wilderness areas is used by humans annually (Haberl *et al.*, 2007a). In Europe, human appropriation of forest NPP reaches ~15% (Luyssaert *et al.*, 2010). Thus, even in the absence of industrial production of wood-based bioenergy, humans already seize a remarkable share of forest production. To produce 20% of current primary energy consumption from wood-based bioenergy, as suggested by policy targets, it would require more than doubling the global human appropriation of NPP (HANPP) to 18–21% (Table 1; ratio of row 1 and 6). Such an increase in human appropriation would have serious consequences for global forests. Due to its nature, much of forest NPP cannot be harvested, e.g. fine root NPP, NPP for mycorrhizal associations and NPP in volatile organic emissions. Further, forests are harvested after decades of growth; hence, much of the NPP is already consumed by herbivores, added to the litter pool or decomposed in the detritus food chains long before harvest, e.g. leaves, fruits, fine

Table 1 Global HANPP in forests in the year 2000 and future HANPP that would result from providing 20% of world primary energy from forest harvest. NPP denotes net primary production and HANPP the human appropriation of net primary production. Using a gross caloric value of 19 kJ g^{-1} forest biomass or 38 kJ g^{-1} biomass carbon and a net caloric value of 41.9 GJ for 1 ton of oil equivalent. Conversion from net to gross calorific value was based on the following multipliers (gross/net): coal 1.1, oil 1.06, natural gas 1.11 and biomass 1.1 (Haberl *et al.*, 2006)

	Global C-flux (PgC yr^{-1})	Energy equivalent (EJ yr^{-1})	Source
(1) Current NPP of forest ecosystems	27–29	1030–1100	Haberl *et al.* (2007a) and Pan *et al.* (2011)
(1a) Belowground NPP (40%)	10–11	–	Luyssaert *et al.* (2007)
(1b) Leaf + twigs NPP (30%)	8.4–8.7	–	Luyssaert *et al.* (2007)
(1c) Aboveground woody NPP (30%)	8.4–8.7	330	Luyssaert *et al.* (2007)
(2) Primary energy use in 2006–2008	–	550	IEA (2008) and BP (2009)
(3) Global fossil energy use in 2006–2008	6–7	450	IEA (2008) and BP (2009)
(4) Additional fuel wood to produce 20% of primary energy	2.3	87	From 3 and 5
(5) NPP lost in harvest (10–30%)	0.5–1.4	19–53	From 2 and 6
(6) New HANPP level in forests	4.4–5.3	170–200	From 2, 6 and 7

roots, mycorrhiza and plants in early succession stages. Last, part of the NPP could be harvested but typically has no economic value, e.g. perennials, mosses and lichens. Consequently, the maximum HANPP is about 30% of the total NPP; hence, the proposed HANPP of 18–21% already represents ca. 60% of the global increment of woody biomass (Table 1; ratio of rows 1c and 6). Note that our maximum level of harvestable increment of woody biomass is most likely overestimated because the estimate did not account for economic (e.g. distance to population centre), logistic (e.g. steep mountain slopes) and legal (e.g. conservation areas) constraints on harvest. In addition to the increased GHG emissions that would result from such a programme due to reduced biomass stocks (see above), this increase in human appropriation of forest production would likely contribute to forest biodiversity loss, according to recent evidence on the correlation between HANPP and species richness (Haberl *et al*., 2005, 2007b).

Typically, the most fertile lands are in urban and agricultural use (Scott *et al*., 2001), leaving the poorer soils for forest use. The industrial-scale of envisioned forest bioenergy production would export substantial amounts of nutrients, further depleting the soil nutrient stock, particularly if wood removal includes relatively nutrient-rich biomass residues (slash) and root stocks (Peckham & Gower, 2011) as for total tree use. Nutrient and cation losses would have to be compensated for by fertilization, which in turn increases GHG emissions and increases N and P levels in nearby rivers leading to eutrophication of aquatic ecosystems (for a crop related example see Secchi *et al*., 2011).

A persistent 60–70% appropriation of woody biomass increment for bioenergy production from forest harvest over decades will erode current biomass pools, lower average stand age, deplete soil fertility and could thus only be sustained by amendments to nitrogen and phosphorous-depleted soils, activities that also produce GHG (N_2O) emissions.

Conclusion

Although bioenergy from forest harvest could supply ~20% of current energy consumption, this would increase human appropriation of NPP in forests to ~20% which is equivalent to 60–70% of the global increment in woody biomass. We argue that the scale of such a strategy will result in shorter rotations, younger forests, lower biomass pools and depleted soil nutrient capital. This strategy is likely to miss its main objective to reduce GHG emissions because depleted soil fertility requires fertilization that would increase GHG emissions, and because deterioration of current biomass pools requires decades to centuries to be paid back by fossil fuel substitution, if paid back at all. Further, shorter rotations would simplify canopy structure and composition, impacting ecosystem diversity, function and habitat. In our opinion, reasonable alternatives are afforestation of lands that once carried forests and allowing existing forests to provide a range of ecosystem services. Yet, on arable or pasture land, such a strategy would compete with food and fodder production. Society should fully quantify direct and indirect GHG emissions associated with energy alternatives and associated consequences prior to making policy commitments that have long-term effects on global forests. Reasonable alternatives for reducing GHG emissions on the order of the proposed bioenergy substitution include increased energy efficiency and reduced waste of energy via technological improvements and behaviour modification. There is a substantial risk of sacrifying forest integrity and sustainability for maintaining or even increasing energy production with no guarantee to mitigate climate change.

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complement prior studies that highlight the importance of short- and medium-lived pollutants (*14–17*).

The top 10 pollutant-generating activities contributing to net RF (positive RF minus negative RF) in year 20 are shown in the bottom chart, page 526), which takes into account the emission of multiple pollutants from each source activity (*18*). The seven sources that appear only on the left side (purple bars) would be overlooked by mitigation strategies focusing exclusively on long-lived pollutants.

The distinctly different sources of near-term and long-term RF lend themselves to the aforementioned two-pronged mitigation approach. This decoupling is convenient for policy design and implementation; whereas the importance of long-term climate stabilization is clear, the perceived urgency of near-term mitigation will evolve with our knowledge of the climate system. Additionally, optimal near-term mitigation strategies will reflect decadal oscillations (*19*), seasonal and regional variations (*20*, *21*), and evolving knowledge of aerosol-climate effects (*22*, *23*) and methane-atmosphere interactions (*22*)—considerations unique to the near term.

Thus, short- and medium-lived sources (black carbon, tropospheric ozone, and methane) must be regulated separately and dynamically. The long-term mitigation treaty should focus exclusively on steady reduction of long-lived pollutants. A separate treaty for short- and medium-lived sources should include standards that evolve based on periodic recommendations of an independent international scientific panel. The framework of "best available control technology" (strict) and "lowest achievable emissions rate" (stricter) from the U.S. Clean Air Act (*24*) can be used as a model.

Such a two-pronged institutional framework would reflect the evolving scientific understanding of near-term climate change, the scientific certainty around long-term climate change, and the opportunity to separately adjust the pace of near-term and long-term mitigation efforts.

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CLIMATE CHANGE

Fixing a Critical Climate Accounting Error

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Rules for applying the Kyoto Protocol and national cap-and-trade laws contain a major, but fixable, carbon accounting flaw in assessing bioenergy.

The accounting now used for assessing compliance with carbon limits in the Kyoto Protocol and in climate legislation contains a far-reaching but fixable flaw that will severely undermine greenhouse gas reduction goals (*1*). It does not count CO_2 emitted from tailpipes and smokestacks when bioenergy is being used, but it also does not count changes in emissions from land use when biomass for energy is harvested or grown. This accounting erroneously treats all bioenergy as carbon neutral regardless of the source of the biomass, which may cause large differences in net emissions. For example, the clearing of long-established forests to burn wood or to grow energy crops is counted as a 100% reduction in energy emissions despite causing large releases of carbon.

Several recent studies estimate that this error, applied globally, would create strong incentives to clear land as carbon caps tighten. One study (*2*) estimated that a global CO_2 target of 450 ppm under this accounting would cause bioenergy crops to expand to displace virtually all the world's natural forests and savannahs by 2065, releasing up to 37 gigatons (Gt) of CO_2 per year (compa-

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rable to total human CO_2 emissions today). Another study predicts that, based solely on economic considerations, bioenergy could displace 59% of the world's natural forest cover and release an additional 9 Gt of CO_2 per year to achieve a 50% "cut" in greenhouse gases by 2050 (*3*). The reason: When bioenergy from any biomass is counted as carbon neutral, economics favor large-scale land conversion for bioenergy regardless of the actual net emissions (*4*).

The potential of bioenergy to reduce greenhouse gas emissions inherently depends on the source of the biomass and its net land-use effects. Replacing fossil fuels with bioenergy does not by itself reduce carbon emissions, because the CO_2 released by tailpipes and smokestacks is roughly the same per unit of energy regardless of the source (*1*, *5*). Emissions from producing and/or refining biofuels also typically exceed those for petroleum (*1*, *6*). Bioenergy therefore reduces greenhouse emissions only if the growth and harvesting of the biomass for energy captures carbon above and beyond what would be sequestered anyway and thereby offsets emissions from energy use. This additional carbon may result from land management changes that increase plant uptake or from the use of biomass that would otherwise decompose rapidly. Assessing such carbon gains requires the same accounting principles used to assign credits for other land-based carbon offsets.

For example, if unproductive land supports fast-growing grasses for bioenergy, or if forestry improvements increase tree growth rates, the additional carbon absorbed offsets emissions when burned for energy. Energy use of manure or crop and timber residues may also capture "additional" carbon. However, harvesting existing forests for electricity adds net carbon to the air. That remains true even if limited harvest rates leave the carbon stocks of regrowing forests unchanged, because those stocks would otherwise increase and contribute to the terrestrial carbon sink (*1*). If bioenergy crops displace forest or grassland, the carbon released from soils and vegetation, plus lost future sequestration, generates carbon debt, which counts against the carbon the crops absorb (*7*, *8*).

The Intergovernmental Panel on Climate Change (IPCC) has long realized that bioenergy's greenhouse effects vary by source of biomass and land-use effects. It also recognizes that when forests or other plants are harvested for bioenergy, the resulting carbon release must be counted either as land-use emissions or energy emissions but not both. To avoid double-counting, the IPCC assigns the CO_2 to the land-use accounts and exempts bioenergy emissions from energy accounts (*5*). Yet it warns, because "fossil fuel substitution is already 'rewarded'" by this exemption, "to avoid underreporting . . . any changes in biomass stocks on lands . . . resulting from the production of biofuels would need to be included in the accounts" (*9*).

This symmetrical approach works for the reporting under the United Nations Framework Convention on Climate Change (UNFCCC) because virtually all countries report emissions from both land and energy use. For example, if forests are cleared in Southeast Asia to produce palm biodiesel burned in Europe, Europe can exclude the tailpipe emissions as Asia reports the large net carbon release as land-use emissions.

However, exempting emissions from bioenergy use is improper for greenhouse gas regulations if land-use emissions are not included. The Kyoto Protocol caps the energy emissions of developed countries. But the protocol applies no limits to land use or any other emissions from developing countries, and special crediting rules for "forest management" allow developed countries to cancel out their own land-use emissions as well (*1*, *10*). Thus, maintaining the exemption for CO_2 emitted by bioenergy use under the protocol (*11*) wrongly treats bioenergy from all biomass sources as carbon neutral, even if the source involves clearing forests for electricity in Europe or converting them to biodiesel crops in Asia .

This accounting error has carried over into the European Union's cap-and-trade law and the climate bill passed by the U.S. House of Representatives (*1*, *12*, *13*). Both regulate emissions from energy but not land use and then erroneously exempt CO_2 emitted from bioenergy use. In theory, the accounting system would work if caps covered all land-use emissions and sinks. However, this approach is both technically and politically challenging as it is extremely hard to measure all land-use emissions or to distinguish human and natural causes of many emissions (e.g., fires).

The straightforward solution is to fix the accounting of bioenergy. That means tracing the actual flows of carbon and counting emissions from tailpipes and smokestacks whether from fossil energy or bioenergy. Instead of an assumption that all biomass offsets energy emissions, biomass should receive credit to the extent that its use results in additional carbon from enhanced plant growth or from the use of residues or biowastes. Under any crediting system, credits must reflect net changes in carbon stocks, emissions of non-CO_2 greenhouse gases, and leakage emissions resulting from changes in land-use activities to replace crops or timber diverted to bioenergy (*1*).

Separately, Europe and the United States have established legal requirements for minimum use of biofuels, which assess greenhouse gas consequences based on life-cycle analyses that reflect some land-use effects (*1*, *14*). Such assessments vary widely in comprehensiveness, but none considers biofuels free from land-based emissions. Yet the carbon cap accounting ignores land-use emissions altogether, creating its own large, perverse incentives.

Bioenergy can provide much energy and help meet greenhouse caps, but correct accounting must provide the right incentives.

References and Notes

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The Burning Question: Does Forest Bioenergy Reduce Carbon Emissions? A Review of Common Misconceptions about Forest Carbon Accounting

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ABSTRACT

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Critical errors exist in some methodologies applied to evaluate the effects of using forest biomass for bioenergy on atmospheric greenhouse gas emissions. The most common error is failing to consider the fate of forest carbon stocks in the absence of demand for bioenergy. Without this demand, forests will either continue to grow or will be harvested for other wood products. Our goal is to illustrate why correct accounting requires that the difference in stored forest carbon between harvest and no-harvest scenarios be accounted for when forest biomass is used for bioenergy. Among the flawed methodologies evaluated in this review, we address the rationale for accounting for the fate of forest carbon in the absence of demand for bioenergy for forests harvested on a sustained yield basis. We also discuss why the same accounting principles apply to individual stands and forest landscapes.

Keywords
bioenergy, no-harvest baseline, reference point baseline, carbon sequestration parity, carbon debt repayment, dividend-then-debt, stand versus landscape, plantations

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Interest in industrial-scale bioenergy production using forest biomass is part of a larger movement to reduce climate change by using renewable energy in place of fossil fuels. However, if climate change mitigation is indeed a driver for using forest bioenergy, then this energy source must be assessed for its effects on the greenhouse gas (GHG) concentration in the atmosphere. Misconceptions and errors in methodologies continue to affect this topic, both in the scientific and “gray” literature (e.g., magazines, reports, and opinion letters), despite having been addressed in prominent publications (e.g., Searchinger et al. 2009, Haberl et al. 2012). A common misconception is that forest bioenergy is immediately carbon neutral, with no net GHG emissions as long as the postharvest forest regrows to its preharvest carbon level. From a forest manager's perspective, this logic can be appealing because it appears to fit a sustained yield paradigm. But, as we shall show, this paradigm fails to account for other aspects of bioenergy use needed for proper assessment of its effect on GHG emissions.

The purpose of this review is to present the theory and principles for correctly assessing the GHG effects of forest bioenergy. We discuss common errors that appear in the forest bioenergy literature and explain why, in the absence of forest management to increase forest carbon before bioenergy harvesting, the use of forest bioenergy often increases atmospheric carbon dioxide (CO_2), at least temporarily.

PRINCIPLES OF FOREST BIOENERGY GHG ACCOUNTING

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The primary consideration in GHG accounting for forest bioenergy is to accurately determine the fate of forest biomass in the absence of demand for its use to produce bioenergy. This theme will be repeated throughout this article, because failure to correctly address this consideration is the cause of most errors in forest bioenergy accounting.

When tree biomass is burned for energy production, sequestered carbon is released to the atmosphere, mainly as CO_2. Typical sources of forest biomass for biofuel production include standing live trees, harvest residue, biomass recovered during salvage operations, thinnings and residue from thinning operations, and mill processing residue (e.g., sawdust and wood chips); here and throughout the text, biofuel and bioenergy refer to fuel produced from live or dead biomass and to energy derived from burning of biofuel, respectively. Forest carbon is contained in live trees, understory vegetation, and in aboveground (standing dead trees, down woody debris, and forest floor) and belowground dead organic matter (mineral soil and dead roots). The processes determining changes in carbon pools include growth and mortality of live trees, decomposition of dead organic matter, and its combustion if burned. Tree growth and mortality are the main driving forces determining changes in carbon pools. Live trees transfer carbon to dead organic matter pools through self-pruning and mortality; in turn, dead organic matter pools release carbon to the atmosphere through decomposition. In temperate and boreal forests, the largest amount of carbon in a forest is typically contained in live trees and mineral soil, followed by forest floor, with other pools normally accounting for less than 15% of total forest carbon (Pan et al. 2011).

Given the large amounts of woody biomass that stands accumulate, it is intuitive that carbon accrues as they mature (Figure 1). After a stand-replacing disturbance, stand-level forest carbon stocks usually decrease, because carbon losses from decomposing dead organic matter are temporarily not compensated for by carbon sequestered by live trees that are still small. As trees grow, the pattern of net carbon accumulation is sigmoidal, characterized by initially rapid increases that slow as a stand reaches maturity (Figure 1). The slowdown in stand net carbon accumulation at maturity results from the death of individual trees with ongoing growth distributed among the remaining live trees.

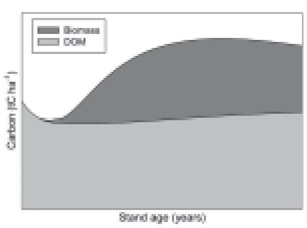

Figure 1.
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Figure 1.

Typical change in forest carbon stocks after stand harvest (modified from Ter-Mikaelian et al. 2014a). Dark and light gray areas represent carbon stocks in live biomass and dead organic matter (DOM), respectively.

Figure 2A shows the accumulation of carbon in live trees in the absence of harvest. Harvesting a stand for bioenergy removes most live tree carbon, leaving unutilized biomass on site, which in traditional harvesting includes stumps, branches and tops, and roots. In temperate and boreal forests, recovery of live tree carbon stocks takes decades because of slow stand regrowth after harvest (Figure 2B). Forest carbon stocks following harvest for bioenergy constitute a *forest bioenergy scenario* (black line in Figure 2B). Forest carbon in the absence of demand for bioenergy represents a *forest baseline scenario* (red line in Figure 2A and D); in some literature reports on forest bioenergy, the forest baseline scenario is referred to as either “business-as-usual,” “counterfactual,” or “protection scenario.”

Figure 2.

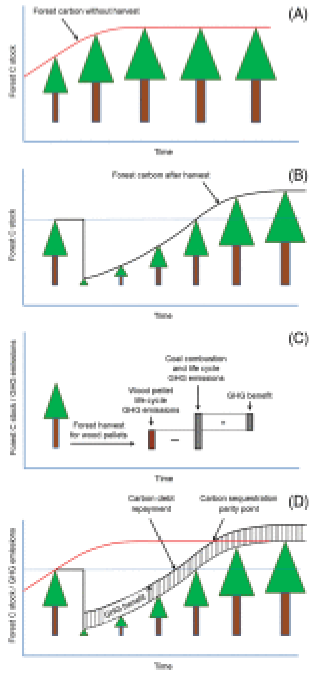

Figure 2.
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Effect of harvest for bioenergy used to replace coal on forest carbon stock changes and total greenhouse gas (GHG) emissions (stand level, from Ter-Mikaelian et al. 2014b). A. Accumulation of carbon in an unharvested forest stand. B. Carbon in the stand regenerating after harvest. C. Harvested biomass is used to produce wood pellets; life cycle GHG emissions from obtaining and producing wood pellets are lower than life cycle and combustion emissions of coal, resulting in a GHG benefit of using wood pellets to replace coal. D. Carbon sequestration parity is achieved when the sum of carbon in the regenerating stand and the GHG benefits of using wood pellets to replace coal reaches the amount of carbon in the stand if it had remained unharvested; carbon debt repayment is achieved when the sum of carbon in the regenerating stand and GHG benefits of using wood pellets to replace coal reaches the preharvest amount of carbon in the stand.

Management and Policy Implications

A growing market for energy produced from forest biomass has arisen because of the potential to mitigate climate change by replacing fossil fuel energy. However, managers who want to access this market should be aware that the benefits of forest bioenergy depend on evaluation of forest management options against a baseline scenario considering what happens to carbon stocks if biomass is not harvested for energy. Among the more favorable options are the use of residue from ongoing harvest operations for traditional wood products (lumber and pulp) and application of intensive silviculture to regeneration of harvested stands. Establishment of new bioenergy-designated plantations on abandoned/degraded lands requires more time for forest biomass to become available for harvest but has the advantage of a low carbon stock value baseline. The least favorable options include harvest of standing live trees, both in addition to and in lieu of ongoing harvest operations for traditional wood products. Policies for bioenergy use also need to recognize that accounting for emission benefits when fossil fuels are replaced requires accounting for forest carbon (either in forest or in traditional wood products) that would have continued to exist if fossil fuels were not replaced by bioenergy.

Forest bioenergy production involves the use of fossil fuels, resulting in GHG emissions that are estimated using life cycle analysis (LCA). An LCA accounts for emissions associated with all phases of bioenergy production and use (the so-called “cradle-to-grave” approach): silvicultural activities, use of logging equipment, transportation of harvested biomass to a biofuel processing facility, conversion of biomass into biofuel, transportation to the energy plant, and non-CO_2 products of combustion (e.g., Zhang et al. 2010). This is the GHG “cost” of producing and using forest bioenergy. The LCA of forest bioenergy does not include CO_2 GHG emissions from biofuel combustion, because these emissions are accounted for when the effects of bioenergy demand on carbon in forest stocks are evaluated (Figure 2C).

When forest bioenergy displaces energy from a fossil fuel, it eliminates GHG emissions from producing and burning the fossil fuel (the *reference fossil fuel scenario*). The LCA for a fossil fuel includes all GHG emissions from obtaining and processing the fuel, but, unlike bioenergy, the fossil fuel LCA also includes all GHG emissions from combustion (Figure 2C). The difference in LCA emissions between forest bioenergy and a fossil fuel constitutes the *GHG benefit* of displacing this fossil fuel with forest bioenergy (Figure 2C and D).

Thus, accounting for the GHG emission reduction potential of forest bioenergy must include the following:

A. Forest carbon following biomass harvest for energy production (the forest bioenergy scenario);
B. Forest carbon in the absence of demand for bioenergy (the forest baseline scenario);
C. Life cycle GHG emissions (upstream fossil fuel emissions) from producing forest bioenergy (excluding GHG combustion emissions); and
D. Life cycle GHG emissions (including those from combustion) for the fossil fuel displaced by forest biomass (the reference fossil fuel scenario).

Components A and B are required to assess CO_2 emissions to the atmosphere or lost potential CO_2 sequestration resulting from extracting biomass from the forest to meet the demand for bioenergy, relative to that without bioenergy demand (i.e., no harvest). Component C (LCA of bioenergy production) includes GHG emissions from producing the biofuel and its use in place of a fossil fuel; it includes non-CO_2 emissions from biomass combustion but not CO_2 emissions, which are accounted for in components A and B. Finally, component D (LCA of the reference fossil fuel) is required to assess the GHG emission benefits of displacing fossil fuel use with forest bioenergy.

Component A should include losses of forest carbon stocks due to the construction of access roads to harvest sites. Similarly, upstream emissions for fossil fuel-based energy (component D) may require accounting for changes in forest carbon stocks if extraction of fossil fuels is associated with forestland cover changes due to mining and road construction. While such losses of forested area in North America may be small at the regional and national scales (e.g., Sleeter et al. 2012, Natural Resources Canada 2013), their local effect on forest carbon stocks can be significant (e.g., Campbell et al. 2012, Drohan et al. 2012).

It should be noted that this review focuses primarily on solid biofuels used for combustion for heat and electricity generation. Although second-generation biofuels (e.g., bioethanol for vehicular use, and biogas) made from wood are currently not commercial energy sources (Naik et al. 2010, Bonin and Lal 2012), early research suggests that wood has a potential to become the main feedstock for production of liquid and gaseous biofuels (Hedegaard et al. 2008, Havlik et al. 2011). However, the principles for assessing the GHG effects of liquid and gaseous biofuels, in particular the methodology to account for changes in forest carbon stocks are the same as those described above.

The difference between components A and B constitutes the change in forest carbon stocks resulting from biomass harvest for bioenergy; the difference between components C and D indicates the GHG benefit of replacing a reference fossil fuel with forest bioenergy (Figure 2C). The estimated total GHG emissions caused by demand for bioenergy to replace fossil fuel are given by

For a detailed mathematical form of Equation 1, see McKechnie et al. (2011). This numerical approach is used for an individual stand. The same approach is used for a forest landscape in which the annual biomass harvest is used to produce energy by integrating Equation 1 over time, starting from the first year of biomass collection.

The LCA of bioenergy and fossil fuel-based energy production usually includes emissions of CO_2 and two other GHGs: methane (CH_4) and nitrous oxide (N_2O) (Intergovernmental Panel on Climate Change [IPCC] 2006). Non-CO_2 GHG emissions are converted into CO_2 equivalents based on their global warming potential (GWP) (IPCC 2007). Despite growing criticism (e.g., Shine 2009, Fuglestvedt et al. 2010), GWP factors remain the standard approach for assessing the effects of GHGs on climate change (IPCC 2007). The amount of CH_4 and N_2O (in units of mass) released during combustion of biofuels and fossil fuels is several orders of magnitude lower than that of CO_2 (IPCC 2006). Release of these GHGs may also result from nitrogen fertilizer application (N_2O emissions) and organic matter decomposition in soil (CH_4 and N_2O emissions) (Cherubini et al. 2009).

Accounting for changes in forest carbon stocks relative to the baseline scenario is paramount for proper assessment of bioenergy GHG emissions: without demand for bioenergy, harvesting either does not occur and the forest continues growing and sequestering additional carbon or it is harvested for traditional wood products (lumber and pulpwood). This is also true when bioenergy replaces fossil fuel energy: replacement of fossil fuels means harvest for bioenergy, whereas no replacement of fossil fuels means no harvest for bioenergy. This link results in an inextricable connection between the reference fossil fuel and forest baseline scenarios: accounting for GHG benefits when fossil fuels are replaced requires accounting for forest carbon losses (either in forest or in traditional wood products) that would *not* have occurred if use of fossil fuels continued.

At the onset of biomass harvesting for bioenergy, the total GHG emissions in Equation 1 are usually negative because the reductions in forest carbon outweigh the GHG benefits of displacing fossil fuel with forest bioenergy. Over time, however, net GHG emissions in the forest bioenergy scenario become smaller as harvested stands regenerate and sequester carbon (Figure 2D). The point at which the change in forest carbon (the difference between forest carbon in the bioenergy and baseline scenarios) equals the accumulated GHG benefit of using forest bioenergy in place of fossil fuel is called *carbon sequestration parity* (Mitchell et al. 2012). Consequently, the time from beginning biomass harvest to carbon sequestration parity is called *time to carbon sequestration parity.* Only after passing the time to carbon sequestration parity does forest bioenergy reduce atmospheric GHG compared with the reference fossil fuel scenario.

Time to carbon sequestration parity is also referred to as the “carbon offset parity point” (e.g., Jonker et al. 2014, p. 371), “break-even period” (e.g., Ter-Mikaelian et al. 2011, p. 644), or “time to carbon neutrality” (Domke et al. 2012, p. 146). We prefer the term carbon sequestration parity rather than carbon neutrality because the latter has been defined in a variety of ways (National Council for Air and Stream Improvement [NCASI] 2013).

Time to carbon sequestration parity depends on factors such as the source of forest biomass (e.g., standing live trees versus harvest residue), growth of regenerating stands after harvest, and emissions from the reference fossil fuel. The peer-reviewed literature contains many studies with estimates of time to carbon sequestration parity for forest bioenergy replacing coal (e.g., McKechnie et al. 2011, Ter-Mikaelian et al. 2011, Holtsmark 2012, Repo et al. 2012, Jonker et al. 2014, Lamers et al. 2014), natural gas (Domke et al. 2012), oil (Repo et al. 2012), and automotive gasoline (Hudiburg et al. 2011). These studies consistently show that harvesting live trees to produce bioenergy initially increases GHG emissions, which may take decades to centuries to offset. However, it has also been shown that intensive forest management of areas harvested for bioenergy may substantially reduce time to carbon sequestration parity (e.g., Jonker et al. 2014, Ter-Mikaelian et al. 2014b).

Figure 3 presents carbon stock changes and GHG emissions for the scenario of annual demand for bioenergy being met by harvesting standing live trees on a landscape scale to displace coal-fired power generation (from McKechnie et al. 2011). The study area covered a total of 52,494 km^2 in the Great Lakes-St. Lawrence forest region (Ontario, Canada); the supply of biomass came from clearcut harvesting of low-intensity managed stands composed of a mix of hardwood (sugar maple, yellow birch, and red oak) and softwood (jack pine, black spruce, and balsam fir) species. Emissions from reduced forest carbon stocks initially outweigh GHG benefits, resulting in positive GHG emissions overall (solid line above zero in Figure 3, indicating increased atmospheric CO_2). The trend is reversed by continued accumulation of GHG benefits from fossil fuel displacement and plateauing of landscape losses in forest carbon, although carbon sequestration parity (and net atmospheric reduction of GHG) is not reached until 38 years after harvesting begins (the time at which the solid line crosses below the zero line), beyond which total GHG emissions are negative (solid line below zero in Figure 3), indicating net removal of CO_2 from the atmosphere.

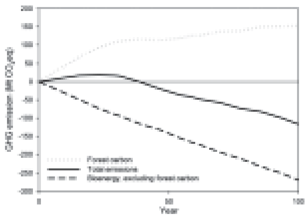

Figure 3.
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Figure 3.

Changes in forest landscape carbon stocks (dotted line), cumulative total GHG emissions (solid line), and GHG benefits (dashed line) from displacing coal with bioenergy generated from harvest of standing live trees (modified from McKechnie et al. 2011). Positive values correspond to emissions, whereas negative values show removals (sequestration) of carbon from the atmosphere.

The approach we describe is based on counting carbon fluxes between the biosphere and atmosphere, referred to as a *mass balance* or *carbon balance* approach (Sathre and Gustavsson 2011). For approaches that enhance the mass balance approach by accounting for the timing of GHG emissions and radiative forcing, the reader is referred to Sathre and Gustavsson (2011), Cherubini et al. (2011), Repo et al. (2012), and Agostini et al. (2013).

REVIEW SCOPE

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Studies accounting for the GHG effects of forest bioenergy are characterized by spatial and temporal boundaries, type of LCA, and forest baseline and reference fossil fuel scenarios (Helin et al. 2013). This review pertains spatially to studies of forest landscapes managed for bioenergy production. We focus primarily on accounting for the carbon effects of harvesting standing live trees for bioenergy, because this biomass source has the greatest potential to produce large, long-lasting effects on the atmospheric carbon concentration. Nevertheless, the same basic premises for determining the atmospheric effects of bioenergy apply to other sources of biomass and are also discussed.

The spatial boundary used in bioenergy GHG accounting is interrelated with the issue of land-use change (LUC), which can be either direct or indirect (Berndes et al. 2010, Bird et al. 2011). Direct LUC involves changes on the land where bioenergy feedstock production occurs, such as a change from farmland to bioenergy plantation. Indirect LUC refers to changes in land use that take place elsewhere as a consequence of harvesting for bioenergy. An example of indirect LUC is conversion in another country of natural forest to farmland in response to the above direct LUC, where farmland in the study area was converted to a bioenergy plantation. Here, we focus on forest landscapes managed for bioenergy production; indirect LUC associated with forest bioenergy production is discussed in a section of this review devoted to that topic.

Bioenergy LCAs can be attributional or consequential (Brander et al. 2008, Lippke et al. 2011, Helin et al. 2013). An attributional LCA provides information about the direct effects of processes used for a given product (e.g., production, consumption, and disposal) but does not consider indirect effects arising from changes in the output of a product (Brander et al. 2008). Studies included in this review use a consequential LCA approach, because they assess the consequences of changes in the level of output of a product, including effects both inside and outside the life cycle of the product (Brander et al. 2008). Some reports (e.g., NCASI 2013) erroneously suggest that the consequential LCA approach is appropriate only for large-scale evaluations of forest carbon policies. In reality, all bioenergy studies reviewed here, regardless of their scale and objective, use a consequential LCA approach, at least partially. Indeed, it is most common to include reference fossil fuel scenarios to demonstrate the GHG benefits of using forest bioenergy. This inclusion automatically places such studies in the category of a consequential LCA approach, because fossil fuel displacement occurs as a consequence of forest bioenergy use.

This review considers three potential forest baseline scenarios: the *no-harvest baseline,* constituting the natural evolution of the forest in the absence of harvest for bioenergy; the *traditional wood products baseline,* in which forest in the absence of harvest for bioenergy is harvested for traditional wood products (lumber and pulpwood); and the *reference point baseline*, which will be introduced later in this review. Of the three baselines, the no-harvest baseline appears to be at the core of many misconceptions discussed in this review. This baseline is also referred to as an “anticipated future baseline” (e.g., AEBIOM 2013, p. 5), a “biomass opportunity cost baseline” (Johnson and Tschudi 2012, p. 12), and a “natural relaxation baseline” (Helin et al. 2013, p. 477). We prefer the term no-harvest baseline because it intuitively suggests what happens to the forest in the absence of harvest for bioenergy. Other baselines considered in the literature, such as the comparative baseline (US Environmental Protection Agency 2011) and the marginal fossil fuel baseline (Johnson and Tschudi 2012), combine forest and reference fossil fuel baselines to estimate net atmospheric balance.

As noted by Helin et al. (2013), there are no scientific criteria governing what the time frame for assessing GHG effects of forest bioenergy must be, because it depends on the aims of the assessment. Typically, studies cover at least one silvicultural rotation, with the time horizon ranging from several decades to hundreds of years. Unlike traditional LCA studies, in which results are presented as one estimate covering the entire time frame, studies on the GHG effects of forest bioenergy often provide a temporal profile of GHG emissions (Helin et al. 2013). It is worth noting, however, that short- and long-term effects of bioenergy emissions are likely to be different (Sedjo 2011). Miner et al. (2014) correctly point out that use of short time frames for assessing the GHG effects of bioenergy is inconsistent with application of GWP factors estimated over a 100-year period (GWP-100). Using a fixed time frame of 100 years is acceptable as long as it is clearly understood that such estimates of GHG effects will be realized 100 years after the beginning of bioenergy production. However, using only a 100-year time frame would obscure time to carbon sequestration parity, which is an important indicator of how long it takes forest bioenergy to start yielding climate mitigation benefits. In addition, the GWP factor for N_2O is reasonably constant over the first 100 years (e.g., GWP-20 and GWP-100 are equal to 289 and 298, respectively) (IPCC 2007). The GWP factor for CH_4 estimated over shorter periods would be higher than that for 100 years (e.g., GWP-20 and GWP-100 are 72 and 25, respectively) (IPCC 2007). However, the numerical error in estimating time to carbon sequestration parity introduced by applying GWP-100 to CH_4 is small because of the relatively low amounts produced during both bioenergy and fossil fuel energy production (e.g., Zhang et al. 2010; also see the sensitivity analysis in Ter-Mikaelian et al. 2014b). Next, we examine common errors in forest bioenergy carbon accounting using live tree harvest for bioenergy, summarized in Table 1.

Table 1.
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Table 1.

Main types of errors in approaches used to assess the carbon effects of forest bioenergy.

COMMON ERRORS IN ACCOUNTING FOR CARBON WHEN USING FOREST BIOENERGY

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Renewable Equals Carbon Neutral

One of the earliest misconceptions about the effects of forest bioenergy is the erroneous conclusion that forest bioenergy is carbon neutral because forests harvested for bioenergy eventually grow back, reabsorbing carbon emitted during energy combustion. Although the flaw in this assumption has been identified repeatedly (e.g., Marland 2010, Agostini et al. 2013), some government documents, forest industry reports, and websites claim that forest bioenergy is carbon neutral because forests regrow. One such statement among many found on the worldwide web is as follows:

> The carbon dioxide (CO_2) emitted on combustion of biomass is taken up by new plant growth, resulting in zero net emissions of CO_2—bioenergy is considered to be carbon neutral (Sustainable Energy Authority of Ireland)[1]

Statements such as this one disregard the time factor for forests to achieve the same forest carbon level relative to the no-bioenergy demand scenario. Although the statement is generally correct in that the forest carbon deficit resulting from biomass harvest for energy might be eventually offset by carbon sequestration in regenerating forests, it is made implicitly incorrect by not acknowledging that decades to centuries are needed to erase this deficit. In the meantime, elevated levels of CO_2 in the atmosphere have numerous potential direct (independent of climate change) and indirect (through changes to climate) biological consequences (Ziska 2008).

Sustained Yield Equals Carbon Neutral

An assumption that bioenergy harvesting in forests managed on a sustained yield (also called sustainable yield) basis does not create a carbon deficit is one of the most common errors in forest bioenergy accounting. This argument is often presented as a “stand versus landscape” approach, implying that the accounting principles presented in the previous section of this review are valid for an individual stand but do not apply to forest landscapes managed for sustained yield. The stand versus landscape approach has been discussed in both the peer-reviewed (e.g., Lamers and Junginger 2013, Jonker et al. 2014) and non-peer-reviewed (e.g., Strauss 2011, 2013, Ray 2012, AEBIOM 2013) literature. The common argument is that because biomass removal from a fraction of the area in a sustained yield landscape is compensated for by growth in the remaining forest, harvesting causes no net loss of biomass, which leads to an incorrect claim that there is no carbon deficit from bioenergy harvest in a sustained yield landscape.

Although sustained yield harvesting is a valid approach in traditional forestry for providing a steady flow of wood, the claim that it is carbon neutral can only be made by ignoring the principles of carbon mass balance accounting (for examples of incorrect accounting, see Strauss and Schmidt 2012, AEBIOM 2013). To repeat these principles, to claim an emissions reduction from using forest biomass to produce energy in place of a fossil fuel, two scenarios must be accepted: one where fossil fuels are used and forests are not harvested for bioenergy; and the other where forests are harvested with the biomass used for energy generation. Stating that sustained yield management is carbon neutral is incorrect because it fails to account for the case involving no harvest for bioenergy in the reference fossil fuel scenario.

Furthermore, in a regulated forest, harvested biomass is maximized on a sustained yield basis when stands are harvested as they reach the maximum mean annual growth rate, which occurs before they attain maximum yield, i.e., if left unharvested the stand would gain more biomass and consequently increase live tree carbon stocks for a period of time (Cooper 1983). A stand may continue to accumulate carbon stocks even past the point of maximum fiber yield, because carbon from dead trees is transferred to dead organic matter pools, which, depending on climate, can have slow decomposition rates (Kurz et al. 2009). Therefore, increased harvest applied to an existing regulated (i.e., sustained yield) forest landscape results in a loss of potential carbon sequestration. This may also be the case in old-growth landscapes (Luyssaert et al. 2008), which may continue to increase total carbon stocks, albeit slowly, in the absence of harvesting. Thus, in a regulated forest landscape, any harvest (and harvest for bioenergy in particular) would in all instances result in increased atmospheric CO_2 for a period of time due to lost future carbon sequestration. Such increases in atmospheric CO_2 cannot be ignored simply because the landscape is being harvested on a sustained yield basis.

In summary, it is an error to conclude that bioenergy from a sustained yield forest is automatically carbon neutral, because, on the one hand, it accepts carbon emissions reductions associated with reduced fossil fuel use, but then fails to acknowledge the “other half” of the reference fossil fuel scenario; i.e., if fossil fuels are used, then forests are not harvested for bioenergy.

Diversion from Traditional Wood Products

An argument can be made that in the sustained yield approach the no-harvest baseline does not need to be considered if, in the absence of demand for bioenergy, forests would be harvested for traditional wood products (e.g., lumber and pulp). This argument may not be relevant, however, because bioenergy is one of the lowest value uses for forest biomass and market forces would be unlikely to result in bioenergy harvest in lieu of harvest for traditional wood products (Werner et al. 2010, AEBIOM 2013). Furthermore, even if the choice was made to harvest for bioenergy, this would shift the harvest for traditional wood products elsewhere (see the section on Indirect LUC), because many studies predict continued growth in demand for traditional wood products both at the national and global scales (Ince et al. 2011, Daigneault et al. 2012, Nepal et al. 2012, Latta et al. 2013).

If these issues were addressed and a legitimate case was made that forest biomass was diverted from harvest for traditional wood products to bioenergy, then the traditional wood products scenario is the correct forest baseline (Agostini et al. 2013). This would include accounting for the large and long-lasting stock of carbon that is retained in some traditional wood products (Chen et al. 2008, 2013). Retention of carbon in wood products is characterized by product “half-life”: the time it takes half of a type of wood product to be removed from service. Estimates of wood product half-life range from 67 to 100 years for construction lumber in the United States and from 1 to 6 years for paper (Skog and Nicholson 2000). After wood product use ends, some carbon may be emitted to the atmosphere through decomposition or burning (with or without producing energy), or wood products may be recycled or disposed of in landfills. In landfills, a fraction of the carbon slowly releases to the atmosphere through decomposition, and the rest remains indefinitely due to its resistance to decomposition (Micales and Skog 1997). The traditional wood products baseline for building materials and other solidwood products should also include the displacement value from using wood compared with using more CO_2 emission-intensive materials (Richter 1998, Gustavsson et al. 2006), so that accounting for wood used for bioenergy in place of use in traditional wood products must include LCA emissions associated with substitution of wood by nonwood materials (Matthews et al. 2012).

Dividend-Then-Debt

Proponents of the dividend-then-debt approach to forest carbon accounting argue that studies on the effects of forest bioenergy are incorrect if they use the moment of harvest as the starting point for carbon cycle analysis (e.g., Strauss 2011, Ray 2012). As stated by Strauss (2013, p. 14),

> all of the studies that show that wood-to-energy adds to the carbon stock of the atmosphere assume a carbon debt is created that has to be repaid by new growth over 30–80 years (or more in some studies)

The dividend-then-debt approach is based on the idea that harvest does not create a loss of forest carbon because it merely returns CO_2 that was previously absorbed by the trees to the atmosphere. To quote, “carbon deficit is only real if you ignore the fact that the trees gobbled up carbon before they were harvested” (Ray 2012).

However, the dividend-then-debt approach ignores the fact that, in most cases, new stands replace previously harvested stands. Those stands were in turn preceded by other stands, and so on. Thus, moving the starting point of carbon accounting backwards in time to when carbon stocks in a given piece of land were low takes credit for the latest cycle of carbon accumulation but ignores the fact that over time, on average, forests contain substantial amounts of carbon. The point in question in dividend-then-debt comes down to the original natural state of the land, which, for most current forestland, was forest. In that case, it is incorrect to use dividend-then-debt accounting.

Plantations Used for Bioenergy Carry No Carbon Debt

Some studies conclude that forest bioenergy obtained from plantations that are already in a sustained yield state carries no carbon debt because the plantations were specifically established to be harvested for bioenergy, and, therefore, all the biomass in such forests can be considered to have been grown for the purpose of burning (e.g., AEBIOM 2013, Jonker et al. 2014). On this basis, it is argued that since carbon in such forests was sequestered for the purpose of burning, without a bioenergy market they would never have existed in the first place. Sedjo (2011) calls this a forward-looking approach:

> if trees are planted in anticipation of their future use for biofuels, then the carbon released on the burning of the wood was previously sequestered in the earlier biological growth process (Sedjo 2011, p. 4)

We contend that this is an acceptable interpretation, but only as long as such plantations were established on deforested land specifically to be harvested for bioenergy. However, we are unaware of large existing areas of plantations in the United States established specifically for bioenergy (short-rotation bioenergy plantations are not uncommon in Europe). For these reasons, the concept is largely hypothetical, and it is a mistake to apply this premise to plantations in general. Furthermore, plantations are usually established on land that historically held natural forest, which either was converted to plantation forest or was deforested and converted to another land use before the plantation forest was established. In such cases, bioenergy plantations would be subject to the criticisms made of the dividend-then-debt approach if they replace plantations for traditional wood products.

In conclusion, existing plantations used for bioenergy cannot be considered exempt from the need to account for carbon using the mass balance approach described in this review, although it may be the case in future for bioenergy plantations established on long-deforested land.

Abandoned Plantations Carry No Carbon Debt

Several studies (e.g., Lamers and Junginger 2013, Jonker et al. 2014) discuss plantations established for traditional wood products but “abandoned” due to diminishing fiber demand (referring primarily to the southeastern United States). They suggest that protection (no harvest) of such plantations is an unlikely scenario, and more realistic alternatives are conversion to agriculture or urban development. Lamers and Junginger (2013) argue that these plantations should therefore be considered a “free” source of bioenergy, since deforestation would be the baseline in the fossil fuel scenario, whereas Jonker et al. (2014) propose using the carbon debt repayment approach discussed later in this review. Here we note that such an approach is in error because it ignores the fate of forest carbon in the baseline scenario where there is no harvest for bioenergy.

Although production of certain traditional wood products (e.g., pulp and paper) has indeed been declining since 2000 (Hujala et al. 2013), the likelihood of there being large numbers of abandoned plantations contradicts national and global projections of increasing demand for traditional wood products (Ince et al. 2011, Daigneault et al. 2012, Nepal et al. 2012, Latta et al. 2013). If, however, there are plantations abandoned due to regional deviations from global trends for which the no-harvest baseline is an unrealistic scenario, then for such plantations the appropriate baseline for forest bioenergy scenario is deforestation followed by LUC. Because it is highly unlikely that the act of deforestation results in disposal of standing live trees as waste, the deforestation baseline should include a single harvest of standing live trees and their utilization for either traditional wood products or bioenergy, with carbon stocks in deforested areas determined by the new land use.

To conclude, the correct baseline scenarios for abandoned plantations are either the no-harvest scenario or, where this is deemed unrealistic, a deforestation scenario that accounts for the fate of forest biomass carbon due to deforestation and carbon stocks in deforested land.

Use of the Carbon Debt Repayment Approach to Carbon Accounting

The concept of carbon debt repayment (Mitchell et al. 2012, Jonker et al. 2014) calls for calculation of the forest carbon deficit relative to the amount of forest carbon at time of harvest. Unlike carbon sequestration parity, carbon debt repayment, referred to as "atmospheric carbon parity" by Agostini et al. (2013, p. 33), assumes that a forest carbon deficit created by harvest is completely repaid once the combined balance of carbon stocks in the postharvest forest and LCA benefits from substituting for fossil fuel equals carbon stocks in the preharvest forest (Figure 2D).

Recent defense of the carbon debt repayment approach was made in a report published by AEBIOM (2013). In its discussion of harvest for bioenergy of standing live trees in southeastern US forests, the no-harvest baseline is called "completely inappropriate" and "unrealistic" and is listed among the

> fundamental flaws in key assumptions and methodology that underlie prominent studies that have found forest-based bioenergy to be associated with significant carbon deficits (AEBIOM 2013, p. 5–6)

Instead, the report advocates using the so-called "reference point baseline" (p. 36), which is identical to carbon debt repayment.

Proponents of carbon debt repayment (such as AEBIOM 2013, Jonker et al. 2014) make the fundamental error of ignoring the fate of forests in the reference fossil fuel scenario. As noted earlier, in the fossil fuel scenario, when GHG emissions from fossil fuel combustion occur, they do so in lieu of bioenergy, and so carbon stored in forests increases over time. To claim emissions reductions from avoided fossil fuel use, it is logically required that forest growth be accounted for in the case where fossil fuels are used (no harvest for bioenergy is needed). Therefore, use of the carbon debt repayment method results in incorrect estimates of bioenergy GHG emissions.

INDIRECT LUC

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As noted earlier, indirect LUC refers to changes in land use outside the area managed for bioenergy that occur as a consequence of harvesting for bioenergy (Berndes et al. 2010, Bird et al. 2011). For this reason, the spatial scale of bioenergy studies where indirect LUC is considered typically are regional or national in scope (for examples, see Abt et al. 2010, 2012, Ince et al. 2011, Galik and Abt 2012, Daigneault et al. 2012, Nepal et al. 2012, Sedjo and Tian 2012, Latta et al. 2013).

The above cited studies share in common the use of econometric models to analyze the effects of market prices and wood products and bioenergy demand scenarios on forest growing stock and/or carbon. Carbon accounting in these studies often has serious shortcomings; for example, some do not account for LCA emissions, whereas others do not consider forest carbon pools beyond those in harvested wood. Such shortcomings can potentially alter whether or when forest biomass produces a net atmospheric carbon benefit. Generally, and with these caveats in mind, such studies conclude that greater bioenergy demand would increase biomass supply and that growth in forest carbon due to indirect land use effects, such as increased planting or silviculture, may outpace forest carbon stock reductions caused by bioenergy harvest.

In the event that indirect LUC is accounted for, the estimation of GHG emissions attributed to forest bioenergy still requires quantification of forest carbon stocks in an appropriate forest baseline, as well as LCA emissions for the bioenergy and reference fossil fuel scenarios. This is because direct LUC associated with forest bioenergy (forest landscape managed for bioenergy) is “nested” in indirect LUC (changes to forest and/or nonforested areas outside of the landscape managed for forest bioenergy) (Berndes et al. 2010). In other words, inclusion of indirect LUC may alter the time to carbon sequestration parity for a given forestry system, but it does not alter the methodology of assessing the forest bioenergy contribution to GHG emissions from this system. In addition, it is important to verify that potential indirect LUC does in practice occur, taking note of Rabl et al. (2007), who recommend that emissions and removals of CO_2 be accounted for explicitly during each stage of the bioenergy life cycle. We consider the recommendation by Rabl et al. (2007) key, given some highly uncertain potential consequences to indirect LUC resulting from increased bioenergy demand.

OTHER SOURCES OF FOREST BIOMASS

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Residue from ongoing harvest operations is the second most common potential source of biomass considered in the literature on forest bioenergy. The GHG effects of using harvest residue for bioenergy have been studied by several authors (e.g., McKechnie et al. 2011, Domke et al. 2012, Repo et al. 2012). The key difference between assessing GHG effects of using harvest residue versus live trees as a source of biomass is in the baseline scenario: in the case of harvest residue, the baseline scenario must include a projection of the amount of carbon stored in harvest residue if it were not collected because of an absence of demand for bioenergy (an exception to the need to account for the fate of harvest residue is if it came from plantations established specifically for bioenergy production). Consequently, studies not including an analysis of a residue baseline scenario are bound to show shorter periods to reach a net reduction in GHG emissions (e.g., Yoshioka et al. 2005, Froese et al. 2010, Gustavsson et al. 2011).

Studies accounting for the fate of residues in the event they are not used for bioenergy are consistent in concluding that an overall reduction in GHG emissions is achieved within the first few years of biomass collection. Based on literature reports reviewed by Lamers and Junginger (2013), the time required to achieve the reduction in total GHG emissions ranges from 0 to 16 years from the onset of harvest residue collection for bioenergy. A variation in the time to overall GHG emission reduction is caused by assumptions about the fate of residue in the baseline scenario (e.g., decomposition rate and rate of slash burning) and the reference fossil fuel.

The assumption that harvest residue is a carbon “free” source of biomass for energy because otherwise it would be burned is an exaggeration of its fate (for example, in AEBIOM 2013, p. 18: “the majority of the biomass left following harvest is burned as a waste management measure”). The reality of the residue baseline scenario is more complex. First, in some regions, all harvest residue is left on site to decompose; i.e., none is burned (e.g., McKechnie et al. 2011). Decomposition varies by region, but it is not instantaneous. Second, even where harvest residue is burned, a substantial fraction does not get burned for logistical reasons (e.g., insufficient staffing and weather conditions). Analysis of annual forest management reports by Ter-Mikaelian et al. (2014b) revealed that fewer than 50% of slash piles were burned in northwestern Ontario, Canada. Differences in slash burning rates are also apparent among the administrative regions of British Columbia, Canada (Lamers et al. 2014). Even in the case of slash burning, the net effect of collecting it for bioenergy is not zero, contrary to the suggestion by Miner et al. (2014), because of incomplete combustion, with between 5 and 25% of residue in piles remaining after burning (e.g., Hardy 1996). Incomplete combustion of slash when burned produces black carbon, which resists biological and chemical degradation (Forbes et al. 2006). Although the black carbon pool is relatively small, its stability makes it an important component of total forest carbon. Thus, the baseline for harvest residue is not straightforward and should reflect local conditions and practices.

Sawmill residue (sawdust and wood chips), because it is a by-product of traditional wood products, has a substantially lower GHG baseline scenario compared with that of other sources of biomass because its LCA emissions include only those from production and transportation of biofuel and non-CO_2 GHGs from its combustion. However, according to Gronowska et al. (2009), in the United States about 98 and 60% of primary and secondary mill residue, respectively, is already used for energy or other value-added products; in Canada, 70% of mill residue is currently used. Properly assessing the GHG effects of mill residue used for bioenergy thus requires knowledge of the existing fate of mill residue to correctly define its baseline scenario in the absence of use for bioenergy.

IS FOREST BIOENERGY “BAD” FOR CLIMATE?

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The aim of this review is to promote accurate accounting of the atmospheric effects of bioenergy, not to argue against using forest biomass for energy generation. When correctly accounted for, GHG emissions from live tree forest biomass used for energy exceed those from fossil fuels for periods of a few years to more than a century, and the difference can be substantial, depending on the characteristics of the forest harvested and the fossil fuel replaced by bioenergy. Even when bioenergy from live tree biomass from temperate forests replaces coal, a CO_2-intensive fossil fuel, the time to obtain a net reduction in atmospheric CO_2 can be decades; if it is replacing a less CO_2-intensive fossil fuel, the time to achieve an atmospheric benefit may be more than 100 years.

Nevertheless, as correctly pointed out by AEBIOM (2013) and NCASI (2013), biomass combustion for bioenergy emits carbon that is part of the biogenic carbon cycle. Despite delays that may occur in achieving a net reduction in atmospheric carbon, as long as forests regrow, the total amount of carbon in the biosphere-atmosphere system remains approximately the same, with small increases due to consumption of fossil fuels to obtain, process, and transport the biofuel. It is considerably more damaging when energy is generated from fossil fuels because this increases total carbon in the biosphere-atmosphere system and is essentially permanent. We also note that the long-term GHG benefits of substituting fossil fuels with forest bioenergy will greatly surpass those of carbon sequestration in forests (e.g., see Miner et al. 2014) because net carbon accumulation in the no-harvest baseline scenario will slow substantially as forests reach maturity, whereas the benefits of substituting fossil fuels with forest bioenergy will keep accumulating at a steady pace. In addition, forest bioenergy may be needed as a stopgap until sufficient nonfossil fuel energy generation methods, with better atmospheric CO_2 consequences than forest biomass, can be implemented. Until then, even a century-long increase in atmospheric CO_2 caused by using forest bioenergy may be preferable to burning fossil fuels. As stated by Dehue (2013), mitigation of climate change may not be possible without broad-scale use of forest bioenergy; in other words, human society is probably going to require use of all available options to mitigate climate change, whether such options provide a short- or long-term GHG reduction benefit.

There may be reasons beyond climate change to harvest forests to produce bioenergy, such as the opportunity for forest landowners to receive economic benefits (as mentioned in AEBIOM 2013), the economic benefits to society overall of reducing dependence on imported fossil fuels (US Department of Energy 2013), or achievement of ecological objectives for which forest disturbance is necessary (Colombo et al. 2012). However, the rationale for using forest bioenergy should avoid the false promises of instant benefits to climate change mitigation. In this regard, we note that the principles of carbon accounting discussed in this review should not be confused with those described by the United Nations Framework Convention on Climate Change, the latter reflecting international carbon accounting entailing political compromises needed to reach agreement among participating parties (Prag et al. 2013).

In conclusion, some biomass sources used for forest bioenergy may indeed provide near-immediate GHG reduction, whereas others produce decades- to century-long increases in atmospheric GHGs. Our goal in this review was to support what we consider the use of scientifically sound knowledge for informed decisionmaking about using forest bioenergy for climate change mitigation and to help remove confusion caused by flawed approaches to bioenergy carbon accounting.

NOTES

ENDNOTE

[1.] For more information, see www.seai.ie/Renewables/Bioenergy/Introduction_to_Bioenergy/.

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UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON D.C. 20460

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September 28, 2012

EPA-SAB-12-011

The Honorable Lisa P. Jackson
Administrator
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, N.W.
Washington, D.C. 20460

Subject: SAB Review of EPA's Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources (September 2011)

Dear Administrator Jackson:

EPA's Science Advisory Board (SAB) was asked by the EPA's Office of Air and Radiation to review and comment on the EPA's *Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources (Framework, September 2011)*. The *Framework* considers the scientific and technical issues associated with accounting for emissions of biogenic carbon dioxide (CO_2) from stationary sources and develops a method to adjust the stack emissions from stationary sources using biological material based on the induced changes in carbon stocks on land (in soils, plants and forests).

Assessing the greenhouse gas implications of using biomass to produce energy is a daunting task and the EPA is to be commended for its effort. The context for the *Framework* arose when the EPA established thresholds for greenhouse gas emissions from stationary sources for the purposes of Clean Air Act permits under the New Source Review (Prevention of Significant Deterioration program) and Title V operations program. The agency needed to consider how to include biogenic emissions in determining whether thresholds for regulation have been met. In July 2011, the EPA deferred the application of permitting requirements to biogenic carbon dioxide emissions from bioenergy and other biogenic stationary sources for three years, while conducting a detailed examination of the issues associated with biogenic CO_2.

The agency sought a method of "adjusting" biogenic carbon emissions from stationary sources to credit those emissions with carbon uptake during sequestration or, alternatively, avoided emissions from natural decay (e.g., from residues and waste materials). Without a way of adjusting those emissions, the agency's options would be either a categorical inclusion (treating biogenic feedstocks as equivalent to fossil fuels) or a categorical exclusion (excluding biogenic emissions from determining applicability thresholds for regulation). The purpose of the *Framework* was to propose a method for calculating the adjustment, or a Biogenic Accounting Factor (BAF) for biogenic feedstocks, based on their interaction with the carbon cycle. The BAF is an accounting term developed in the *Framework* to denote the offset to total emissions (mathematical adjustment) needed to reflect a biogenic feedstocks' <u>net</u> greenhouse gas emissions after taking into account its offsite sequestration, in biomass or land, or avoided emissions. Avoided emissions are emissions that would occur anyway without removal of the feedstock for bioenergy.

The SAB was asked to comment on the science and technical issues relevant to accounting for biogenic CO_2 emissions. We found the issues are different for each feedstock category and sometimes differ within a category. Forest-derived woody biomass stands out uniquely for its much longer rotation period than agricultural (short-rotation) feedstocks. The *Framework* includes most of the elements that would be needed to gauge changes in CO_2 emissions; however, the reference year approach employed does not provide an estimate of the additional emissions and the sequestration changes in response to biomass feedstock demand. 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 question at hand. To do so requires an anticipated baseline approach. Because forest-derived woody biomass is a long-rotation feedstock, 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. Although this would not be an easy task, it would be necessary to estimate carbon cycle changes associated with the biogenic feedstock. In addition, an anticipated baseline would be needed to estimate additional changes in soil carbon stock over time. 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. In the attached report, the SAB provides some suggestions for an "anticipated baseline" approach while acknowledging the uncertainty and difficulty associated with modeling future scenarios.

For agricultural feedstocks, the variables in the *Framework* capture most of the factors necessary for estimating the carbon change associated with the feedstock use. For short rotation agricultural feedstocks where carbon accumulation occurs within one to a few years, the *Framework* can, with some adjustments to address estimation problems (including an anticipated baseline for soil carbon changes) and careful consideration of data and implementation, represent direct carbon changes in a particular region. As recognized by the agency, for many waste feedstocks (municipal solid waste, construction and demolition waste, industrial wastes, manure, tire-derived wastes and wastewater), combustion to produce energy releases CO_2 that would have otherwise been returned to the atmosphere from the natural decay of waste. The agency chose not to model natural decomposition in the *Framework*; however, modeling the decay of agricultural and forest residues based on their alternate fate (e.g., whether the materials would have been disposed in a controlled or uncontrolled landfill or left on site, or subject to open burning) could be incorporated to improve scientific accuracy.

The *Framework* does not discuss the different time scales inherent in the carbon cycle nor does it characterize potential intertemporal tradeoffs associated with the use of biogenic feedstocks. However the SAB recommends that intertemporal tradeoffs be made transparent in the *Framework* for policymakers. For forest-derived roundwood, carbon debts and credits can be created in the short run with increased harvesting and planting respectively but in the long run, net climate benefits can accrue with net forest growth. While it is clear that the agency can only regulate emissions, its policy choices about regulating emissions will be better informed with consideration of the temporal distribution of biogenic emissions and associated carbon sequestration or avoided emissions.

The SAB was asked whether we supported EPA's distinction between policy and technical considerations. We do not. In fact, the lack of information in the *Framework* on EPA's policy context and the menu of options made it more difficult to fully evaluate the *Framework*. Because the reasonableness of any accounting system depends on the regulatory context to which it is applied, the *Framework* should describe the Clean Air Act motivation for this proposed accounting system, including how the agency regulates point sources for greenhouse gases and other pollutants. This SAB review would have been enhanced if the agency had made explicit all Clean Air Act policy options for regulating greenhouses gases, including any potential implementation of carbon offsets or certification of sustainable forestry practices, as well as its legal boundaries regarding upstream and downstream emissions.

Overall, the SAB found that quantification of most components of the *Framework* has uncertainties, technical difficulties, data deficiencies and implementation challenges. These issues received little attention in the *Framework*, but are important considerations relevant to scientific integrity and operational efficiency. Moreover, the agency should consider consistency between biogenic carbon accounting and fossil fuel emissions accounting. Ideally both fossil fuels and biogenic feedstocks should be subject to the same emissions accounting. While there are no easy answers to accounting for the greenhouse gas implications of bioenergy, further consideration of the issues raised by the SAB and revisions to the *Framework* could result in more scientific rigor in accounting for biogenic emissions. One SAB Panel member expressed a dissenting opinion and recommended that the agency abandon the *Framework* altogether and instead choose to exempt biogenic CO_2 emissions from greenhouse gas regulation so long as aggregate measures of land-based carbon stocks are steady or increasing. This dissenting opinion is based on an accounting guideline from the Intergovernmental Panel on Climate Change (IPCC) which recommends that emissions from bioenergy be accounted for in the forestry sector. This is not the general consensus view of the SAB. The IPCC approach to carbon accounting would not allow for a causal connection to be made between a stationary facility using a biogenic feedstock and the source of that feedstock, and thus cannot be used for permit granting purposes. Also, the IPCC approach would not capture the marginal effect of increased biomass harvesting for bioenergy on atmospheric carbon levels.

The SAB found a number of important limitations in the *Framework*, including the lack of definition of several key features, such that the *Framework's* implementation remains ambiguous. Also, the *Framework* does not incorporate the three feedstock groupings into the details of the methodology or the case studies, thus limiting useful evaluation. The *Framework* also does not discuss the likely event of unintended consequences.

The SAB was not asked to recommend alternatives to the *Framework* but given the challenges associated with improving and implementing the *Framework*, the SAB recommends that EPA consider developing default BAFs by feedstock category and region. Under EPA's current *Framework*, facility-specific BAFs would be calculated to reflect the incremental carbon cycle and net emissions effects of a facility's use of a biogenic feedstock. Rather than trying to calculate a BAF at the facility-level, a default BAF could be calculated for each feedstock category, and might vary by region, prior land use and current land management practices. The defaults would also have administrative advantages in that they would be easier to implement and update. Facilities could also be given the option of demonstrating a lower BAF for their feedstocks.

The SAB acknowledges that practical considerations will weigh heavily in the agency's decision making. In fact, any method that might be adopted or considered, including methods proposed by the SAB, should be subject to an evaluation of the costs of compliance and the carbon emissions savings likely to be achieved as compared to both a categorical inclusion and a categorical exclusion. Uncertainties in the assessment of both the costs and the emissions savings should be analyzed and used to inform the choice of policy.

The SAB appreciates the opportunity to provide advice on the *Framework* and looks forward to your response.

Sincerely,

/Signed/

Deborah L. Swackhamer, Ph.D.
Chair
Science Advisory Board

/Signed/

Madhu Khanna, Ph.D.
Chair
Biogenic Carbon Emissions Panel

Enclosure

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Science Advisory Board
Biogenic Carbon Emissions Panel

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[*] Dr. Sedjo provided a dissenting opinion (See Appendix E.)

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Acronyms and Abbreviations

AVOIDEMIT	Avoided Emissions
BAF	Biogenic Accounting Factor
BAU	Business as Usual
CH_4	Methane
CO_2	Carbon Dioxide
CO_2e	Carbon Dioxide Equivalent
DOE	Department of Energy
EPA	Environmental Protection Agency
FASOM	Forestry and Agricultural Sector Optimization Model
GHG	Greenhouse gases
GROW	Growth
GtC/y	Gigatons of carbon per year
GTMM	Global Timber Market Model
GTP	Global Temperature Potential
GWP	Global Warming Potential
GWPbio	Global Warming Potential of biomass
I	Carbon Input
K	Proportion of Carbon Lost per unit of time
LAR	Level of Atmospheric Reduction
LEAK	Leakage
N_2O	Nitrous Oxide
NSR	New Source Review
PRODC	Carbon in Products
PSD	Prevention of Significant Deterioration
RPA	Resources Planning Act
SAB	Science Advisory Board
SEQP	Sequestered Fraction
SITE_TNC	Total Net Change in Site Emissions
SRTS	Sub-regional Timber Supply Model
USDA	United States Department of Agriculture

1. EXECUTIVE SUMMARY

Biogenic CO_2 emissions from bioenergy are generated during the combustion or decomposition of biologically-based material. Biogenic feedstocks differ from fossil fuels in that they may be replenished in a continuous cycle of planting, harvesting and regrowth. The same plants that provide combustable feedstocks for electricity generation also sequester carbon from the atmosphere. Plants convert raw materials present in the ecosystem such as carbon from the atmosphere and inorganic minerals and compounds from the soil (including nitrogen, potassium, and iron) and make these elemental nutrients available to other life forms. Carbon is returned to the atmosphere by plants and animals through decomposition and respiration and by industrial processes, including combustion. Biogenic CO_2 is emitted from stationary sources through a variety of energy-related and industrial processes. Thus, the use of biogenic feedstocks results in both carbon emissions and carbon sequestration.

EPA's *Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources (Framework, September 2011)* explores the scientific and technical issues associated with accounting for emissions of biogenic carbon dioxide (CO_2) from stationary sources and develops a method to adjust the stack emissions from bioenergy based on the induced changes in carbon stocks on land (in soils, plants and forests). The context for the *Framework* is the treatment of biogenic CO_2 emissions in stationary source regulation given the unique feature of plant biomass in providing uptake of carbon dioxide (CO_2) from the atmosphere during the photosynthesis. Under the Clean Air Act, major new sources of certain air pollutants, defined as "regulated New Source Review (NSR) pollutants" and major modifications to existing major sources are required to obtain a permit. The set of conditions that determine which sources and modifications are subject to the agency's permitting requirements are referred to as "applicability" requirements. Since greenhouse gases are included in the definition of a "regulated NSR pollutant," EPA has to make a determination about whether a source meets the "applicability threshold" to trigger permitting requirements. As of January 2011, for facilities already covered by the Prevention of Significant Deterioration (PSD) or Clean Air Act Title V programs, greenhouse gas emission increases of 75,000 tons per year (tpy) or more, on a carbon dioxide equivalent (CO_2e) basis, would be subject to technology requirements under the PSD program. As of July 1, 2011, more facilities became subject to regulation based on their greenhouse gas emissions. Specifically new and existing stationary sources (that are not already covered by the PSD or Title V programs) that emit greenhouse gas emissions of at least 100,000 tpy became subject to greenhouse gas regulation even if they do not exceed the permitting thresholds for any other pollutant. The question before the agency, and hence, the motivation for the *Framework*, is whether and how to consider biogenic greenhouse gas emissions in determining these thresholds for permitting. The SAB's consensus advice is highlighted in this Executive Summary with more details in the attached report. A dissenting opinion is found in Attachment E.

Evaluation of the Underlying Science

The SAB was asked to comment on the Framework's assessment and characterization of the underlying science and the implications for biogenic CO_2 accounting. EPA has accurately captured the global carbon cycle's flows and pools of carbon. The *Framework* does an admirable job describing the task of quantifying the impact of transforming biologically based carbon from a terrestrial storage pool (such as aboveground biomass) into CO_2 via combustion, decomposition or processing at a stationary source. At the same time, there are several important scientific issues that are not addressed in the *Framework*.

Time scale

The *Framework* seeks to determine annual changes in emissions and sequestration rather than assessing the manner in which these changes will impact the climate over longer periods of time. In so doing, it does not consider the different ways in which use of bioenergy impacts the carbon cycle and global temperature over different time scales. Nor does it consider temporal differences of climate effects on the environment. Some recent studies have shown that there could be intertemporal tradeoffs with the use of long rotation feedstocks that should be highlighted for policymakers. In the short/medium run, at the forest stand level, there can be a lag time between emissions (through combustion) and sequestration (through regrowth) with the use of forest biomass. At the landscape level, there can be concurrent debts and credits with harvesting and planting. The impacts of the temporal pattern on climate response depend on the metric used for measuring climate impacts and the time horizon being considered. Some modeling exercises have shown that the probability of limiting warming to or below 2°C in the twenty-first century is dependent upon cumulative emissions by 2050 (Meinshausen et al. 2009). This suggests that an early phase of elevated emissions from forest biomass could reduce the odds of limiting climate warming to 2°C in the near term. On the other hand, the use of forest biomass to displace fossil energy with forest regrowth rates that match harvest rates could leave cumulative emissions unchanged over a 100 year horizon and thereby have minimal effect on peak warming rates 100 years later as compared to the use of fossil energy (Allen et al. 2009; NRC 2011; Cherubini et al. 2012). If the climate effect of biogenic feedstocks is explored, the degree to which biogenic feedstocks curtail fossil fuel use should be assessed and quantified. In addition, the net accumulation of forest and soil carbon over a 100 year period should not be assumed to occur automatically or be permanent; rather growth and accumulation should be monitored and evaluated for changes resulting from management, market forces or natural causes.

An accounting framework that incorporates consideration of time will result in a Biogenic Accounting Factor (BAF) estimate that depends on the time horizon chosen for measuring the climate impact and recognition of the benefits from displacing fossil fuels. Given the slow response of the carbon and climate system, if biogenic feedstocks displace the use of fossil fuels for longer than 100 years, then there may be a beneficial climate effect. In contrast, if the use of biogenic feedstocks does not displace fossil fuels, then any presumed beneficial climate consequences of biogenic carbon may be overestimated.

Spatial Scale

The use of unspecified “regions” as fuelsheds in combination with a reference year baseline is a central weakness of the *Framework* with respect to forest-derived feedstocks. The EPA used a variable for the Level of Atmospheric Reduction (LAR) to capture the proportion of potential gross emissions that are offset by sequestration during feedstock growth, however the calculation of LAR captures landscape wide changes rather than facility-specific carbon emissions associated with actual fuelsheds. As a result, the estimates of the BAFs are sensitive to the choice of the spatial region as shown in the agency’s own case study.

Intergovernmental Panel on Climate Change Approach

The SAB was asked whether we agreed with the EPA's concerns about applying the Intergovernmental Panel on Climate Change (IPCC) approach to biogenic CO_2 emissions at individual stationary sources. The IPCC provides guidelines for countries to estimate and report all of their anthropogenic greenhouse gas emissions to the United Nations in a consistent manner. In these guidelines, biogenic CO_2 emissions were assigned to the land areas where carbon is stored, regardless of where the emissions actually take place. The application of the IPCC approach would lead to the outcome that biogenic CO_2 emissions at stationary facilities are considered part of the land-based accounts assigned to landowners and, hence, stationary source facilities would not be held responsible. The SAB agrees with the agency that this approach would not be appropriate because it does not allow a link between the stationary source that is using biomass feedstocks and the emissions that are being measured. This link is critical in order to be able to regulate emissions at a stationary source level which is the way that greenhouse gas emissions are mandated to be regulated under the Clean Air Act. To adjust the stack emissions from stationary facility bioenergy based on the induced changes off-site in carbon stocks on land, a chain of custody has to be established with the source of the feedstock. Furthermore, while the IPCC approach can be used to determine if stock of carbon is increasing or decreasing over time, it cannot be used to determine the net impact of using a biogenic feedstock on carbon emissions as compared to what the emissions would have been if the feedstock had not been used. In order to adjust the emissions of a stationary facility using biogenic material it is important to know the net impact of that facility on carbon emissions – which requires knowing what the emissions would have been without the use of bioenergy and comparing it with emissions with the use of bioenergy. If EPA were to apply the IPCC approach, as long as carbon stocks are increasing, bioenergy would be considered carbon neutral. Under this approach, forest carbon stocks may be increasing less with the use of bioenergy than without but forest biomass would still be considered carbon neutral. Application of the IPCC accounting approach is not conducive to considering the incremental effect of bioenergy on carbon emissions.

Categorical Inclusion or Exclusion

The SAB was asked whether we agreed with EPA's conclusion that the categorical approaches (inclusion and exclusion) are inappropriate for regulatory purposes based on the characteristics of the carbon cycle. A categorical inclusion would treat all biogenic carbon emissions at the combustion source as equivalent to fossil fuel emissions, while a categorical exclusion would exempt biogenic carbon emissions from greenhouse gas regulation. The agency rejected both extremes and asked the SAB whether it supported their conclusion that a priori categorical approaches are inappropriate for the treatment of biogenic carbon emissions.

The decision about a categorical inclusion or exclusion will likely involve many considerations that fall outside the SAB's scientific purview such as legality, feasibility and, possibly, political will. The SAB cannot speak to the legal or regulatory complexities that could accompany any policy on biogenic carbon emissions but this Advisory offers some scientific observations that may inform the Administrator's policy decision.

Carbon neutrality cannot be assumed for all biomass energy a priori. There are circumstances in which biomass is grown, harvested and combusted in a carbon neutral fashion but carbon neutrality is not an appropriate a priori assumption; it is a conclusion that should be reached only after considering a particular feedstock's production and consumption cycle. There is considerable heterogeneity in feedstock types, sources and production methods and thus net biogenic carbon emissions will vary considerably. Of course, biogenic feedstocks that displace fossil fuels do not have to be carbon neutral to be better than fossil fuels in terms of their climate impact.

Given that some biomass could have positive net emissions, a categorical exclusion would exempt the stationary source from the responsibility of controlling CO_2 emissions from its use of biogenic material and provide no incentive for the development and use of best management practices. Conversely, a categorical inclusion would provide no incentive for using biogenic sources that compare favorably to fossil energy in terms of greenhouse gas emissions.

A dissenting opinion in Attachment E offers support for applying the IPCC approach, discussed above, to regulatory decisions about biogenic feedstocks. Such an approach would not be consistent with EPA's responsibility under the Clean Air Act, nor would it capture the marginal effect of increased biomass harvesting on forest carbon stocks and atmospheric carbon levels. Specifically, EPA is not charged with regulating regional or national forest carbon stocks: it must regulate stationary facilities. The dissenting opinion expressed a preference for exempting bioenergy from greenhouse gas regulation so long as land carbon stocks are rising. However, the general consensus view of the SAB is that the IPCC inventories, a static snapshot of emissions at any given point in time, are a reporting convention that lacks connection to any associated policies or implementation. Merely knowing whether carbon sequestration at the landscape level has increased or decreased tells us nothing about the incremental effect that bioenergy production has on carbon emissions. The IPCC inventories do not explicitly link biogenic CO_2 emission sources and sinks to stationary sources, nor do they provide a mechanism for measuring changes in emissions as a result of changes in the building and operation of stationary sources using biomass.

Issues with Biogenic Accounting Factor (BAF) Calculation

The *Framework* presents an alternative to a categorical inclusion or exclusion by offering an equation for calculating a Biogenic Accounting Factor (BAF) that would be used to adjust the onsite biogenic emissions at the stationary source emitting biogenic CO_2 on the basis of information about growth of the feedstock and/or avoidance of biogenic emissions and more generally the carbon cycle. Note that in the comments below, the SAB's advice on the *Framework* (i.e., the application of the BAF equation to biogenic feedstocks) differs by feedstock category. In particular, the SAB is more critical of the *Framework*'s treatment of biomass from roundwood trees than from agricultural and waste feedstocks.

Agricultural and Waste Feedstocks

For faster growing biomass like agricultural crops, the anticipated future baseline approach is still necessary to reflect changes in dynamic processes, e.g., soil carbon, "anyway" emissions (those that would occur anyway without removal or diversion of nongrowing feedstocks, for example, corn stover) , and landscape changes. For agricultural feedstocks in general, the *Framework* captures many of the factors necessary for estimating the offsite carbon change associated with use of short rotation (agricultural) feedstocks. These include factors to represent the carbon embodied in products leaving a stationary source, the proportion of feedstock lost in conveyance, the offset represented by sequestration, the site-level difference in net carbon flux as a result of harvesting, "anyway" emissions and other variables. In addition to the anticipated baseline, a noticeable omission is the absence of consideration of nitrous oxide (N_2O) emissions from fertilizer use, potentially a major onsite greenhouse gas loss that could be induced by a growing bioenergy market.

For short rotation feedstocks where carbon accumulation and "anyway" emissions are within one to a few years (i.e., agricultural residues, perennial herbaceous crops, mill wood wastes, other wastes), the *Framework* may, with some adjustments to address estimation problems (including an anticipated baseline for soil carbon changes, residue disposition and land management) and careful consideration of data and implementation, accurately represent direct carbon changes in a particular region. For logging residues and other feedstocks that decay over longer periods, decomposition cannot be assumed to be instantaneous and the *Framework* could be modified to incorporate the time path of decay of these residues if they are not used for bioenergy. This time path should consider the alternative fate of these residues, which in some cases may involve removal and burning to reduce risks of fire or maintain forest health.

For waste materials (municipal solid waste), the *Framework* should consider the alternate disposition of waste material (what would happen if not used as feedstock) in an anticipated baseline (counterfactual) framework. This anticipated baseline should include emissions and partial capture of methane (CH_4) emissions from landfills. In general, when accounting for emissions from wood mill waste and pulping liquor, the EPA should recognize these emissions are part of a larger system that includes forests, solid wood mills, pulp mills and stationary energy sources. Accounting for greenhouse gases in the larger system should track all emissions or forest stock changes over time across the outputs from the system so as to account for all fluxes. Within the larger system, the allocation of fluxes to wood/paper products or to a stationary source is a policy decision. The agency should consider how its *Framework* meets the scientific requirement to account (allocate) all emissions across the larger system of forests, mills and stationary sources over time.

Forest-Derived Woody Biomass

The EPA's stated objective was to accurately reflect the carbon outcome of biomass use by stationary sources. For forest-derived woody biomass, the *Framework* did not achieve this objective. To calculate BAF for biomass from roundwood trees, the agency proposed the concept of regional carbon stocks (with the regions unspecified) and posed a "rule" whereby any bioenergy usage that takes place in a region where carbon stocks are increasing would be assigned a BAF of 0 (and hence carbon emissions would not be subject to greenhouse gas regulation). This decouples the BAF from a particular facility's biogenic emissions and the sequestration (offset) associated with its particular feedstock. Emissions from a stationary facility would be included or excluded from greenhouse gas regulation depending on a host of factors in the region far beyond the facility's control.

To accurately capture the carbon outcome, an anticipated baseline approach and landscape level perspective are needed. An anticipated baseline requires selecting a time period and determining what would have happened anyway without the harvesting and comparing that impact with the carbon trajectory associated with harvesting of biomass for bioenergy. Although any "business as usual" projection would be uncertain, it is the only means by which to gauge the incremental impact of woody biomass harvesting. The *Framework* discusses this anticipated future baseline approach but does not attempt it. Instead a fixed reference point and an assumption of geographic regions were chosen to determine the baseline for whether biomass harvesting for bioenergy facilities is having a negative impact on the carbon cycle. The choice of a fixed reference point may be the simplest to execute, but it does not properly address the additionality question, i.e., the extent to which forest stocks would have been growing or declining over time in the absence of bioenergy. The agency's use of a fixed reference point baseline coupled with a division of the country into regions implies that forest biomass emissions could be granted an exemption simply because the location of a stationary facility is in an area where forest stocks are increasing. The reference point estimate of regionwide net emissions or net sequestration does not indicate, or estimate, the difference in greenhouse gas emissions (the actual carbon gains and losses) over time that stem from biomass use. As a result, the *Framework* fails to capture the causal connection between forest biomass growth and harvesting and atmospheric impacts and thus may incorrectly assess net CO_2 emissions of a facility's use of a biogenic feedstock.

A landscape, versus stand or plot, perspective is important because land-management decisions are simultaneous, e.g., harvesting, planting, silvacultural treatments. Thus, there are concurrent carbon stock gains and losses that together define the net implications over time. A landscape level analysis, and BAF calculation, will capture these.

Leakage

Leakage is a phenomenon by which efforts to reduce emissions in one place affect market prices that shift emissions to another location or sector. "Bad" leakage (called "positive" leakage in the literature) occurs when the use of biogenic feedstocks causes price changes which, in turn, drive changes in consumption and production outside the boundary of the stationary source, even globally, that lead to increased carbon emissions. One type of positive leakage could occur if land is diverted from food/feed production to bioenergy production which increases the price of conventional agricultural and forest products in world markets and leads to conversion of carbon -rich lands to crop production and the release of carbon stored in soils and vegetation. The use of biogenic feedstocks can also affect the price of fossil fuels by lowering demand for them and thereby increasing their consumption elsewhere. "Good" leakage (called "negative" leakage in the literature) could occur if the use of biomass leads to carbon-offsetting activities elsewhere. The latter could arise for example, if increased demand for biomass and higher prices generate incentives for investment in forest management, beyond the level needed directly for bioenergy production, which increases net forest carbon sequestration. The assessment of the overall magnitude of leakage, associated with the use of bioenergy for fuel is highly uncertain and differs considerably across studies and within a study, depending on underlying assumptions. It will also differ by feedstock and location. The *Framework's* equation for BAF includes a term for leakage, however the agency did not specify an approach to calculate the value for leakage.

In dealing with leakage, we suggest measuring the magnitude of leakage to the extent possible or at least examining the directionality of net leakage – whether it is positive (leading to increased carbon emissions elsewhere) or negative (leading to carbon offsetting activities). In some cases even net directionality may be hard to establish. This information can be used to develop supplementary policies to control leakage before it occurs. We do not recommend incorporating a measure of leakage in the estimate of BAF which would effectively hold a stationary facility responsible for emissions that are outside its control and occurring due to market effects. There is no literature in the social sciences to show that this is an effective way to control emissions. Moreover, when this is coupled with the uncertainties inherent in measuring it in the first place the net benefits of doing this are even more unclear. Supplementary policies that restrict the types of land and management practices that can be used to grow biomass for bioenergy and the types of feedstocks that can be used can reduce the leakage effects of bioenergy use. In addition, the agency should be alert to leakage that may occur in other media (e.g., fertilizer runoff into waterways) and the need for targeted policies to prevent or abate it.

Implementation details

The EPA's *Framework* was lacking in implementation details. Implementation is crucial and some of the agency's current proposals will be difficult to implement. Data availability and quality, as well as procedural details (e.g., application process, calculation frequency) are important considerations for assessing the feasibility of implementation and scientific accuracy of results.

Consistency with fossil fuel emissions accounting

For comparability, there should be consistency between fossil fuel and biogenic emissions accounting. Fossil fuel feedstock emissions accounting from stationary sources under the Clean Air Act are not adjusted for offsite greenhouse gas emissions and carbon stock changes. Unlike fossil fuels, however, biogenic feedstocks have carbon sequestration that occurs within a relevant timeframe. While EPA's primary goal is to account for this offsetting sequestration, its biogenic emissions accounting should be consistent with emissions accounting for fossil fuels for other emissions accounting categories—including losses, international leakage, and fossil fuel use during feedstock extraction, production and transport. Including some accounting elements for biomass and not for fossil fuels would be a policy decision without the underlying science to support it.

Case Studies

The case studies provided in EPA's *Framework* were useful for informing the reader with examples of how the *Framework* would be applied but they did not fully cover the relevant variation in feedstocks, facilities, regions and land uses that would be required to more fully evaluate the *Framework*. Additional case studies for landfills and waste combustion, dedicated energy crops like switchgrass and a variety of waste feedstocks would have been useful to see the implementation of the *Framework.* Case studies on different cropping systems with different land and soil types, internal reuse of process materials (e.g., black liquor in pulp and paper mills) and municipal solid waste would have greatly aided the SAB's evaluation of the *Framework*.

Recommendations for Revising BAF

The SAB was asked for advice regarding potential revisions to the *Framework*. We recognize the agency faces daunting technical challenges if it wishes to implement the *Framework's* facility-specific BAF approach. If the EPA decides to retain and revise a facility-specific *Framework*, the SAB recommends consideration of the following improvements.

- Develop a separate BAF equation for each feedstock category as broadly categorized by type, region, prior land use and current management practices. Feedstocks could be categorized into short rotation dedicated energy crops, crop residues, forest residues, municipal solid waste, trees/forests with short accumulation times, trees/forests with long accumulation times and agricultural residue, wood mill residue and pulping liquor.
    - For long-accumulation feedstocks like roundwood, use an anticipated baseline approach to compare emissions from increased biomass harvesting against a baseline without increased

biomass demand. For long rotation woody biomass, sophisticated modeling is needed to capture the complex interaction between electricity generating facilities and forest markets and landscape level effects, in particular: market driven shifts in planting, management and harvests; induced displacement of existing users of biomass; land use changes, including interactions between agriculture and forests; and the relative contribution of different feedstock source categories (logging residuals, pulpwood or roundwood harvest).

  - For residues, consider alternate fates (e.g., some forest residues may be burned if not used for bioenergy) and information about decay. An appropriate analysis using decay functions would yield information on the storage of ecosystem carbon in forest residues.
  - For materials diverted from the waste stream, consider their alternate fate, whether they might decompose over a long period of time, whether they would be deposited in anaerobic landfills, whether they are diverted from recycling and reuse, etc. For feedstocks that are found to have relatively minor impacts, the agency may need to weigh ease of implementation against scientific accuracy. 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 BAF, or setting BAFs equal to 0 or possibly negative values in the case where methane emissions are avoided.
  - For short rotation energy crops grown specifically for bioenergy, the anticipated baseline approach should be used to determine soil carbon sequestration. The BAF for such feedstocks could be negative since they have considerable potential to sequester carbon in soils and roots.

- Incorporate various time scales and consider the tradeoffs in choosing between different time scales when estimating the BAF.
- For all feedstocks, develop supplementary policies to reduce carbon leakage based on at least an assessment of the directionality of leakage.

Consider Default BAFs

The SAB was not asked to recommend an approach that was outside the *Framework*, however, given the conceptual and scientific deficiencies of the *Framework* described above, and the prospective difficulties with implementation, the SAB recommends consideration of default BAFs by feedstock category and region. Under EPA's current *Framework*, facilities would use individual BAFs designed to capture the incremental carbon cycle and net emissions effects of their use of a biogenic feedstock. Rather than trying to calculate a BAF at the facility-level, the SAB recommends that EPA consider calculating a default BAF for each feedstock category. With default BAFs by feedstock category, facilities would use a weighted combination of default BAFs based on their particular bundle of feedstocks. The defaults could rely on readily available data and reflect landscape and aggregate demand effects, including previous land use. Default BAFs might also vary by region and current land management practices due to differences these might cause in the interaction between feedstock production and the carbon cycle. The defaults would also have administrative advantages in that they would be easier to implement and update. Default BAFs for each category of feedstocks would differentiate among feedstocks using general information on their role in the carbon cycle. An anticipated baseline would allow for consideration of prior land use, management, alternate fate (what would happen to the feedstock if not combusted for energy) and regional differences. They would be applied by stationary facilities to determine their quantity of biogenic emissions that would be subject to the EPA's greenhouse gas regulations. Facilities could also be given the option of demonstrating a lower BAF for the feedstock they are using. This would be facilitated by making the BAF calculation transparent and based on data readily available to facilities. Properly designed, a default BAF approach could provide incentives to facilities to choose feedstocks with the lower greenhouse gas impacts.

The SAB also explored certification systems as a possible way to obviate the need to quantify a specific net change in greenhouse gases associated with a particular stationary facility. Carbon accounting registries have been developed to account for and certify CO_2 emissions reductions and sequestration from changes in forest management. Theoretically, for the EPA's purposes, a certification system could be tailored to account for emissions of a stationary facility after a comprehensive evaluation. Ultimately, the SAB concluded that it could not recommend certification without further evaluation because such systems could also encounter many of the same data, scientific and implementation problems that bedevil the *Framework*.

Conclusion

Given the need to address the pressing realities of climate change, biomass resources are receiving much greater attention as a potential energy source. According to the U.S. Department of Energy, the U.S. has the capacity to produce a billion dry tons of biomass resources annually for energy uses (U.S. Department of Energy, 2011). As these materials play a greater role in the nation's energy future, it will be increasingly important to have scientifically sound methods to account for greenhouse gas emissions from bioenergy. However, its greenhouse gas implications are more complex and subtle than the greenhouse gas impacts of fossil fuels. Unlike fossil fuels, forests and other biological feedstocks can grow back and sequester CO_2 from the atmosphere. Given the complicated role that bioenergy plays in the carbon cycle, the *Framework* was written to provide a structure to account for net CO_2 emissions. The *Framework* is a step forward in considering biogenic carbon emissions.

The focus of the *Framework* is on point source emissions from stationary facilities with the goal of accounting for any offsetting carbon sequestration that may be attributed to the facility's use of a biogenic feedstock. To create an accounting structure, the agency drew boundaries narrowly in accordance with its regulatory domain. These narrow regulatory boundaries are intended to account for biogenic carbon uptake and release associated with biomass that is combusted for energy purposes. As such, this *Framework* does not consider, nor is it intended to consider, all greenhouse gas emissions associated with the production and use of biomass energy. Ideally, comprehensive accounting for both biogenic and fossil fuels would extend through time and space to estimate the long-term impacts on net greenhouse gas emissions but the agency was constrained by its regulatory authority. To fully estimate net impact that can be attributed to bioenergy, the EPA would need to calculate the net change in global emissions over time resulting from increased use of biomass feedstocks as compared to a future without increased use of biogenic feedstocks. To capture this difference, the boundaries of analysis would need to include all factors in the life cycle of the feedstock and its products although computing global emissions changes for individual facilities has its own daunting challenges.

The boundaries imposed by the EPA's regulatory authority necessarily restrict its policy choices, however economic research has shown that the most cost-effective way to reduce greenhouse gas emissions (or any other pollution) is to regulate or tax across all sources until they face equal marginal costs. Given the agency's authority under the Clean Air Act, the most cost-effective economy-wide solution is not within its menu of choices. The agency's regulation of stationary sources does not include other users of biomass (e.g., consumers of ethanol) that also have impacts on the carbon cycle as well as downstream consumers of products produced by these facilities. Note that EPA can only regulate end-of-stack emissions and thus has to design a system that fits within its regulatory authority.

The agency has taken on a difficult but worthy task and forced important questions. Practical considerations will, no doubt, weigh heavily in the agency's decisions. In fact, any method that might be adopted or considered, including methods proposed by the SAB, should be subject to an evaluation of the costs of compliance and the carbon emissions savings likely to be achieved as compared to both a categorical inclusion and a categorical exclusion. Uncertainties in the assessment of both the costs and the emissions savings should be analyzed and used to inform the choice of policy. The U.S. Department of Agriculture (USDA) also is developing in parallel an accounting approach for forestry and agricultural landowners. It would be beneficial if the EPA and USDA approaches could be harmonized to avoid conflicts and take advantage of opportunities for synergy. In this Advisory, the SAB offers suggestions for how to improve the *Framework* while encouraging the agency to think about options outside its current policy menu. While the task of accounting for biogenic carbon emissions defies easy solutions, it is important to assess the strengths and limitations of each option so that a more accurate carbon footprint can be ascribed to the various forms of bioenergy.

2. INTRODUCTION

Greenhouse gas emissions from the largest stationary sources became subject to regulation under the Prevention of Significant Deterioration (PSD) and Title V Operating Permit Programs of the Clean Air Act in January 2011. To target these regulations, EPA enumerated specific conditions under which these Clean Air Act permitting requirements would apply. Initially, only sources currently subject to the PSD permitting program or Title V (i.e., those that are newly-constructed or modified in a way that significantly increases emissions of a pollutant other than greenhouse gases) would be subject to permitting requirements for their greenhouse gas emissions. For these projects, only greenhouse gas emission increases of 75,000 tons per year (tpy) or more, on a carbon dioxide equivalent (CO_2e) basis, would be subject to technology requirements under the PSD program. As of July 1, 2011, more facilities became subject to regulation based on their greenhouse gas emissions. Specifically, new and existing stationary sources (that are not already covered by the PSD or Title V programs) that emit greenhouse gas emissions of at least 100,000 tpy are subject to greenhouse gas regulation even if they do not exceed the permitting thresholds for any other pollutant. For these facilities, the PSD and Title V requirements would be triggered. The PSD program imposes "best available control technology" requirements to control greenhouse gas emissions. Title V generally does not impose technology requirements but rather requires covered facilities to report an overall compliance plan for meeting the requirements of the Clean Air Act.

EPA's staged-approach to regulating greenhouse gases from stationary sources sought to focus on the nation's largest greenhouse gas emitters and hence "tailored" the requirements of these Clean Air Act permitting programs to cover power plants, refineries, and cement production facilities that meet certain conditions while exempting smaller sources like farms, restaurants, schools and other facilities. The question before the agency, and hence, the motivation for this SAB review, is whether and how to consider biogenic greenhouse gas emissions in determining whether facilities meet certain thresholds (as defined above) for Clean Air Act permitting. Biogenic CO_2 emissions from bioenergy are generated during the combustion or decomposition of biologically based material.

It is in this context that the EPA Office of Air and Radiation requested the EPA's Science Advisory Board (SAB) to review and comment on its *Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources (Framework, September 2011).* The *Framework* considers the scientific and technical issues associated with accounting for emissions of biogenic carbon dioxide (CO_2) from stationary sources and develops a framework to adjust the stack emissions from stationary sources using bioenergy based on the induced changes in carbon stocks on land (in soils, plants and forests). Because of the unique role of biogenic feedstocks in the overall carbon cycle, EPA deferred for a period of three years the application of permitting requirements to biogenic CO_2 emissions from bioenergy and other biogenic stationary sources. In its deferral, EPA committed to conduct a detailed examination of the science and technical issues associated with biogenic CO_2 emissions and submit its study for review by the Science Advisory Board. To conduct the review, the SAB Staff Office formed the Biogenic Carbon Emissions Panel with experts in forestry, agriculture, greenhouse gas measurement and inventories, land use economics, ecology, climate change and engineering.

The SAB was asked to review and comment on (1) the agency's characterization of the science and technical issues relevant to accounting for biogenic CO_2 emissions from stationary sources; (2) the agency's framework, overall approach, and methodological choices for accounting for these emissions;

and (3) options for improving upon the framework for accounting for biogenic CO_2 emissions (See Appendix A: Charge to the SAB Panel).

The Biogenic Carbon Emissions Panel held a face-to-face meeting on October 25 – 27, 2011, and teleconferences on January 27, 2012, March 20, 2012, May 23, 2012 and May 26, 2012. The Panel's draft report was reviewed by the chartered SAB on August 31, 2012. During the course of deliberations, the SAB Panel reviewed background materials provided by the Office of Air and Radiation and considered written and oral comments from members of the public.

3. RESPONSES TO EPA's CHARGE QUESTIONS

3.1. The Science of Biogenic CO_2 Emissions

Charge Question 1: In reviewing the scientific literature on biogenic CO_2 emissions, EPA assessed the underlying science of the carbon cycle, characterized fossil and biogenic carbon reservoirs, and discussed the implications for biogenic CO_2 accounting.

Does the SAB support EPA's assessment and characterization of the underlying science and the implications for biogenic CO_2 accounting?

EPA has done an admirable job of reviewing the science behind the carbon cycle and greenhouse gas emissions and their relationship to climate change, extracting some of the critical points that are needed to create the proposed *Framework*. Figure 2-1 in the *Framework* captures the global carbon cycle showing the flows and pools of carbon. The chapter goes on to describe the task of quantifying the impact of transforming biologically based carbon from a terrestrial storage pool (such as aboveground biomass) into CO_2 via combustion, decomposition or processing at a stationary source. At the same time, there are several important scientific issues that are not addressed in the *Framework*, as well as scientific issues that are briefly discussed but not sufficiently explored in terms of how they relate to the *Framework*. In the following section, the SAB describes a series of deficiencies with the EPA characterization of the science behind biogenic CO_2 accounting and suggests some areas where the science could be strengthened.

Time scale

One fundamental deficiency in the EPA report is the lack of discussion of the different time scales inherent in the carbon cycle and the climate system that are critical for establishing an accounting system. This is a complicated subject because there are many different time scales that are important for the issues associated with biogenic carbon emissions. At the global scale, there are multiple time scales associated with mixing of carbon throughout the different reservoirs on the Earth's surface. When carbon dioxide is released into the air from burning fossil fuels, roughly 45% stays in the air over the course of the following year. Of the 55% that is removed, roughly half is taken up by the ocean, mostly in the form of bicarbonate ion, and the other half is taken up by the terrestrial biosphere, primarily through reforestation and enhanced photosynthesis. The airborne fraction (defined as the fraction of emissions that remains in the air) has been remarkably constant over the last two decades.

There is considerable uncertainty over how the magnitude of ocean and terrestrial uptake will change as the climate warms during this century. If the entire ocean were to instantly reach chemical equilibrium with the atmosphere, the airborne fraction would be reduced to 20 to 40% of cumulative emissions, with a higher fraction remaining in scenarios with higher cumulative emissions. In other words, the ocean chemical system by itself cannot remove all the CO_2 released in the atmosphere. Because carbon uptake by the ocean is limited by the rate of mixing between the shallow and deeper waters, this complete equilibration is expected to take thousands of years. Over this century, if global CO_2 emissions continue to rise, most models predict that ocean uptake will stabilize between 3 to 5 gigatons per year (GtC/y), implying that the fraction of emissions taken up by the ocean will decrease. For the terrestrial biosphere, there is a much wider envelope of uncertainty; some models predict that CO_2 uptake will continue to keep pace with the growth in emissions, while other models suggest that CO_2 uptake will decline, even becoming a net source of CO_2 to the atmosphere if processes such as release of carbon from the tundra or aridification of the tropics were to occur.

Over the time scale of several thousand years, once ocean equilibration is complete and only 20 to 40% of cumulative emissions remains in the atmosphere, dissolution of carbonate rocks on land and on the ocean floor will further reduce the airborne fraction to 10 to 25% over several thousand years to ten thousand years. Excess anthropogenic CO_2 emissions will stay in the atmosphere for more than 100,000 years, slowly drawn down by silicate weathering that converts the CO_2 to calcium carbonate, as well as slow burial of organic carbon on the ocean floor. The size of this "tail" of anthropogenic CO_2 depends on the cumulative emissions of CO_2, with higher cumulative emissions resulting in a higher fraction remaining in the atmosphere.

Another important time scale for considering accounting systems for biogenic carbon emissions is the period over which the climate responds to carbon dioxide and other greenhouse gases. The importance of the timing of emissions depends on whether one uses a global warming limit or a cumulative emissions limit. Some modeling exercises have shown that the probability of limiting warming to 2 °C or below in the twenty-first century is dependent upon cumulative emissions by 2050 (Meinshausen et al. 2009). This suggests that an early phase of elevated emissions from forest biomass could reduce the odds of limiting climate warming if warming is limited to 2 °C. Another climate modeling study has demonstrated that peak warming in response to greenhouse gas emissions is primarily sensitive to cumulative greenhouse gas emissions over a period of roughly 100 years, and, so long as cumulative emissions are held constant, is relatively insensitive to the emissions pathway within that time frame (Allen et al. 2009). What this means is that an intervention in forests or farming that results in either an increase or decrease in storage of carbon or emissions reductions must endure longer than 100 years to have an influence on the peak climate response as long as cumulative emissions from all sources are constant. Conversely, if these changes last less than 100 years, harvesting of biomass for bioenergy resulting in release of carbon dioxide will have a relatively small effect on peak warming. While the harvesting of trees for bioenergy can result in a carbon debt even at the landscape level (Mitchell et al. 2012), this may not reflect potential climate benefits at longer time scales if biomass is regrown repeatedly and substituted for coal over successive harvest cycles (Galik and Abt 2012).

Time scales are also important for individual feedstocks and their regeneration at a more local scale. Given that the EPA's objective is to account for the atmospheric impact of biogenic emissions, it is important to consider the turnover times of different biogenic feedstocks in justifying how they are incorporated into the *Framework*. The fundamental differences in stocks and their turnover times as they relate to impacts on the atmosphere are not well discussed or linked. If a carbon stock is cycling quickly on land and regrowth is sufficient to compensate for carbon losses from harvesting, it may have a beneficial impact when it displaces fossil fuel over successive cycles of growth and harvest (assuming this temporal displacement exceeds 100 years). If the carbon stock, or some part of it, turns over more slowly, if regrowth is not assured or if feedstocks are not being used to continuously displace fossil fuels, the impact on climate worsens.

There is a continuum of carbon stock size and turnover among the biogenic feedstock sources included in the *Framework*, but there is little background discussion of the variation in stock and turnover and how that informs the accounting method. The *Framework* sets up categories of feedstocks based on their source, but these groupings do not translate into differential treatment in the *Framework*. In Table 1, the SAB offers a rudimentary framework for thinking about climate impacts over time for various feedstock groups. The direct climate impact refers to the effect of growing and harvesting the feedstock on the land based carbon stocks. The indirect/leakage effect refers to the effect on carbon emissions that arises because the production of bioenergy competes for land with conventional crops and raises crop prices which, in turn, can lead to changes in land uses like deforestation. Price signals can also lead to cropland expansion in other locations, thus releasing carbon stocks from soil and vegetation. The column labeled "leakage" is explained further in Section 3.3 where the SAB offers some comments on the treatment of "leakage" or the phenomena by which efforts to reduce emissions in one place affect market prices that shift emissions to another location or sector. As shown in Table 1, the time scale matters most for long rotation trees where term refers to the length of rotation of trees. In the case of forest residues, "near term" is the length of time it would take for residue to decompose if left in the forest.

Table 1. Temporal Carbon Effects of Feedstock Groups

Feedstock	Direct Climate Impact		Indirect/Leakage Impact	Comments
	Near Term	Long Term		
Agricultural Residues	+/ 0 -	+/0 -	None	Could be zero if stover removal rates are low. Also depends on nitrogen application rates. Negative if carbon remains sequestered in ash/biochar or if accompanied by carbon capture and storage.
Forest Residues	+ -	0 -	None	Depends on the rate constant of loss, and the interval of residue or slash creation and the alternative use of the residue Negative if carbon remains sequestered in ash/biochar or if accompanied by carbon capture and storage.
Energy Crops/Short Rotation Woody Crops	-	-	Small if grown on idle land /noncropland, positive in the short run otherwise negative in the long run	Depends on the extent of soil carbon sequestration which may be substantial in the short and medium term but reach a plateau in the long term. Also depends on land use history, land management practices
Long Rotation Trees	+	-	Could be negative or positive in the short run; negative in the long run	Depends on harvest rotation and regrowth rates

Negative sign (-) indicates a reduction in greenhouse gas emissions in the atmosphere and/or increase in carbon stocks. Positive (+) sign refers to an increase in greenhouse gas emissions in the atmosphere or a reduction in soil carbon stocks.

Appendix B discusses a set of studies by Cherubini and co-authors (Cherubini et al. 2011, 2012) that provide examples for estimating the temporal distribution of atmospheric impacts from biomass harvesting by framing the analysis in terms of global warming potentials (GWPs) and global temperature potentials (GTPs) for harvested biomass. Figure B-1 in Appendix B, adapted from Cherubini et al. (2012), depicts mean surface temperature changes for a simple contrived comparison of biogenic emissions from a single forest stand over hundreds of years as compared to comparable fossil emissions. While much is assumed regarding global activity (emissions, landscape responses, investment behavior), Figure B-1 demonstrates the importance of the time horizon and the weight to place on temperature increases that occur in the short term versus temperature increases that occur later. As shown in Figure B-1, a 50-year time horizon (or less) would obscure the longer-term climate consequences of bioenergy. The Global Temperature Potential of Biomass, denoted as GTPbio, would continue to decline for time horizons beyond 100 years since there is no net temperature increase after 100 years. The choice of weighting of temperature effects at different time horizons could be influenced by the estimated damages associated with the temperature increases as well as the social rate of time preference for avoiding damages. The discussion by Kirschbaum (2003, 2006) of the impact of temporary carbon storage (the inverse of temporary carbon release from biomass harvesting for bioenergy) points out that the exact climate impact of temporary CO_2 storage (or emissions) depends on the type of impact, as some depend on peak temperature, whereas others, such as melting of polar ice sheets, depend more on time-averaged global temperature. There is no scientifically correct answer when choosing a time horizon, although the *Framework* should be clear about what time horizon it uses, and what that choice means in terms of valuing long term versus shorter term climate impacts.

Disturbance

Because ecosystems respond in complicated ways to disturbances (e.g., harvesting, fire) over long periods of time, and with a high degree of spatial heterogeneity, the state of knowledge about disturbance and impacts on carbon stocks and turnover should be reviewed within the context of relevant time scales and spatial extents. This is highly relevant to producing accurate estimates of biogenic emissions from the land. There is also insufficient treatment given to the existing literature on the impact of different land management strategies on soil carbon, which is important for understanding how carbon stocks may change over many decades.

Non-CO_2 Greenhouse Gases

The *Framework* does not incorporate greenhouse gases other than CO_2. Ideally both fossil fuels and biogenic fuels should be subject to the same emissions accounting to fully capture the difference between the two types of fuels in terms of their greenhouse gas emissions. For biogenic feedstocks, the most important source of non-CO_2 emissions is likely to be N_2O produced by the application of fertilizer (Crutzen et al. 2007). In particular, if the biomass feedstock is from an energy crop that results in different N_2O emissions vis-a-vis other crops, should this be counted? Is it negligible? This issue is not introduced in the science section. N_2O is relatively long-lived (unlike methane) and therefore the climate impacts of heavily fertilized biomass (whether in forests or farms) are greater than non-fertilized biomass. There is a substantial literature on N_2O from fertilizer use that was not discussed in the *Framework*. If the decision to not count non-CO_2 greenhouse gases stems from a need to render the carbon accounting for biogenic sources parallel with fossil fuels, this needs to be explicitly discussed.

3.2. Biogenic CO_2 Accounting Approaches

Charge Question 2: In this report, EPA considered existing accounting approaches in terms of their ability to reflect the underlying science of the carbon cycle and also evaluated these approaches on whether or not they could be readily and rigorously applied in a stationary source context in which onsite emissions are the primary focus. On the basis of these considerations, EPA concluded that a new accounting framework is needed for stationary sources.

2(a). Does the SAB agree with EPA's concerns about applying the IPCC national approach to biogenic CO_2 emissions at individual stationary sources?

The SAB concurs with EPA's rejection of the application of the Intergovernmental Panel on Climate Change (IPCC) national accounting approach to biogenic carbon emissions at individual stationary sources. The IPCC develops guidelines for countries to report their anthropogenic greenhouse gas emissions. These emissions are reported as aggregate numbers by sectors, e.g., the Land-Use change and Forestry Sector, the Energy Sector, Industrial Processes and Product Use, etc. The IPCC's inventory of global greenhouse emissions (i.e., all emissions are counted) is comprehensive in quantifying all emissions sources and sinks, but does not describe linkages among supply chains. In other words, it is essentially a "production-based inventory" or "geographic inventory" rather than a "consumption-based inventory" (Stanton et al. 2011). The IPCC inventory offers a static snapshot of emissions at any given time, but it does not expressly show changes in emissions over time.

A dissenting opinion presented by Dr. Roger Sedjo in Appendix E expresses a preference to exclude bioenergy from greenhouse gas regulation so long as aggregate national forest carbon stocks are rising relative to a fixed point baseline. The SAB notes that such an approach would not be consistent with EPA's responsibility under the Clean Air Act as it would not capture the marginal effect of increased biomass harvesting on forest carbon stocks and atmospheric carbon levels. Specifically, EPA is not charged with regulating regional or national forest carbon stocks: it must regulate stationary facilities. As such, the IPCC inventories, a static snapshot of emissions at any given point in time, are a reporting convention that has no associated connections to policies or implementation. These inventories do not explicitly link biogenic CO_2 emission sources and sinks to stationary sources, nor do they provide a mechanism for measuring changes in emissions as a result of changes in the building and operation of stationary sources using biomass.

2(b). Does the SAB support the conclusion that the categorical approaches (inclusion and exclusion) are inappropriate for this purpose, based on the characteristics of the carbon cycle?

A decision about a categorical inclusion or exclusion will likely involve many considerations that fall outside the SAB's scientific purview, such as legality, feasibility and, possibly, political will. The SAB cannot speak to the legal or full implementation difficulties that could accompany any policy on biogenic carbon emissions but some scientific observations that may inform the Administrator's policy decision are offered below.

The notion that biomass is carbon neutral arises from the fact that the carbon released as CO_2 upon combustion was previously removed from the atmosphere as CO_2 during plant growth. While it is true that emissions from burning a single tree will equal the same amount of carbon sequestered by that tree at a micro level, at a macro level, net emissions will depend upon rates of harvest vis-a-vis rates of sequestration over time. Thus, the physical flow of carbon in the biomass combusted for bioenergy represents a closed loop that passes through a stationary source. Under an accounting framework where life cycle emissions associated with the production and use of biomass are attributed to a stationary source, assuming carbon neutrality of biomass implies that the net sum of carbon emissions from all sources and sinks is zero, including all supply chain and market-mediated effects. Carbon neutrality cannot be assumed for all biomass energy *a priori* (Rabl et al. 2007; Johnson 2009; Searchinger et al. 2009). There are circumstances in which biomass is grown, harvested and combusted in a carbon neutral fashion but carbon neutrality is not an appropriate *a priori* assumption; it is a conclusion that should be reached only after considering a particular feedstock production and consumption cycle. There is considerable heterogeneity in feedstock types, sources, production methods and leakage effects; thus net biogenic carbon emissions will vary considerably.

Given that some biomass combustion could have positive net emissions, a categorical *exclusion* would remove any responsibility on the stationary source for CO_2 emissions from its use of biogenic material from the entire system (i.e., the global economy) and provide no incentive for the development and use of best management practices. Conversely, a categorical *inclusion* would provide no incentive for using biogenic sources that compare favorably to fossil energy in terms of greenhouse gas emissions.

The commentary above merely reflects some scientific considerations. The SAB recognizes that, in reality, the EPA may face difficult tradeoffs between ease of implementation and other goals (e.g., maximizing scientific accuracy by modeling the decomposition of logging residues). While an alternative approach of default Biogenic Accounting Factors (BAFs) is offered for the agency's consideration (see Section 4), the SAB cannot advise the agency on the legal feasibility of any approach.

2(c). Does the SAB support EPA's conclusion that a new framework is needed for situations in which only onsite emissions are considered for non-biologically-based (i.e., fossil) feedstocks?

Through discussions with the Panel at the public meeting, the EPA agreed that this question is redundant with other charge questions and therefore does not require a separate response.

2(d). Are there additional accounting approaches that could be applied in the context of biogenic CO_2 emissions from stationary sources that should have been evaluated but were not?

Several other agencies are developing methods for assessing greenhouse gas emissions by facilities. These methods could inform the approach developed by the EPA. The methods that are being developed include the DOE 1605(b) voluntary greenhouse gas registry targeted to entities, which has many similar characteristics to the approach proposed by EPA for stationary sources. There is also the Climate Action Registry developed in California that uses a regional approach to calculate baselines based on inventory data and may inform the delineation of geographic regions and choice of baselines in the EPA approach. USDA also is developing in parallel an accounting approach for forestry and agricultural landowners. It would be beneficial if the EPA and USDA approaches could be harmonized to avoid conflicts and take advantage of opportunities for synergy.

3.3. Methodological Issues

Charge Question 3: EPA identified and evaluated a series of factors in addition to direct biogenic CO_2 emissions from a stationary source that may influence the changes in carbon stocks that occur offsite, beyond the stationary source (e.g., changes in carbon stocks, emissions due to land-use and land management change, temporal and spatial scales, feedstock categorization) that are related to the carbon cycle and should be considered when developing a framework to adjust total onsite emissions from a stationary source.

3(a). Does SAB support EPA's conclusions on how these factors should be included in accounting for biogenic CO_2 emissions, taking into consideration recent advances and studies relevant to biogenic CO_2 accounting?

The SAB's response to this question differs by feedstock. On balance, the *Framework* includes many important factors but some factors suffer from significant estimation and implementation problems.

For agricultural feedstocks, the factors identified by EPA to adjust the CO_2 emissions from a stationary source for direct off-site changes in carbon stocks are appropriate but suffer from significant estimation and implementation problems. The *Framework* includes factors to represent the carbon embodied in products leaving a stationary source, the proportion of feedstock lost in conveyance, the offset represented by sequestration, the site-level difference in net carbon flux as a result of harvesting, the emissions that would occur "anyway" from removal or diversion of non-growing feedstocks (e.g., corn stover) and other variables. In some cases, energy crops like miscanthus and switchgrass have significant potential to sequester carbon in the soil and be sinks for carbon rather than a source (Anderson-Teixeira et al. 2009). In other cases, the production of bioenergy could result in by-products like biochar which sequester significant amounts of carbon. A large value of the Total Net Change in Site Emissions (SITE_TNC) and/or Sequestered Fraction (SEQP) variables in the accounting equation could result in a negative BAF for such feedstocks. The *Framework* should clarify how a negative BAF would be used and whether it could be used by a facility to offset fossil fuel emissions. Restricting BAF to be non-negative would reduce incentives to use feedstocks with a large sequestration potential.

For waste materials (municipal solid waste, manure, wastewater, construction debris, etc.), the *Framework* assigns a BAF equal to 0 for biogenic CO_2 released from waste decay at waste management systems, waste combustion at waste incinerators or combustion of captured waste-derived CH_4. The *Framework* further states that for any portion of materials entering a waste incinerator that is harvested for the purpose of energy production at that incinerator, biogenic CO_2 emissions from that material would need to be accounted according to the *Framework* calculations. Municipal solid waste biomass is either disposed of in a landfill or combusted in facilities at which energy is recovered. Smaller amounts of certain waste components (food and yard waste) may be processed by anaerobic digestion and composting. The SAB concurs with the *Framework* that the CO_2 released from the decomposition of biogenic waste in landfills, compost facilities or anaerobic digesters could reasonably be assigned a BAF of 0. In addition, given that methane (CH_4) is a more potent greenhouse gas than CO_2, the *Framework* should account for CH_4 emissions from landfills in cases where the methane is not captured. The SAB recognizes that EPA may address methane in other regulatory contexts.

When accounting for emissions from waste sources including logging residue, wood mill waste and pulping liquor, the EPA should recognize that these emissions are part of a larger system that includes co-products with commercial products. For logging residue, wood mill waste and pulping liquor the larger system includes forests, solid wood mills, pulp mills and stationary energy sources. Accounting for greenhouse gases in the larger system needs to track all biomass emissions or forest stock changes and needs to assure they are allocated over time across the outputs (product and co-products) from the system so as to account for all fluxes. Within the larger system, the allocation of fluxes to wood/paper products or to emissions from a stationary source can be supported by scientific reasoning but is ultimately a policy decision. The agency should consider how the *Framework* meets the scientific requirement to account for (allocate) all emissions to products and co-products across the larger system of forest, mills and stationary sources over time.

For roundwood, the calculation of BAF would need to account for the time path of carbon accumulation and emissions from logging residue and apply a landscape perspective. The landscape perspective is important because of simultaneous management decisions that emit and sequester greenhouse gases concurrently and therefore define the net implications over time. The *Framework* recognizes some of the challenges associated with defining the spatial and temporal time scale and in choosing the appropriate baseline. Ultimately, however, the *Framework* chooses an approach that disregards any consideration of the time scales over which biogenic carbon stocks are accumulated or depleted and does not actually estimate carbon stock changes associated with biomass use. Instead the *Framework* attempts to substitute a spatial dimension for time and creates an accounting system that generates outcomes sensitive to the regional scale at which carbon emissions attributed to a stationary source are evaluated.

Below are some comments on particular factors.

<u>Level of Atmospheric Reduction (LAR)</u>: The term refers to the proportional atmospheric carbon reduction from sequestration during feedstock regrowth (GROW) or avoided emissions (AVOIDEMIT) from the use of residues that would have been decomposed and released carbon emissions “anyway.” The scientific justification for constraining the range of LAR to be greater than 0 but less than 1 is not evident since it is possible for feedstock production to exceed feedstock consumption. These two terms are not applicable together for a particular feedstock and representing them as additive terms in the accounting equation can be confusing. Additionally, the value of LAR for forest biomass is sensitive to the size of the region for which growth is compared to harvest.

<u>Loss (L)</u>: This term is included in the *Framework* to explicitly adjust the area needed to provide the total feedstock for the stationary facility. It is a term used to include the emissions generated by the feedstock lost during storage, handling and transit based on the strong assumption that most of the carbon in the feedstock lost during transit is immediately decomposed. To more accurately estimate the actual loss of carbon due to these losses, one would need to model the carbon storage and fluxes associated with the feedstock lost, which are likely to be a function of time. The number of years considered would be a policy decision; the longer the period, the larger the proportion of loss that would be counted. The *Framework* tacitly assumes an infinitely long horizon that results in the release of all the carbon stored in the lost feedstock.

<u>Products (PRODC)</u>: The removal of products from potential gross emissions is justified scientifically; however, the scientific justification for treating all products equally in terms of their impact on emissions is not clear. For some products (e.g., ethanol and paper), the stored carbon will be released rapidly while for other products, such as furniture, it might be released over a longer period of time. The

Framework implicitly assumes that all products have infinite life-spans, an assumption without justification or scientific foundation. For products that release their stored carbon rapidly, the consequences for the atmosphere are the same as for combustion of the feedstock. To precisely estimate the stores of products so as to estimate the amount of carbon released, one would need to track the stores as well as the fluxes associated with product pools. The stores of products could be approximated by modeling the amount stored over a specified period of time.

A second way in which PRODC is used is as a means of prorating all area–based terms such as LAR, SITE-TNC and Leakage. This is potentially problematic because it makes the emissions embodied in co-products dependent on the choice of regional scale at which LAR is estimated. As the size of the region contracts, LAR tends towards zero and the amount of gross emissions embodied in PRODC increases and exacerbates the implications of the scale sensitivity of the LAR value.

<u>Avoided Emissions (AVOIDEMIT)</u>: This term refers to transfers of emissions that would occur "anyway" from removal or diversion of non-growing feedstocks like corn stover and logging residues. In the *Framework*, feedstocks may be mathematically credited with avoided emissions if the residues would have decayed "anyway." Specifically, AVOIDEMIT is added to Growth (GROW) in the numerator in determining the LAR or proportion of emissions that are offset by sequestration or avoided emissions. As with the Loss term, there is an implicit assumption of instantaneous decomposition that appears to be a simplifying assumption. While this may be a convenient assumption, it should be explained and justified. To improve scientific accuracy, the EPA could explore some sample calculations (as described below), taking into account regional differences in decay rates. Once this information is gathered and analyzed, the EPA may then need to make a decision that weighs scientific accuracy against administrative expediency and other factors.

Since the concept reflected in "avoided emissions" is actually "equivalent field-site emissions," it would be clearer to refer to it this way since emissions are not so much avoided as they are shifted to another venue. With residues left in the forest, some of the materials might take decades to fully decompose. For accuracy, the hypothetical store of carbon would have to be tracked. To approximate these stores, one could compute the average amount of carbon remaining after a period of years.

The scientific theory behind losses and stores of ecosystem carbon was developed by Olson (1963) and could be applied to the fate of residues and slash in both forest and agricultural systems. The store of carbon in an ecosystem depends upon the amount of carbon being input (I) and the proportion of carbon lost per time unit, referred to as the rate-constant of loss (k). Specifically the relationship is I/k. In the case of residues or slash that are burned in the field or in a bioenergy facility, the store of carbon is essentially zero because most of the input is lost within a year (k> 4.6 per year assuming at least 99% of the material is combusted within a year). On the other hand, if the residue or slash does not lose its carbon within a year, the store of carbon would be greater than zero and, depending on the interval of residue or slash creation, could be greater than the initial input. Appendix C provides more information on the fate of residue after harvest and landscape storage of carbon. For example, if slash is generated every 25 years (I=100 per harvest area/25=4 per year) and the slash is 95% decomposed within 25 years (k=0.12 per year), one cannot assume a store of zero because the average ecosystem store in this case would actually be 33% of the initial input (4/0.12=33.3). If the input occurred every 5 years (I=100 per harvest/5=20 per year) for the same decay rate-constant, then the average store would be 167% of the initial input (20/0.12=167). Moreover, it cannot be assumed that because the rate-constant of loss (k) is high, that the stores will always be low. That is because the input (I) is a function of the interval of residue or slash generation; the shorter the interval of generation, the higher the effective input because a higher proportion of the forest or agricultural system is contributing inputs. For example, if there is 1 unit of residue/slash generation per harvest, then an annual harvest on a system basis creates 1 unit of material; if there is 1 unit of residue/slash generation per harvest, then a harvest every 10 years creates an average harvest of 0.1 units (1 unit/10 years = 0.1 unit per year). This relationship means that if residue or slash is generated annually and 95% is lost to decomposition in that period, then the forest system could store 33% of the initial input (I/k=1/3). For the values of k usually observed in agricultural setting (50% per year), an annual input would lead to a store in excess of 145% of the initial input (I/k=1/0.69). Burning of this material would cause a decrease in carbon stores analogous to that of reducing mineral soil stores as accounted for in SITE_TNC, but this loss is not accounted for in the proposed *Framework*.

There are several ways in which losses from residue/slash decomposition could be used in the *Framework*. One is to track the annual loss of carbon from decomposition. This would be analogous to tracking the regrowth of feedstock annually, but in this case it would be the annual decomposition loss. The annual decomposition loss would then be credited as equivalent to combustion as fuel. The advantage of this system is that it would track the time course of release. The disadvantage is that it increases transaction costs. An alternative based on a fuelshed (or other larger area) would be to calculate the average fraction of residue or slash that would remain over the harvest interval and subtract that from the amount harvested. The difference between the amount harvested and the amount that would have remained is an index of the equivalent amount of release via decomposition. For example, if 10 metric tons of either residue or slash is created per year in a fuelshed and 65% of the slash would have decomposed on average over a given harvest interval, then decomposition would have been equivalent to a release of 65% of the amount of fuel used (6.5 metric tons). This would mean that 3.5 metric tons that would have been stored was lost by combustion; hence 6.5 metric tons would be credited in the current calculation of LAR. However, if 35% of the slash would have decomposed on average over the harvest interval, then use of 10 metric tons as fuel would reduce carbon stores of residues and slash by 6.5 metric tons. This would result in a so-called "avoided emissions" credit of 3.5 metric tons.

In addition to considering actual decomposition losses, the *Framework* needs to consider the starting point of residue and slash harvest. The carbon released by combustion will be a function of the starting point, with systems that start with residues and slash having a different timeline of release than those that newly create residue and slash. The former will have the release rate linearly related to the harvest interval, whereas the latter will likely have a curvilinear relationship that is a function of the rate-constant of loss (k).

Instead of a simplifying assumption of instantaneous decomposition, a more accurate calculation could be developed that determines a loss rate-constant appropriate to the material and climate to estimate the amount of carbon that could have been stored had the material not been burned. This amount could be approximated by using the relationships developed by Olson (1963) and reducing the number of calculations involved. When approximations are used, they should be checked against more precise methods to determine the magnitude of possible approximation errors. Several mechanisms could be used to simplify the estimation of these numbers, ranging from calculators that require entry of a few parameters (e.g., average amount of residue or slash generated, the area of source material, the interval of harvest) to look-up tables that are organized around the parameters used to generate them. While there is some uncertainty regarding the loss rate-constants, these sorts of parameters are routinely used in scientific assessments of the carbon cycle and their uncertainty is not much greater than any other parameter required by the *Framework*.

The *Framework* should provide guidance on how logging residue will be distinguished from forest feedstock since that will influence the BAF for that biomass and create incentives to classify as much material as possible as residue and slash despite the fact that some of the “residue/slash” material such as cull trees would be “regenerated” via feedstock regrowth.

Total Net Change in Site Emissions (SITE_TNC): This term is the annualized difference in the stock of land-based carbon (above and below ground, including changes in standing biomass and soil carbon) that results on the site where the feedstock is produced.

The estimates of SITE_TNC will be site-specific and will depend on the knowledge about previous history of land use at that site, the specific agricultural or forestry management practices utilized and the length of time over which they have been practiced. To the extent that the use of bioenergy leads to a change in these practices relative to what would have been the case otherwise, it will be important to use an anticipated baseline approach to determine the stock of land based carbon in the absence of bioenergy and to compare that to the stock with the use of bioenergy. As discussed below in response to charge question 4(f), this anticipated baseline could be developed at a regional or national scale and include behavioral responses to market incentives. Alternatively, look-up tables could be developed based on estimates provided by existing large scale models such as CENTURY or Forestry and Agricultural Sector Optimization Model (FASOM) for feedstock based and region specific SITC_TNC estimates.

It should be noted that soil carbon sequestration is not a permanent reduction in CO_2 emissions. The *Framework,* however, treats permanent reductions in emissions, for example, due to a reduction in the LOSS of biomass to be equivalent to reductions due to an increase in soil carbon sequestration which could be temporary. Since soil carbon sequestration is easily reversible with a change in land management practices, the implementation of this *Framework* will need to be accompanied by frequent monitoring to determine any changes in soil carbon stocks and to update the BAF value for a facility.

Sequestration (SEQP): This term from EPA’s *Framework* refers to the proportion of feedstock carbon embodied in post-combustion residuals such as ash or biochar. Including sequestration in the *Framework* is appropriate; however, the approach taken is subject to the same problems as those described for Products. There is no scientific literature cited to support the idea that all the materials produced by biogenic fuel use do not decompose. This is the subject of ongoing research, but it seems clear that these materials do decompose. The solutions to creating a more realistic and scientifically justified estimate are the same as for the Products term (see above).

Leakage (LEAK): The *Framework* includes this term for leakage but is silent on the types of leakage that would be included and how leakage would be measured. EPA representatives said the *Framework* did not provide a quantification methodology for leakage because assessing leakage requires policy- and program-specific details that are beyond the scope of the report. However, there are several conceptual and implementation issues that merit further discussion in the *Framework*.

The use of biogenic feedstocks could lead to leakage by diverting feedstocks and land from other uses and affecting the price of conventional forest and agricultural products, which can lead to indirect land use changes that release or increase carbon stored in soils and vegetation. The use of these feedstocks could also affect the price of fossil fuels by lowering demand for them and increasing their consumption elsewhere (also referred to as the rebound effect on fuel consumption); this would offset the greenhouse gas savings from the initial displacement of fossil fuels by bioenergy (Chen and Khanna 2012). Leakage effects will vary by feedstock and location and could be positive (if they lead to carbon emissions elsewhere) or negative (if they lead to carbon uptake activities). As will be discussed in Section 3.4 [in response to question 4(f)], the latter could arise, for example, if increased demand for biomass and higher prices generate incentives for investment in forest management that increases forest carbon sequestration. Some research has shown that when a future demand signal is strong enough, expectations about biomass demand for energy (and thus revenues) can reasonably be expected to produce anticipatory feedstock production changes with associated changes in land management and land-use (e.g., Sedjo and Sohngen, in press, 2012). Thus price changes can lead to changes in consumption and production decisions outside the boundary of the stationary source, even globally.

While the existence of non-zero leakage is very plausible, the appropriateness of attributing emissions that are not directly caused by a stationary facility to that facility has been called into question (Zilberman et al. 2011). While first principles in environmental economics show the efficiency gains from internalizing externalities by attributing direct environmental damages to responsible parties, they do not unambiguously show the social efficiency gains from attributing economic or environmental effects (such as leakage) that occur due to price changes induced by its actions to that facility (Holcombe and Sobel, 2001). Moreover, leakage caused by the use of fossil fuels is not included in assessing fossil emissions generated by a stationary facility. Liska and Perrin (2009) show that military activities to secure oil supplies from the Middle East lead to indirect emissions that could increase the carbon intensity of gasoline. Thus, the technical basis for attributing leakage to stationary sources and inherent inconsistency involved in including some types of leakage and for some fuels makes the inclusion of leakage as a factor in the BAF calculation a subjective decision. Including some types of leakage (for example, due to agricultural commodity markets) and not others (such as those due to the rebound effect in fossil fuel markets) and for biomass and not fossil fuels would be a policy decision without the underlying science to support it.

Empirically, assessing the magnitude of leakage is fraught with uncertainty. Capturing leakage would entail using complex global economic models that incorporate production, consumption and land use decisions to compare scenarios of increased demand for biogenic feedstocks with a baseline scenario without increased demand. Global models that include trade across countries in agricultural and forest products can aid in determining the leakage effects on land use in other countries. Global models of the forestry sector include Sedjo and Sohngen (2012) and Ince et al. (2011). Existing models would need to be expanded to include the multiple lignocellulosic feedstocks considered in this *Framework* that can compete to meet demand for bioenergy to determine net leakage effects. Methods would then need to be developed to assign leakage factors to individual feedstocks. The existing literature assessing the magnitude of leakage from one use of a biogenic feedstock (corn ethanol) shows that its overall magnitude in the case of leakage due to biofuel production is highly uncertain and differs considerably across studies and within a study depending on underlying assumptions (Khanna et al. 2011; Khanna and Crago 2012). Other feedstock-use combinations would also need to be evaluated. If the magnitude of leakage is plagued with too much uncertainty, if possible, its direction should at least be stated and recognized in making policy choices. Depending on the level of uncertainty, supplementary policies might be possible to reduce leakage due to changes in land use, such as restrictions on the types of land that could be used to produce the biogenic feedstocks and the types of biogenic feedstocks that could be used to qualify for a BAF less than 1. Some of these implementation issues with estimating BAF and leakage will be discussed further in Section 3.4.

3(b). Does SAB support EPA's distinction between policy and technical considerations concerning the treatment of specific factors in an accounting approach?

A clear line cannot be drawn between policy and technical considerations in an accounting approach. In fact, the lack of information on EPA's policy context and the menu of options made it more difficult to fully evaluate the *Framework*. Because the reasonableness of any accounting system depends on the regulatory context to which it is applied, the *Framework* should describe the Clean Air Act motivation for this proposed accounting system, including how the agency regulates point sources for greenhouse gases and other pollutants. The document should make explicit the full gamut of Clean Air Act policy options for how greenhouses gases could be regulated, including any potential implementation of carbon offsets or certification of sustainable forestry practices. The *Framework* also should describe the EPA's legal boundaries regarding upstream and downstream emissions. Technical considerations can influence the feasibility of implementing a policy just as policy options can influence the technical discussion. The two need to go hand in hand rather than be treated as separable.

The *Framework* explicitly states that it was developed for the policy context where it has been determined that a stationary source emitting biogenic CO_2 requires a means for "adjusting" its total onsite biogenic emissions estimate on the basis of information about growth of the feedstock and/or avoidance of biogenic emissions and more generally the carbon cycle. However, in the discussion on the treatment of specific factors it states in several places that this treatment could depend on the program or policy requirements and objectives. Certain open questions described as "policy" decisions (e.g., the selection of regional boundaries, marginal versus average accounting, inclusion of working or non-working lands, inclusion of leakage) made the evaluation of the *Framework* difficult. Clearly, the policy context matters and the EPA's reticence in describing the policy context and in taking positions on open questions (as well as lack of implementation details) meant that the *Framework* was inadequately defined for proper review and evaluation.

Specifically, if the policy context is changed – for example, if carbon accounting is needed to support a carbon cap and trade or carbon tax policy – then the appropriateness of the *Framework* would need to be evaluated relative to alternative approaches such as life cycle analysis for different fuel streams. Modifying how certain factors are measured or included may not be sufficient. In fact, a different *Framework* would likely be needed if a national or international greenhouse gas reduction commitment exists. Furthermore, the BAFs developed for regulating the emissions from stationary sources would likely conflict with measures of greenhouse gas emissions from bioenergy used in other regulations such as California's cap and trade system for regulating greenhouse gases.

Economic research has shown that the most cost-effective way to reduce greenhouse gas emissions (or any other pollution) is to regulate or tax across all sources until they face equal marginal costs. The most cost-effective solution would involve setting carbon limits (or prices) on an economy-wide basis and not selectively for particular sources or sectors. Given the EPA's limited authority under the Clean Air Act, the most efficient economy-wide solution is not within its menu of policy choices. EPA's regulation of stationary sources will exclude other users of biomass that also have equivalent impacts on the carbon cycle as well as downstream emissions from consuming the products produced by these facilities. Note that biogenic emissions accounting would still be an issue even under an economy-wide emissions policy.

3(c). Are there additional factors that EPA should include in its assessment? If so, please specify those factors.

As stated above, for agricultural biomass from energy crops and crop residues, the factors included in the *Framework* capture most of the direct off-site adjustments needed to account for the changes in carbon stocks caused by a facility using agricultural feedstocks although they do not account for leakage. However, an anticipated baseline is needed for soil carbon, residue disposition and land management changes. For forest biomass, the *Framework* needs to incorporate the time path of carbon accumulation in forests (after energy emissions from harvested roundwood) and forest investment and multi-stand decisions. As discussed in Section 3.1, EPA should consider the time path of the "anyway" emissions that would have occurred on the land if logging residue were not used for energy production and weigh the benefits of scientific accuracy against the administrative simplicity of assuming instantaneous decomposition. For municipal solid waste biomass, the *Framework* needs to consider other gases and CH_4 emissions from landfills. Given that methane emissions from landfills are sometimes not captured, crediting waste material for avoided emissions of methane may be inappropriate. As the *Framework* states, the carbon impact of using waste for energy production in combustion facilities should nonetheless be subjected to a biogenic accounting framework. It should be gauged relative to the CH_4 emissions, if any, that would be released during decomposition in a landfill. N_2O emissions, especially from fertilizer use, should also be considered. Furthermore, the inclusion of non-CO_2 greenhouse gases in general should be consistent between biogenic and fossil fuel accounting. For instance, there are also transportation -related emissions losses in the delivery of natural gas.

3(d). Should any factors be modified or eliminated?

For reasons discussed above, factors such as PRODC, AVOIDEMIT and SEQP could be improved by incorporating the time scale over which biomass is decomposed or carbon is released back to the atmosphere. LAR needs to be modified to be scale insensitive and to address additionality. Factors can be separated by feedstocks according to their relevance for accounting for the carbon emissions from using those feedstocks. For example, GROW and leakage may not be relevant for crop and forest residues.

3.4. Accounting Framework

Charge Question 4: EPA's Accounting Framework is intended to be broadly applicable to situations in which there is a need to represent the changes in carbon stocks that occur offsite, beyond the stationary source, or in other words, to develop a "biogenic accounting factor" (BAF) for biogenic $C0_2$ emissions from stationary sources.

Question 4(a). Does the Framework accurately represent the changes in carbon stocks that occur offsite, beyond the stationary source (i.e., the BAF)?

For agricultural biomass, the variables in EPA's proposed equation for BAF represent the basic factors necessary for estimating the offsite carbon change associated with stationary source biomass emissions, including changes in storage of carbon at the harvest site. For short accumulation feedstocks, where carbon accumulation and "anyway" emissions are within one to a few years (i.e., agricultural residues, perennial herbaceous crops, mill wood wastes, other wastes), with some adjustments to address estimation problems (including an anticipated baseline for soil carbon changes, residue disposition and land management) and careful consideration of data and implementation, the *Framework* can accurately represent carbon changes offsite. However, for long accumulation feedstocks where carbon accumulation and "anyway" emissions occur over decades [i.e., wood harvested specifically for energy use (roundwood) and logging residue], the *Framework* does not accurately account for changes in carbon stocks offsite for several reasons discussed below in response to charge question 4(b).

The *Framework* also does not consider other greenhouse gases (e.g., N_2O from fertilizer use and CH_4 emissions from landfills). Excluding CH_4 because it is not "CO_2" is not a legitimate rationale. It would need to be included to estimate the "difference in carbon dioxide equivalent (CO_2e) the atmosphere sees." In addition, excluding CH_4 emissions from landfills is inconsistent with the *Framework's* purpose of accounting for displaced on-site changes in CO_2. For the same reasons, the basis for excluding N_2O emissions from biomass production is unclear. It also needs to be included to estimate the net changes in atmospheric greenhouse gases. Accounting for N_2O from fertilization would be consistent with tracking changes in soil carbon which are a response to agricultural management systems that include fertilizer decisions.

Question 4(b). Is the Framework scientifically rigorous?

The SAB did not find the *Framework* to be sufficiently comprehensive. Specifically, the SAB identified a number of deficiencies that need to be addressed.

Time scale: As discussed previously, one deficiency in the *Framework* is the lack of proper consideration of the different time scales inherent in the carbon cycle and the climate system that are critical for establishing an accounting system. This is a complicated subject because there are many different time scales that are important for the issues associated with biogenic carbon emissions.

Scientific understanding of the time scale over which the climate system responds to cumulative emissions implies that the carbon release caused by harvesting and combusting biomass at stationary sources is a serious problem if carbon storage, on average, is reduced over long periods of time. So long as rates of growth across the landscape are sufficient to compensate for carbon losses from harvesting over the long run, the climate system is less sensitive to the imbalance in the carbon cycle that might occur in the short run from harvesting of biomass for bioenergy facilities. A scientifically rigorous evaluation of the impact of biomass harvest on the carbon cycle should consider the temporal characteristics of the cycling as well as the spatial simultaneous decisions made across stands and plots. Annual accounting of carbon stocks, while helpful in tracking net carbon emissions, is likely to give an inaccurate assessment of the overall climate and atmospheric carbon cycle impacts.

The *Framework* also does not consider the length of time it takes ecosystems to respond to disturbances, such as those due to the harvesting of biomass, nor does it consider the spatial heterogeneity in this response. This has implications for the accuracy with which the impact of different land management strategies on carbon stocks in soil and vegetation is estimated.

The *Framework* subtracts the emissions associated with products – including ethanol, paper, and timber – from the calculation of emissions from a stationary source, through the PRODC term. While the EPA may not have the discretion to treat all emissions equally, distinguishing between immediate emissions from the facility and downstream emissions (as these products will inevitably be consumed within a short period of time) does not make sense scientifically. From the perspective of the carbon cycle and the climate system, all these facilities extract biomass from the land and the vast majority of that biomass is converted to carbon dioxide, adding to cumulative emissions and, hence, a climate response.

Spatial scale: There is no peer reviewed literature cited to support the delineation of spatial scales for biogenic CO_2 accounting and different carbon pools to be accounted for at different spatial scales. For example, the atmospheric impact of feedstocks is gauged on a regional basis in terms of its impact on forest carbon stocks (except for case study 5) while impacts due to land use change are accounted for at the site level.

The *Framework's* use of a regional scale for accounting for the net changes to the atmosphere is an artificial construct developed to (a) avoid the need for site-specific chain of custody carbon accounting with separate streams for each feedstock and (b) as an alternative to capturing changes in carbon stocks over time. The calculation of LAR uses regional landscape wide carbon changes but does not actually estimate changes attributable to biomass demand (see next discussion). This approach attempts to simplify implementation using available forest inventory data and circumvents the need for accounting for changes in carbon stocks specific to the site or feedstock sourcing region (fuelshed), which may be more complex, costly and difficult to verify. However, as noted, it doesn't provide an actual estimate of carbon changes due to stationary source biomass demand, and it makes the estimate of the BAFs sensitive to the choice of the spatial region chosen for accounting purposes. As shown by case study 1, there are significant implications of this choice for the emissions attributed to a facility.

Additionality: A key question is whether the harvesting of biomass for bioenergy facilities is having a negative impact on the carbon cycle relative to emissions that would have occurred in the absence of biomass usage. This requires determining what would have happened anyway without the harvesting and comparing the impact with the increased harvesting of biomass for bioenergy in order to isolate the incremental or additional impact of the bioenergy facility. While the *Framework* discusses the "business as usual" or "anticipated future baseline" approach, it implements a reference point approach that assesses carbon stocks on a regional basis at a given point in time relative to a historic reference carbon stock.

For forest carbon stocks, the choice of a fixed reference point may be the simplest to execute, but it does not actually address the question of the extent to which forest stocks would have been growing/declining over time in the absence of a particular bioenergy facility. The use of a fixed reference point baseline implies that forest biomass emissions could be considered carbon neutral if forest stocks are increasing. This is simply an artifact based on the choice of the baseline that will be used. The problem is thus: a region with decreasing carbon stocks may in actuality have greater carbon stocks than it would have had without the increased harvesting of biomass. Similarly, a region with increasing carbon stocks may have less stores of carbon than would be the case without the facility using biomass. By default, this approach creates "sourcing" and "non-sourcing" regions. Thus, a carbon accumulating region is a "source" of in situ carbon that can be given to support biomass use, and a carbon losing region is a "non-source" of carbon and cannot support biomass use. The reference year approach provides no assurances at all that a "source" region is gaining carbon due to biomass use, or that a "non-source" region is losing carbon due to biomass use.

For example, for roundwood use under the *Framework*, a region may have carbon accumulation with respect to the reference year (and be assigned LAR=1 according to the *Framework*); however, harvest of a 150+ year old forest in the region for energy production would not be counted in a facility's greenhouse gas emissions even though there is less carbon storage than there would have been otherwise and only a portion of the forest's carbon would be recovered within the next 100 years. To estimate the "difference in atmospheric greenhouse gases" over some period, one must estimate how carbon accumulation differs between a biomass use case and a case without biomass use (business as usual case).

Assessing uncertainty: The *Framework* acknowledges uncertainty but does not discuss how it will be characterized and incorporated to assess the potential uncertainty in the estimate of the BAF value. Selecting an acceptable risk level is a policy decision but characterizing uncertainty and risks is a scientific question. There are numerous drivers that can change biogenic carbon stocks, even in the absence of biomass harvesting for energy. These include changes in economic conditions, domestic and international policy and trade decisions, commodity prices, and climate change impacts. There is considerable uncertainty about the patterns of future land use, for example, whether land cleared for bioenergy production will stay in production for decades to come. The potential impact of these forces on biogenic carbon stocks and the uncertainty of accounting need to be considered further. Ideally, the EPA should put its BAF estimates into context by characterizing the uncertainties associated with BAF calculations and estimating uncertainty ranges. This information can be used to give an indication of the likelihood that the BAFs will achieve the stated objective. The uncertainty within and among variables for any estimate may vary widely between feedstocks and across regions. Finally, it should be pointed out that while parameter uncertainty is important to consider throughout the *Framework*, alternative policy options (e.g., categorical inclusion and exclusion) do not have parameter uncertainty yet their effect on atmospheric carbon is also uncertain.

Leakage: The *Framework* states that the likelihood of leakage and the inclusion of a leakage term will be based on a qualitative decision. There is essentially no guidance in the document about how leakage might be quantified and no examination of the literature regarding possible leakage scenarios (consider Murray et al. 2004). A number of statements/assumptions were made regarding the area and intensity of wood harvest increases to accommodate biomass access. There was no examination of the scientific literature on wood markets and therefore no science-based justification for these statements/assumptions.

Other areas: Other areas that require more scientific justification include assumptions regarding biomass losses during transport and their carbon implications, the choice of a 5-year time horizon instead of one that considered carbon cycling, and the decision to include only CO_2 emissions and exclude other greenhouse gas emissions. Additionally, assumptions about the impacts of harvests on soil carbon and land use changes on carbon sequestration need to be more rigorously supported.

Inconsistencies: Below are some inconsistencies within the *Framework* that should be resolved or justified:

(1) Consistency with fossil fuel emissions accounting: Fossil fuel feedstock emissions accounting from stationary sources under the Clean Air Act are not adjusted for offsite greenhouse gas emissions and carbon stock changes. Does that imply that by default BAFs should be zero as well? No, because, unlike fossil fuels, biogenic feedstocks have carbon sequestration that occurs within a timeframe relevant for offsetting CO_2 emissions from the biomass combustion. For comparability, however, biomass and fossil fuels emissions accounting should be similar for other emissions categories. These include non-CO_2 greenhouse gas emissions, losses, leakage, and fossil fuel use during feedstock extraction, production and transport. This issue is also discussed in Section 3.3.1.

(2) Biogenic and fossil fuel emissions accounting for losses: The *Framework's* handling of carbon losses during handling, transport, and storage introduces an inconsistency between how fossil emissions are counted at a stationary source and how biomass emissions are counted. For biomass emissions the *Framework* includes emissions associated with loss of feedstock between the land and the stationary source. For natural gas the emissions attributed to the stationary source do not include fugitive greenhouse gas emissions from gas pipelines. Why would loss emissions be included for biomass when they are not included for natural gas?

(3) Inconsistency in the consideration of land management and the associated greenhouse gas flux accounting: The *Framework* accounts for soil carbon stock changes, which are a function of the land management system, soil, and climatic conditions. However, it does not account for the non-CO_2 greenhouse gas changes like N_2O that are jointly produced with the soil carbon changes. Soil carbon changes influence both the below and above ground carbon stock changes associated with changes in the land management system.

(4) Reference year and business as usual (BAU) baseline use: The *Framework* proposes using a reference year approach: however, it implicitly assumes projected behavior in the proposed approach for accounting for soil carbon changes and municipal waste decomposition.

(5) Definition of soil. There is a good deal of variation in the *Framework* as to the definition of "soil." At one point it appears to be defined as all non-feedstock carbon such as slash, surface litter, and dead roots as well as carbon associated with mineral soil. In other places, the *Framework* seems only to consider the carbon associated with mineral soil. Unfortunately this inconsistency in the use of the term "soil" creates confusion regarding interpretation and implementation. When soil is defined as non-feedstock carbon (that is all forms of dead carbon) and then implemented as mineral soil carbon (one form of dead carbon), it is impossible to ensure a mass balance as dead material above- and belowground is accounted for in one place, but then not elsewhere. Inconsistent definitions of soil carbon mean that statements regarding the impact of management cannot be unequivocally assessed. For example, if the broader definition

of soil is being invoked, then the statement that management of forests can reduce soil carbon could be justified (Harmon et al. 1990; Johnson and Curtis 2001). However, if the narrower definition of mineral soil carbon is being invoked, then there is very little empirical evidence to justify this statement (Johnson and Curtis 2001); and in fact there is evidence that forest management can at least temporarily increase mineral soil carbon.

Soil carbon should be defined and used consistently throughout the document. If defined broadly, then consistent use of subcategories would eliminate much confusion. For example, if organic horizons such as litter are part of the soil, then consistently referring to total soil, organic soil horizons, and mineral horizons would be essential. Had that been done, the confusion about the impact of forest management on soil carbon would have been eliminated as management can greatly influence organic horizons, but have little effect on mineral horizons. If defined narrowly to only include mineral soil, then the EPA should develop a terminology for the other carbon pools (e.g., organic horizons, aboveground dead wood, and belowground dead wood) that ensures that mass balance is possible.

To define soil carbon, EPA should consider the merits of an aggregated soil term versus subcategories based on source of the carbon, the controlling processes, and their time dynamics. While the aggregated term "soil" is simple, it potentially combines materials with very different sources, controlling processes, and time dynamics, creating an entity that will have extremely complex behavior. It also creates the temptation of a broad term being used for a subcategory. Separating into woody versus leafy materials would account for different sources and to some degree time dynamics. In contrast, separating into feedstock versus non-feedstock material (as appears to be done in the *Framework*) creates a poorly defined boundary as woody branches would be soil if they are not used, but could be viewed as not being soil if they are. A feedstock-based system also does not separate materials into more uniform time dynamics (if leaves and wood are not harvested, then materials with lifespans that differ an order of magnitude are combined). Controlling processes, be they management or natural in nature, differ substantially for above- versus belowground carbon; hence they should be divided.

Underlying the need for a clear definition of soil in the document is the complexity of soil outcomes that differ based on conditions. Some noteworthy publications from forest soil science might have informed the *Framework's* treatment of soil carbon in forest ecosystems (Alban and Perala 1992; Mattson and Swank 1989; Binkley and Resh 1999; Black and Harden 1995; Edwards and Ross-Todd 1983; Gilmore and Boggess 1963; Goodale et al. 2002; Grigal and Berguson 1998; Homann et al. 2001; Huntington 1995; Johnson and Curtis 2001; Laiho et al. 2003; Mroz et al. 1985; Nave et al. 2010; Richter et al. 1999; Sanchez et al. 2007; Schiffman and Johnson 1989; Selig et al. 2008; Tang et al. 2005; Tolbert et al. 2000).

Question 4(c). Does the Framework utilize existing data sources?

First, and most importantly, the *Framework* does not provide implementation specifics. Therefore, it is difficult to assess data availability and use. These issues are discussed here and in the sections that follow.

A more meaningful question is "Are the proposed data sets adequate to account for the effects of biogenic carbon cycling on CO_2 emissions from a facility?" The *Framework* does use existing data, but the data are not adequate to attribute emissions to a facility. For example, the *Framework* mentions the use of the USDA Forest Service's Forest Inventory and Analysis (FIA) data at some unspecified scale. However, carbon stock change data are likely not very accurate at the scale of the agricultural or forest feedstock source area for a facility.

The *Framework* requires data and/or modeling of land management activities and their effects on CO_2 emissions and stock changes. For example, for agricultural systems, data are required on the type of tillage and the effect of such tillage on soil carbon stocks for different soil types and climatic conditions. Such data are not likely to be available at the required scales. In one of the case studies, for example, the Century model is used to model soil carbon stocks. Is the use of this particular model proposed as a general approach to implement the *Framework*? Since this model generally addresses soil carbon only to a depth of 20 centimeters, does that represent a boundary for the *Framework*? Recent work has shown that such incomplete sampling can grossly misestimate changes in soil carbon for agricultural practices such as conservation tillage (Baker et al. 2007; Kravchenko and Robertson 2011). Which version of the model would be run? Would EPA run this model and select parameters appropriate for each feedstock production area for each facility? How robust are the predictions of this model for the range of soils, climatic conditions, and management practices expected to be covered by the *Framework*? Could some other model be used that produces different results for a given facility?

The *Framework* implies that data are required from individual feedstock producers. Collecting such data would be costly and burdensome. Additionally, to the extent that feedstocks are part of commodity production and distribution systems that mix material from many sources, it is not likely to be feasible to determine the source of all feedstock materials for a facility.

The *Framework* includes a term for leakage but eschews the need to provide any methodology for its quantification. Example calculations are carried out for leakage in one of the case studies without any explanation for their source. However, leakage can be positive or negative, and while many publications speculate about certain types of leakage, no data are presented, nor are data sources for different types of leakage suggested or discussed. The *Framework* does provide an example calculation of leakage in the footnote to a case study, but this does not a substitute for a legitimate discussion of the literature and justification and discussion of implications of choices. In addition, such data are unlikely to be available at the scales required. The implications and uncertainties caused by using some indicator or proxy to estimate leakage need to be discussed. If leakage cannot be estimated well, is it possible to put an error range on the leakage value (e.g., a uniform distribution) and assess the impact of this uncertainty on the overall uncertainty in the BAF value? For some cases, such as the conversion of agricultural land to biomass production from perennial crops, leakage may be described as likely increasing net emissions. In cases such as this where prior research has indicated directionality, if not magnitude, such information should be used. As previously noted, there is also a consistency issue with the reference year approach because leakage estimation will require an anticipated baseline approach of some sort.

In summary, it is not clear that all of the data requirements of the *Framework* can be met. Furthermore, even if the data are acquired, they may not be adequate to attribute emissions to a facility.

Question 4(d). Is it easily updated as new data become available?

In principle it would be feasible to update the calculations as new data become available. Some kinds of data, such as those from FIA, are updated periodically and could be used to update the analysis.

However, as discussed for other sub-questions, it is not clear exactly what data and resolution are required and whether all the required data are readily available.

The *Framework* uses an annual or five-year interval for updating calculations. For some kinds of data, such as soil and forest carbon stocks, this interval is too short to detect significant changes based on current or feasible data collection methodologies. This implies that statistical or process models would be used to estimate short-term changes for reporting purposes.

Lastly, if BAF is not under the control of the facility, frequent calculation of the BAF would introduce considerable uncertainty for the facility. This would particularly be the case if a leakage factor were included in the BAF and would need to be updated frequently with changes in market conditions. However, if the accounting is infrequent, shifts in the net greenhouse gas impact may not be captured. Clearly, the EPA will have to weigh tradeoffs between the accuracy of greenhouse gas accounting and ease of implementation and other transactions costs.

Question 4(e). Is it simple to implement and understand?

It is neither. While the approach of making deductions from the actual emissions to account for biologically based uptake/accumulation is conceptually sound, it is not intuitive to understand because it involves tracking emissions from the stationary source backwards to the land that provides the feedstock rather than tracking the disposition of carbon from the feedstock and land forwards to combustion and products. The *Framework* also appears to be difficult to implement, and possibly unworkable, especially due to the many kinds of data required to make calculations for individual facilities. Additionally, the factors (variable names) in the *Framework* do not match those used in the scientific literature and may be misunderstood. Lastly, many elements of the *Framework* are implicit rather than explicit. For example, the time frame during which changes in atmospheric greenhouse gases will be assessed is not explicit. The time frame for specific processes is often implicit, such as the emissions of CO_2 from biomass that is lost in transit from the production area to the facility; this loss is assumed to be instantaneous.

Much more detailed information is required about how the *Framework* would be implemented. It would be helpful to know the specific data sources and/or models to be used. To assess the adequacy of data, more information is needed on implementation and the degree of uncertainty acceptable for policymakers to assign BAF values.

Question 4(f). Can the SAB recommend improvements to the framework to address the issue of attribution of changes in land-based carbon stocks?

The *Framework* uses a reference year baseline approach to determining BAF in combination with a regional spatial scale. As mentioned in response to charge question 4(b), this approach is not adequate in cases where feedstocks accumulate over long time periods because it does not allow for the estimation of the incremental effect on greenhouse gas emissions over time of feedstock use. To gauge the incremental effect on forest carbon stocks due to the use of forest-derived woody biomass (specifically, the value of the LAR), an anticipated baseline approach is needed. This involves estimating a “business as usual” trajectory of emissions and forest stocks and comparing it with alternate trajectories that incorporate increased demand for forest biomass over time. The anticipated baseline approach should also be applied to determine soil carbon for all types of feedstocks for forest types, soils, residue, waste disposition and land management. An anticipated baseline approach (comparing “with” and “without” scenarios) was used by EPA in the development of its Renewable Fuel Standard (*Federal Register*, 2012).

An anticipated baseline approach must incorporate market effects even when direct effects of the use of biogenic feedstocks on carbon emissions are being estimated. The projected baseline level of forest carbon stocks will need to be compared with the level in the case when there is demand for roundwood for bioenergy to assess the change in forest stocks due to the demand for bioenergy. The case with demand for bioenergy should consider the possibility that investment in long-lived trees could be driven by expectations about wood product prices and biomass prices, leading landowners to expand or retain land in forests, plant trees, shift species composition, change management intensity and adjust the timing of harvests. The role of demand and price expectations/anticipation is well developed in the economics literature (e.g., see Muth 1992) and also in the forest modeling literature (Sedjo and Lyon 1990; Adams et al. 1996; Sohngen and Sedjo 1998), which includes anticipatory behavior in response to future forest carbon prices and markets (Sohngen and Sedjo 2006; Rose and Sohngen 2011). The U.S. Energy Information Administration (EIA) has projected rising energy demands for biogenic feedstock based on market and policy assumptions, which could be met from a variety of sources, including energy crops and residues, but also short rotation woody biomass and roundwood (EIA 2012; Sedjo 2010; Sedjo and Sohngen 2012). The extent to which price expectations and anticipation of future demand for bioenergy are going to drive forest management decisions, and regional variations in them, would need to be empirically validated. One study shows forest carbon change in a decade (and thereafter) that exceeds the modeled increased cumulative wood energy emissions over the decade (Sedjo and Tian, in press, 2012). This would be the case if demand is anticipated to increase in the future. Some other modeling studies suggest more limited responses to increased wood energy demand that differ across regions. One such model for the United States indicates a large response in the South, in the form of less forest conversion to non-forest use, but much less response in the North and West (USDA FS 2012; Wear 2011).

To capture both the market, landscape and biological responses to increased biomass demand, a bioeconomic modeling approach is needed with sufficient biological detail to capture inventory dynamics of regional species and management differences as well as market resolution that captures economic response at both the intensive (e.g., changing harvest patterns, utilization or management intensity) and extensive margins (e.g., land use changes). While several models have these features [USDA Forest Service Resources Planning Act (RPA) models in Wear 2011; Sub-regional Timber Supply in Abt et al. in press 2012; Forest and Agricultural Sector Optimization Model (FASOM) in Adams et al. 2005; and the Global Timber Market Model (GTMM) in Sohngen and Sedjo 1998], they differ in scope, ecological and market resolution, and how future expectations are formed. FASOM and GTMM employ dynamic long term equilibria that adopt the rational expectations philosophy that decisions incorporate expectations about future prices and market opportunities. In the RPA and SRTS models, agents respond to current supply, demand, and price signals so that expectations are assumed to be driven by current market conditions. While the rational expectations approach has internal logical consistency and can better simulate long-term structural change, it is not designed for prediction but instead to evaluate potential futures and deviations between futures. These models should incorporate the multiple feedstocks (including crop and logging residues) from the agricultural and forest sectors that would compete to meet the increased demand for bioenergy.

Energy policies can influence the mix of feedstocks used, such as the use of logging residues and the level of projected traditional wood demand, and thus the impact of woody bioenergy demand on timber markets (Daigneault et al. in press 2012). A lower level of timber demand from pulp and paper mills and sawmills, for example, will lead to lower harvest levels and fewer available logging residues. If only residues are allowed to qualify as renewable, then the woody bioenergy industry is explicitly tied to the future of the traditional wood industries. However, if roundwood is used for bioenergy, then the market outcome is more complicated. A lower level of traditional harvest could lead to fewer available residues (which could raise the price of residues and set a physical upper limit on residue supply), but could also lead to higher inventory levels and lower roundwood prices, which would favor increased roundwood utilization for bioenergy. Modeling the interaction across traditional wood consumers, bioenergy consumers, changes in the utilization and mix of products and the displacement of one wood consumer by another as markets evolve will be difficult, but could have a significant impact on the estimate of the carbon consequences of bioenergy use.

As with any modeling, uncertainties will need to be assessed. Models that include price expectations effects or the impact of current year prices would need to be validated. However, validation means different things for different kinds of models. For an econometric model, reproducing history is a form of validation, as is evaluating errors in near-term forecasts. Simulation models are not forecast models. They are designed to entertain scenarios. Validation for simulation models is evaluating parameters and judging the reasonableness of model responses – both theoretically and numerically – given assumptions. Evaluation will help improve representation of average forest and agricultural land management behavior. Evidence affirming or indicating limitations of the effect of prices on investment in retaining or expanding forest area across various U.S. regions may be found by a review of empirical studies of land use change.

Selection of an appropriate model requires judgment and understanding of the structure and assumptions of alternative models and their strengths and weaknesses. This could be supplemented with one or more approaches to choosing a model. These include validation of existing models at the relevant temporal and spatial scale by a means appropriate to the model type, as well as using more than one model to compare and triangulate outcomes. Note that models of different types (e.g., projections vs. forecasting models) require different types of evaluation.

The anticipated baseline approach could be based on a national/global scale model or a regional scale after weighing the strengths and weaknesses of the two approaches. An example of a regional scale model is that by Galik and Abt (2012) where they tested the effects of various scales on greenhouse gas outcomes and found that in the southern United States, market impacts (negative leakage) had a significant impact on forest carbon impacts, but the results were dependent on time period evaluated and were particularly sensitive to scale. The authors evaluated carbon consequences of bioenergy impacts from stand level to state level and found that as scale increased, market responses mitigated forest carbon impacts. In addition to being sensitive to scale, another disadvantage of the regional scale models is that they would not account for leakage across different regions. However, regional models can incorporate greater heterogeneity in forest growth rates, their carbon impacts and in the price responsiveness of forest management decisions. The SAB has not conducted a detailed review of these models to suggest which model and which scale would be the most appropriate.

While market effects are important, there is value in making separate estimates of biological land carbon changes alone (without market effects). Specifically, biophysical process response modeling results are a critical input to economic modeling. Ecosystem modeling is not a substitute for economic modeling, which is necessary to estimate behavioral changes driven by biomass feedstock demand that drives changes in emissions and sequestration. Ecosystem modeling would establish carbon storage in the absence of positive or negative leakage and may have lower uncertainty – especially for logging residue

– than the estimate with leakage. Appendix D depicts three biological scenarios for the total carbon storage in a forest system, including live, dead, and soil stores of carbon. Graphically, Figure D-2 in Appendix D shows how the storage of carbon in a forest system could respond to a shorter harvest interval. Note that all graphs in Appendix D show the biological response and do not account for management changes that could be induced through markets or policies.

Modeling physical land carbon responses over time (without market effects) would show how carbon storage varies by such factors as length of harvest rotations, initial stand age and density, thinning fraction, and growth rates. These carbon responses to management decisions are important inputs for economic modeling of management changes and their carbon consequences. Such modeling could also include the effect of avoided fire emissions on forest land due to biomass removal. This information could indicate what forest conditions and practices could provide higher rates of accumulation, information that might be helpful for EPA in designing its policy response so that incentives could be provided to favor harvest in areas with a higher likelihood of carbon accumulation.

Question 4(g). Are there additional limitations of the accounting framework itself that should be considered?

A number of important limitations of the *Framework* are discussed below:

Framework ambiguity: Key *Framework* features were left unresolved, such as the selection of regional boundaries (the methods for determining as well as implications), marginal versus average accounting, inclusion of working or non-working lands in the region when measuring changes in forest carbon stocks, inclusion/exclusion of leakage, and specific data sources for implementation. As a result, the *Framework's* implementation remains ambiguous. The ambiguity and uncertainty in the text regarding what are stable elements versus actual proposals also clouded the evaluation. If the EPA is entertaining alternatives and would like the SAB to comment on alternatives, then the alternatives should be clearly articulated and the proposed *Framework* and case studies should be presented with alternative formulations to illustrate the implementation and implications of alternatives.

Feedstock groups: The proposal designates three feedstock groupings. However, it is not clear what these mean for BAF calculations, if anything. The *Framework* does not incorporate the groupings into the details of the methodology or the case studies. As a result, it is currently impossible to evaluate their implications.

Potential for Unintended consequences: The proposed *Framework* is likely to create perverse incentives for investors and land-owners and result in unintended consequences. For investors, the regional baseline reference year approach will create regions that are one of two types — either able to support bioenergy from forest roundwood (up to the gain in carbon stock relative to the reference year), or not. As a result, a stationary source investor will only entertain keeping, improving, and building facilities using biomass from regions designated as able to support bioenergy. However, as noted previously, regions losing carbon relative to the reference year could actually gain carbon stock in relative terms due to improved biomass use and management to meet market demands. In addition, the definitions of regions would need to change over time. The designation of regions (and their corresponding LARs) that comes from the reference year approach will create economic rents and therefore financial stakes in the determination of regions and management of forests in those regions.

The proposed *Framework* could also create perverse incentives for landowners. For instance, landowners may be inclined to clear forest land a year or more in advance of growing and using energy crops. Similarly, landowners may be more inclined to use nitrogen fertilizers on feedstocks or other lands in conjunction with biomass production. Such fertilization practices have non-CO_2 greenhouse gas consequences (specifically N_2O emissions) that are not presently captured by the *Framework*. It should be noted that agricultural intensification of production via fertilization is a possible response to increased demand for biomass for energy. If onsite N_2O emissions are not accounted for, the carbon footprint of agricultural feedstocks could be significantly underestimated.

Assessment of Monitoring and Estimation Approaches: The *Framework* lacks a scientific assessment of different monitoring/estimation approaches and their uncertainty. This is a critical omission as it is essential to have a good understanding of the technical basis and uncertainty underlying the use of existing data, models and look-up tables. A review of monitoring and verification for carbon emissions from different countries, both from fossil and biogenic sources, was recently released by the National Research Council (National Research Council 2010). This review may provide some guidance.

3.5. Case Studies

Charge Question 5: EPA presents a series of case studies in the Appendix of the report to demonstrate how the accounting framework addresses a diverse set of circumstances in which stationary sources emit biogenic CO_2 emissions. Three charge questions are proposed by EPA.

Overall Comments

In general, case studies are extremely valuable for informing the reader with examples of how the *Framework* would apply for specific cases. While they illustrate the manner in which a BAF is calculated, the data inputs are illustrative only and may or may not be the appropriate values for an actual biomass-to-energy project. Moreover, the case studies are simplistic relative to the manner in which biomass is converted to energy in the real world. For all case studies in the *Framework*, additional definition of the context is needed, along with examples of how the data are collected or measured, and a discussion of the impacts of data uncertainty. Overall, the case studies did not fully cover the relevant variation in feedstocks, facilities, regions, etc. of potential BAFs that is required to evaluate the methodology. For clarity, it might be useful to start with a specific forestry or agricultural feedstock example as the base case, then add the impacts of the more detailed cases, e.g., additional losses, products, land use changes.

Question 5(a). Does the SAB consider these case studies to be appropriate and realistic?

The case studies did not incorporate “real-world” scenarios which would have served as models for other situations that may involve biogenic carbon emissions. More would have been learned about the proposed *Framework* by testing it in multiple, unique case studies with more realistic data development and inclusion. Additional case studies for landfills and waste combustion, switchgrass, waste, and other regions would be useful, as well as illustrations of the implementation of feedstock groups, and *Framework* alternatives.

For example, Case Study 4 considers a scenario where corn stover is used for generating electricity. While it is possible that this scenario could be implemented, this particular case study is not realistic because very few electrical generation facilities would combust corn stover or agricultural crop residues only. A more likely scenario might be supplementing a co-firing facility with a low percentage of corn stover. Additionally, the assumption of uniform corn stover yields across the region is not realistic. Variation should be expected in the yield of corn stover across the region.

In another example, Case Study 5 calculates the net biogenic emissions from converting agricultural land in row crops to poplar for electricity production. This case study is also not representative of “real world” agricultural conditions as switching from one energy crop to another is uncommon. The formula provided for estimating the standing stock of carbon in the aboveground biomass in the poplar system is not intuitive. The methods for determining biomass yield and measuring changes in soil carbon (which will depend on current use of the land) are not described.

Question 5(b). Does the EPA provide sufficient information to support how EPA has applied the accounting framework in each case?

There remained considerable uncertainty in many of the inputs. In addition, some sensitivity/uncertainty analysis would be useful. The results of this analysis may guide the EPA in further model development. For example, if the BAF is determined to be zero, or not statistically different from zero, in most case studies, then this could pave the way for a simpler framework. As discussed in Section 4 below, a simpler approach could be designed to develop default BAFs for categories of feedstocks based on how their management and use interacts with the carbon cycle.

Question 5(c). Are there alternative approaches or case studies that EPA should consider to illustrate more effectively how the framework is applied to stationary sources?

Additional case studies should be designed based on actual or proposed biomass to energy projects to capture realistic situations of biomass development, production and utilization. For example, Case Study 1 describes the construction of one new plant. What would happen if 10 new plants were to be proposed for a region? And how would the introduction of multiple facilities at the same time impact the accounting for each facility?

All terms/values used to determine the BAF need to be referenced to actual conditions throughout the growth/production/generation processes that would occur in each case study. This should include an indication of how these values would actually be implemented by one or more involved parties. Regional look-up tables could be valuable and EPA could learn a great deal by trying to develop look-up tables.

Additional case studies could be developed for perennial herbaceous energy crops, annual energy/biomass sorghums, rotations with food and energy crops, cropping systems on different land and soil types, municipal solid waste and internal reuse of process materials. Each of these feedstocks should be assessed across alternative regions so that the variation in carbon changes across regions could be gauged.

For example it would be very useful to consider the application of the *Framework* to a cellulosic ethanol plant fueled with coal or gas, and consider the emissions of CO_2 from fermentation (not combustion) and the production of ethanol which is rapidly combusted to CO_2 in a non-stationary engine. While such an operation is associated with three major sources of CO_2 emissions (listed here), only one is included in the *Framework*; only two may be considered under EPA's regulatory authority, yet all three are emissions to the atmosphere. It would be useful for EPA to at least describe the emissions that are excluded from consideration so that biogenic carbon emissions from stationary sources can be viewed in context.

At least two case studies are needed on municipal solid waste. One case study should be on waste combustion with electrical energy recovery. EPA should also perform a case study on landfill disposal of municipal solid waste. Here it is important to recognize that landfills are repositories of biogenic organic carbon in the form of lignocellulosic substrates (e.g., paper made from mechanical pulp, yard waste, food waste). There is literature to document carbon storage and the EPA has recognized carbon storage in previous greenhouse gas assessments of municipal solid waste management.

In Case Study 3 the data used in Table 3 to describe the ‘paper co-product’ will vary with the grade of paper. The ‘carbon content of product’ may vary between 30 to 50% depending on the grade and the amount of fillers and additives. Also, some significant carbon streams in a mill can go to landfills and waste water treatment. The submitted comments from the National Council for Air and Stream Improvement (NCASI) include a useful example of the detail/clarity that could be used to enhance the value of the Case Studies.

After completion of the case studies, a formal evaluation would be useful to gauge the ease with which data were developed and the model implemented, whether the results are robust and useful in recognition of the uncertainty in the various input parameters, and whether the model results lead to unintended consequences.

Case studies could be developed to assess and develop a list of feedstocks or applications that could be excluded from accounting requirements as “anyway” emissions. A sensitivity analysis using case studies could be used to develop reasonable offset adjustment factors if they are needed to adjust “anyway” feedstocks for impact on long term stocks like soil if needed.

3.6. Overall Evaluation

Charge Question 6: Overall, this report is the outcome of EPA’s analysis of the science and technical issues associated with accounting for biogenic CO_2 emissions from stationary sources.

Question 6(a). Does the report in total contribute usefully to advancement of understanding of accounting for biogenic CO_2 emissions from stationary sources?

Yes, the *Framework* is a step forward in advancing our understanding how to account for biogenic emissions. It addresses many issues that arise in such an accounting system and it is thoughtful and far reaching in the questions it tackles. Its main contribution is to force important questions and offer some ways to deal with these. It covers many of the complicated issues associated with the accounting of biogenic CO_2 emissions from stationary sources and acknowledges that its choices will have implications for the estimates of CO_2 emissions obtained. These include those raised by SAB and discussed above, related to the choice of baseline, region selection and the averaging of emissions/stocks over space and time. However, the solutions offered in many cases, particularly those related to the use of harvested wood for bioenergy, lack transparency or a scientific justification.

Question 6(b). Does it provide a mechanism for stationary sources to adjust their total onsite emissions on the basis of the carbon cycle?

Clearly the *Framework* offers a mechanism to adjust total on-site emissions. For short accumulation feedstocks (i.e., agricultural residues, perennial herbaceous crops, mill wood wastes, other wastes), the *Framework* could, with some modifications and careful consideration of data and implementation, accurately represent the direct carbon changes offsite. Leakage, however, both positive and negative, remains a troublesome matter if left unresolved. Moreover, the *Framework* offers no scientifically sound way to define a region. The definition of the regional scale can make a large difference to the estimate of emissions from a facility using wood as a biomass. Moreover, if there is no connection between actions of the point source and what happens in the region, there is no foundation for using regional changes in carbon stocks to assign a BAF to the source.

The *Framework* also does not make a clear scientific case for use of waste or what is called "anyway" emissions. Scientifically speaking, all biogenic emissions are "anyway" emissions. Even most woody biomass harvested from old growth forests, would, if left undisturbed, eventually die and decompose, returning carbon to the atmosphere. The appropriate distinction is not whether the product is waste or will eventually end up in the atmosphere anyway, but whether the stationary source is leading to an increase or a decrease in biogenic carbon stocks and associated change in Global Warming Potential (GWP). To do this, the *Framework* must consider an anticipated baseline and the time period for "anyway" emissions and that this may vary across different types of waste feedstocks.

An important limitation of the proposed *Framework* is that the accounting system replaces space for time and applies responsibility for things that happen on the land to a point source, for which the agent who owns that point source has no direct control. Rather than comparing a "with" and "without" bioenergy scenarios over time, the *Framework* is based on spatial regions The proposed approach, which attempts to estimate facility-feedstock specific BAFs, would estimate an individual point source's BAF based on average data in a region in which it is located. Any biogenic carbon accounting system that attempts to create responsibility or give credit at a point source for carbon changes upstream or downstream from the point source must relate those responsibilities and credits to actions under control of the point source. However, the *Framework* does not clearly specify a cause and effect relationship between a facility and the biogenic CO_2 emissions attributed to it. In particular, if the BAF is assigned to a plant when it is approved for construction, as the BAF is currently designed, those emissions related to land use change will have nothing to do with the actual effect of the point source on land use emissions because the data on which it is based would predate the operation of the plant.

The dynamics of carbon accumulation in vegetation and soils and carbon and methane release through decomposition present a challenge for any accounting system because anticipated future changes in vegetation should, in principle, be factored into BAF. These future changes depend on natural processes such as fires and pest outbreaks that are not easily foreseen, and because of climate change and broader environmental change, we face a system that is hard to predict. Projecting forward based on current or historical patterns is subject to biases of unknown direction and magnitude. More importantly, land use decisions are under the control of landowners, who will be responding to unknown future events. The *Framework* recognizes this issue and chooses to use a Reference Point Baseline, the serious limitations of which have been discussed previously.

Overall, the EPA's regulatory boundaries, and hence the *Framework*, are in conflict with a more comprehensive carbon accounting that considers the entire carbon cycle and the possibility of gains from trade between sources, among sources or between sources and sinks to offset fossil fuel combustion emissions. Scientifically, a comprehensive greenhouse gas accounting would extend downstream – to emissions from by-products, co-products or products such as ethanol combustion or ethanol by-products such as distillers dried grains that are sold as livestock feed that ultimately becomes CO_2 (or CH_4). However, such a comprehensive accounting would require consideration of consistency with fossil fuel emissions accounting and emissions currently regulated (such as by EPA with vehicle greenhouse emissions standards). As for gains from trade, by restricting its attention to the regulation of point source emissions, EPA's analysis does not allow for the possibility that a fossil CO_2 emitter could contract with land owners to offset their emissions through forest protection and regrowth or carbon accumulation in soils. Bioenergy would still need to confront the issue of crediting offset carbon accumulation however. By staying within boundaries drawn narrowly around the stationary source, the *Framework* eclipses a more comprehensive approach to greenhouse gas reductions that would address all sources and sinks and take advantage of gains from trade.

Question 6(c). Does the SAB have any advice regarding potential revisions that might enhance the final document?

Overall, the *Framework* would be enhanced by including a description of its regulatory context and specifying the boundaries for regulating upstream and downstream emissions while implementing the regulation. The motivation for the *Framework* should be explained as it relates to Clean Air Act requirements and any recent court rulings. The *Framework* should also make explicit the constraints within which greenhouse gases can be regulated under the Clean Air Act. In doing this, the EPA could be clear that these issues have not been settled but that some assumptions were necessary to make a decision about the *Framework*. The EPA could also stipulate that further development of a regulatory structure might require changes to the accounting system. While the SAB understands the EPA's interest in describing an accounting system as a first step and potentially independent of the regulatory structure, the reader needs this background in order to understand the boundaries and context for the accounting structure and to evaluate the scientific integrity of the approach.

Similarly, the *Framework* is mostly silent on how possible regulatory measures under the Clean Air Act may relate to other policies that affect land use changes or the combustion/oxidation of products from the point sources that will release carbon or other greenhouse gases. For example if a regulatory or incentive system exists to provide credits for carbon offsets through land use management then under some conditions it would be appropriate to assign a BAF of 1 to biogenic emissions given that the carbon consequences were addressed through other policies.

The *Framework* does not make explicit how it does or does not address emissions downstream from a point source such as in the case of a biofuels or paper production facility where the product (biofuels, paper) may lead to CO_2 emissions when the biofuels are combusted or the paper disposed of and possibly incinerated. For example, if paper products are incinerated the incinerator may well be a point source that comes under Clean Air Act regulation. However, biofuels used in vehicles would not be subject to regulation as a point source. Though biofuel combustion emissions are already regulated, along with combustion of gasoline, via EPA's vehicle greenhouse gas emissions standards, the EPA needs to make clear the implicit assumptions on how biogenic carbon will be treated upstream and downstream from the point source if this *Framework* is used to regulate CO_2 emissions under the constraints imposed by the Clean Air Act for regulating stationary sources.

The *Framework* is lacking in implementation details. Implementation is crucial and some of the EPA's current proposals will be difficult to implement. Data availability and quality, as well as procedural details (e.g., application process, calculation frequency) are important considerations for assessing the feasibility of implementation and scientific accuracy of results. Implementation details (e.g., data, technical processes, administrative procedures, timing) need to be laid out, discussed and justified. Among other things, the discussion should note alternatives, uncertainty and implications via case studies.

Recommendations for Revising BAF

In response to the charge to the SAB, recommendations are offered here for revising the *Framework*. In the next section, the SAB suggests an alternative – default BAFs. If EPA decides to revise the *Framework*, the following recommendations for specific improvements to the document (and methodology) are summarized here. Many of the issues raised in previous responses regarding the treatment of specific factors included in the *Framework* are specific to particular feedstocks. The clarity of the *Framework* would be improved by differentiating among feedstocks based on how their management and use interacts with the carbon cycle. A BAF equation could be developed for each of these categories of feedstocks.

If EPA decides to revise the *Framework*, the following recommendations for specific improvements are summarized below.

- Develop a separate BAF equation for each feedstock category as broadly categorized by type, region, prior land use and current management practices. Feedstocks could be categorized into short rotation dedicated energy crops, crop residues, forest residues, perennial crops, municipal solid waste, long rotation trees and waste materials including wood mill residue and pulping liquor. They could be differentiated based on different prior land uses and different management practices.
    - For long-accumulation feedstocks like woody biomass, use an anticipated baseline and landscape approach to compare emissions from increased biomass harvesting against a baseline without increased biomass demand. For long rotation woody biomass, sophisticated modeling is needed to capture the complex interaction between electricity generating facilities and forest markets, in particular, market driven shifts in planting, management and harvests, induced displacement of existing uses of biomass, land use changes, including interactions between agriculture and forests and the relative contribution of different feedstock source categories (logging residuals, pulpwood or roundwood harvest).
    - For residues, consider incorporating information about decay after an appropriate analysis in which storage of ecosystem carbon is calculated based on decay functions.
    - For materials diverted from the waste stream, consider their alternate fate, whether they might decompose over a long period of time, whether they would be deposited in anaerobic landfills, and whether they are diverted from recycling and reuse, etc. Implementation complexity, cost and scientific accuracy should be considered. For feedstocks that are found to have relatively minor impacts, the EPA may need to weigh ease of implementation against scientific accuracy. After calculating decay rates and considering alternate fates, EPA may wish to declare certain categories of feedstocks with relatively low impacts as having a very low BAF or setting it to 0.
- Incorporate various time scales and consider the tradeoffs in choosing between different time scales.
- For all feedstocks, consider information about carbon leakage to determine its directionality as well as leakage into other media.

4. DEFAULT BAFs BASED ON FEEDSTOCK CATEGORIES

There are no easy answers to accounting for the greenhouse gas implications of bioenergy. Given the uncertainties, technical difficulties and implementation challenges associated with implementing the facility-specific BAF approach embodied in the *Framework*, the SAB encourages the EPA to “think outside the box” and look at alternatives to the *Framework* and its implementation as proposed. One promising alternative is default BAFs for each feedstock category. Given the daunting technical challenges of the *Framework*, and the prospective difficulties with implementation, the SAB recommends consideration of default BAFs by feedstock type, region, land management and prior land use. Under EPA’s *Framework*, facilities would use individual BAFs designed to capture the incremental carbon cycle and net emissions effects of their use of a biogenic feedstock. With default BAFs, facilities would use a weighted combination of default BAFs relevant to their feedstock consumption and location.

The defaults BAFs would rely on readily available data and reflect landscape and aggregate demand effects, including previous land use. The defaults would also have administrative advantages in that they would be easier to implement and update. Default BAFs for each category of feedstocks would differentiate among feedstocks using general information on their role in the carbon cycle. An anticipated baseline would allow for consideration of prior land use, management, alternate fate (what would happen to the feedstock if not combusted for energy) and regional differences. Default BAFs might vary by region, prior land use and current land management practices due to differences these might cause in the interaction between feedstock production and the carbon cycle. They would be applied by stationary facilities to determine their quantity of biogenic emissions that would be subject to the agency’s Tailoring Rule. Case studies should be used to evaluate the applicability of default BAFs to heterogeneous facilities. Facilities could also be given the option of demonstrating a lower BAF for the feedstock they are using. This would be facilitated by making the BAF calculation transparent and based on data readily available to facilities. Default BAFs should be carefully designed to provide incentives to facilities to choose feedstocks with the lower greenhouse gas impacts.

The SAB also explored certification systems as a possible way to obviate the need to quantify a specific net change in greenhouse gases associated with a particular stationary facility. Carbon accounting registries have been developed to account for and certify CO_2 emissions reductions and sequestration from changes in forest management. Ultimately, however, the SAB concluded that it could not recommend certification without further evaluation. Moreover, such systems could encounter many of the same data, scientific and implementation problems that bedevil the *Framework*.

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APPENDIX A: Charge to the Panel

<u>MEMORANDUM</u>

To: Holly Stallworth, DFO
Science Advisory Board Staff Office

From: Paul Gunning, Acting Director
Climate Change Division

Subject: Accounting Framework for Biogenic Carbon Dioxide (CO_2) Emissions from Stationary Sources and Charge Questions for SAB peer review

The purpose of this memorandum is to transmit the draft *Accounting Framework for Biogenic CO_2 Emissions* study and the charge questions for consideration by the Science Advisory Board (SAB) during your upcoming peer review in fall 2011.

In January 2011, the U.S. Environmental Protection Agency (EPA) announced a series of steps it would take to address biogenic CO_2 emissions from stationary sources. In addition to specific regulatory action, EPA committed to conduct a detailed examination of the science and technical issues related to accounting for biogenic CO_2 emissions and to develop an accounting framework for those emissions. The study transmitted today is that examination.

The study identifies key scientific and technical factors that should be considered when constructing any framework for accounting for the impact of utilizing biologically-based feedstocks at stationary sources. It then provides EPA's recommendations on those issues and presents a framework for "adjusting" estimates of onsite biogenic CO_2 emissions (i.e., a "biogenic accounting factor" or BAF) on the basis of information about the carbon cycle.

As indicated in the accompanying materials, advice on these issues will be important as EPA moves through the steps to address biogenic CO_2 emissions from stationary sources. We look forward to the SAB's review.

Please contact me if you have any questions about the attached study and charge.

Charge Questions

EPA is providing this study, *Accounting Framework for Biogenic CO_2 Emissions from Stationary Sources* (September 15, 2011), to the Science Advisory Board (SAB) to review EPA's approach on accounting for biogenic CO_2 emissions from stationary sources, including the scientific basis and methodological components necessary to complete that accounting.

Objective

EPA is charging the SAB to review and comment on (1) EPA's characterization of the science and technical issues relevant to accounting for biogenic CO_2 emissions from stationary sources; (2) EPA's framework, overall approach, and methodological choices for accounting for these emissions; and (3) options for improving upon the framework for accounting for biogenic CO_2 emissions.

This charge does not ask the SAB for regulatory recommendations or legal interpretation of the Clean Air Act statutes related to stationary sources.

Charge Questions

1. *Evaluation of the science of biogenic CO_2 emissions*

In reviewing the scientific literature on biogenic CO_2 emissions, EPA assessed the underlying science of the carbon cycle, characterized fossil and biogenic carbon reservoirs, and discussed the implications for biogenic CO_2 accounting. Does the SAB support EPA's assessment and characterization of the underlying science and the implications for biogenic CO_2 accounting?

2. *Evaluation of biogenic CO_2 accounting approaches*

In this report, EPA considered existing accounting approaches in terms of their ability to reflect the underlying science of the carbon cycle and also evaluated these approaches on whether or not they could be readily and rigorously applied in a stationary source context in which onsite emissions are the primary focus. On the basis of these considerations, EPA concluded that a new accounting framework is needed for stationary sources.

- 2(a). Does the SAB agree with EPA's concerns about applying the IPCC national approach to biogenic CO2 emissions at individual stationary sources?
- 2(b). Does the SAB support the conclusion that the categorical approaches (inclusion and exclusion) are inappropriate for this purpose, based on the characteristics of the carbon cycle?
- 2(c). Does the SAB support EPA's conclusion that a new framework is needed for situations in which only onsite emissions are considered for non-biologically-based (i.e., fossil) feedstocks?
- 2(d). Are there additional accounting approaches that could be applied in the context of biogenic CO2 emissions from stationary sources that should have been evaluated but were not?

3. *Evaluation of methodological issues*

EPA identified and evaluated a series of factors in addition to direct biogenic CO_2 emissions from a stationary source that may influence the changes in carbon stocks that occur offsite, beyond the stationary source (e.g., changes in carbon stocks, emissions due to land-use and land management change, temporal and spatial scales, feedstock categorization) that are related to the carbon cycle and should be considered when developing a framework to adjust total onsite emissions from a stationary source.

- 3(a). Does SAB support EPA's conclusions on how these factors should be included in accounting for biogenic CO2 emissions, taking into consideration recent advances and studies relevant to biogenic CO2 accounting?
- 3(b). Does SAB support EPA's distinction between policy and technical considerations concerning the treatment of specific factors in an accounting approach?
- 3(c). Are there additional factors that EPA should include in its assessment? If so, please specify those factors.
- 3(d). Should any factors be modified or eliminated?

4. *Evaluation of accounting framework*

EPA's accounting framework is intended to be broadly applicable to situations in which there is a need to represent the changes in carbon stocks that occur offsite, beyond the stationary source, or in other words, to develop a "biogenic accounting factor" (BAF) for biogenic CO_2 emissions from stationary sources.

- 4(a). Does the framework accurately represent the changes in carbon stocks that occur offsite, beyond the stationary source (i.e., the BAF)?
- 4(b). Is it scientifically rigorous?
- 4(c). Does it utilize existing data sources?
- 4(d). Is it easily updated as new data become available?
- 4(e). Is it simple to implement and understand?
- 4(f). Can the SAB recommend improvements to the framework to address the issue of attribution of changes in land-based carbon stocks?
- 4(g). Are there additional limitations of the accounting framework itself that should be considered?

5. *Evaluation of and recommendations on case studies*

EPA presents a series of case studies in the Appendix to demonstrate how the accounting framework addresses a diverse set of circumstances in which stationary sources emit biogenic CO_2 emissions.

- 5(a). Does the SAB consider these case studies to be appropriate and realistic?
- 5(b). Does the EPA provide sufficient information to support how EPA has applied the accounting framework in each case?
- 5(c). Are there alternative approaches or case studies that EPA should consider to illustrate more effectively how the framework is applied to stationary sources?

6. *Overall evaluation*

Overall, this report is the outcome of EPA's analysis of the science and technical issues associated with accounting for biogenic CO_2 emissions from stationary sources.

6(a). Does the report – in total – contribute usefully to the advancement of understanding on accounting for biogenic CO2 emissions from stationary source?

6(b). Does it provide a mechanism for stationary sources to adjust their total onsite emissions on the basis of the carbon cycle?

6(c). Does the SAB have advice regarding potential revisions to this draft study that might enhance the utility of the final document?

APPENDIX B: Temporal Changes in Stand Level Biogenic Emissions Versus Fossil Emissions

Cherubini et al. (2011) analyzes temperature increases on the basis of GWP (global warming potential) whereas Cherubini et al. (2012) analyzes climate impacts using GTP (global temperature potential). GWP is the time integral of the change in radiative forcing from a pulse emission of CO_2 (in this case, from harvested biomass) and subsequent sequestration by biomass growth, whereas GTP is the integral of actual temperature response to a pulse emission of CO_2 and subsequent sequestration by biomass growth. Both studies use a simple contrived comparison of biogenic emissions from a single stand over hundreds of years to comparable fossil emissions. Much is assumed regarding for instance global activity and emissions, and climate and carbon cycle dynamics. Also, importantly, landscape responses and investment behavior are not reflected which represent concurrent and related emissions and sequestration that affect net global emissions changes.

Both studies incorporate a suite of carbon uptake mechanisms (such as oceanic uptake) in addition to regrowth in forest stands. In this context, the GTPbio, discussed by Cherubini (2012), is a more accurate metric for the actual climate response. The idea of the GTPbio is simple: it represents the increase in global average temperature over a given period due to a transient increase in carbon dioxide in the atmosphere (between the initial biomass combustion or respiration and the ultimate regrowth of the carbon stock) relative to the temperature response to a release of an equivalent amount of fossil CO_2 at time 0 (expressed as a fraction between 0 and 1). To calculate a GTPbio value, a time scale must be specified. The calculation for GTPbio is the ratio of the average temperature increase with biogenic emissions followed by reabsorbtion by biomass regrowth over, say, 100 years divided by the average temperature increase from the initial emission alone over 100 years. For short accumulation feedstocks, such as perennial grasses, GTPbio would be a very small fraction due to fast carbon accumulation times (ignoring leakage effects). For feedstocks with long accumulation times, one must compute the change in global temperature over time, accounting for the decline in temperature change as carbon is reabsorbed.

Cherubini et al. (2011, 2012) provide an artificial simplified example for a single forest stand. The same type of metric could be used to compare temperature changes or changes in radiative forcing associated with increased biomass energy use for one year or more for a landscape or nation – taking into account the land carbon change over time associated with increased biomass energy use. This would involve comparison of a business as usual case to an increased biomass use case. A simpler metric that compares the cumulative radiative forcing of biogenic feedstocks to the cumulative radiative forcing of fossil fuels over time could also be used, e.g.. Cherubini's GWPbio. However the broader literature should be considered regarding the climate implications of alternative emissions pathways (see charge question 1 response) while considering uncertainty in global emissions, climate response and the carbon cycle.

Figure B-1 demonstrates the importance of the time horizon or, more specifically, the weight to place on temperature increases that occur in the short term versus temperature increases that occur later. Consider a scenario in which biomass is harvested, but the carbon stock is replaced within a 100 year time scale. The GTPbio for a 100-year regrowth and a 100 year time horizon is roughly 0.5, meaning that the time-integrated global average temperature increase within that 100 year period is 50% of the temperature increase caused by an equivalent amount of fossil carbon (or straight CO_2 release without regrowth of biomass). However, using the average temperature increase for the biogenic case over 100 years masks the fact that although there will be an initial increase in temperature near the beginning of the 100 year period the reabsorption of carbon in the forest will bring the effect on ground temperature to nearly zero by year 100, giving an average temperature that was 50% of the average fossil temperature increase over 100 years. In fact the instantaneous temperature change for the biogenic case falls below zero slightly before 100 years because oceans initial absorb extra CO_2 in response to the initial biogenic emission (see Figure B-1, adapted from Cherubini 2012, Figure 5a). The temperature effect equilibrates to zero as the ocean CO_2 is balanced. A more precise picture of intertemporal effects is shown in Figure B-1, adapted from Cherubini et al. (2012).

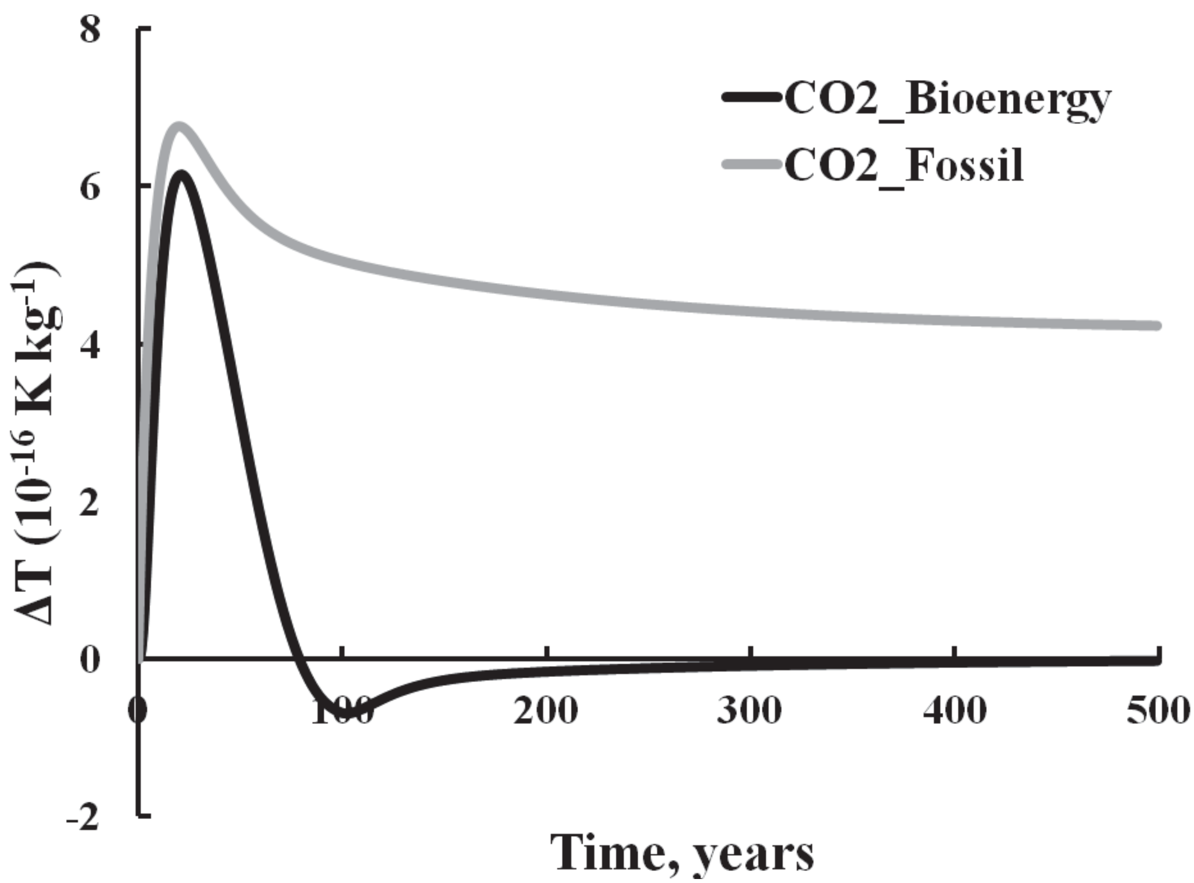


Figure B-1: Surface temperature change from biogenic emissions versus fossil fuel over time. Adapted from Cherubini et al. (2012) and reprinted with copyright permission.

Cherubini et al. (2012) have shown that if biomass is harvested and the carbon is reabsorbed within a 100 year time scale, the global average temperature increase over that 100 year period is 50% of the temperature increase caused by an equivalent amount of fossil carbon. We might conclude that biogenic emissions are roughly 50% as damaging as fossil fuels, however the high point of temperature increase created by biogenic emissions occurs early in the 100 year cycle and is back to zero by the time the carbon is reabsorbed. For the case where carbon is recovered within 100 years Cherubini et al. (2012) have shown that at 20 years, the average temperature increase (over 20 years) from biogenic fuel is 97% of the temperature increase caused by an equivalent amount of fossil carbon; for years 21 to 100 years, the average increased is 0.37 and for years 101 to 500, the increase is 0.02.

A current practice for international reporting under IPCC guidelines and international treaty negotiations is to use greenhouse gas emissions and sink values that represent the cumulative radiative forcing for greenhouse gases over a 100 year period with uniform weighting over 100 years. Greenhouse gas values are reported in tons CO_2 equivalent where one ton of CO_2 equivalent is an index for the cumulative radiative forcing for a pulse emission of one ton of CO_2 over 100 years. The CO_2 equivalent for a ton of other greenhouse gases is given by how many times more radiative forcing it produces over 100 years compared to CO_2 (e.g., 21 times for CH_4) (EPA 2012).

APPENDIX C: Fate of Landscape Residue after Harvest and System Storage of Carbon

The decomposition of materials left after harvest can be estimated from the negative exponential decay equation (Olson 1963): $C_t=C_0 \exp[-kt]$ where C_t=is the amount at any time t, C_0 is the initial amount, k is the rate-constant of loss, and t is time. Solving this function for a range of rate-loss constants results in the relationship shown in Figure C-1 for a range of k that covers the most likely range for decomposition rates of leafy to woody material in North America. In no case does the store instantaneously drop to zero as assumed in the *Framework*.

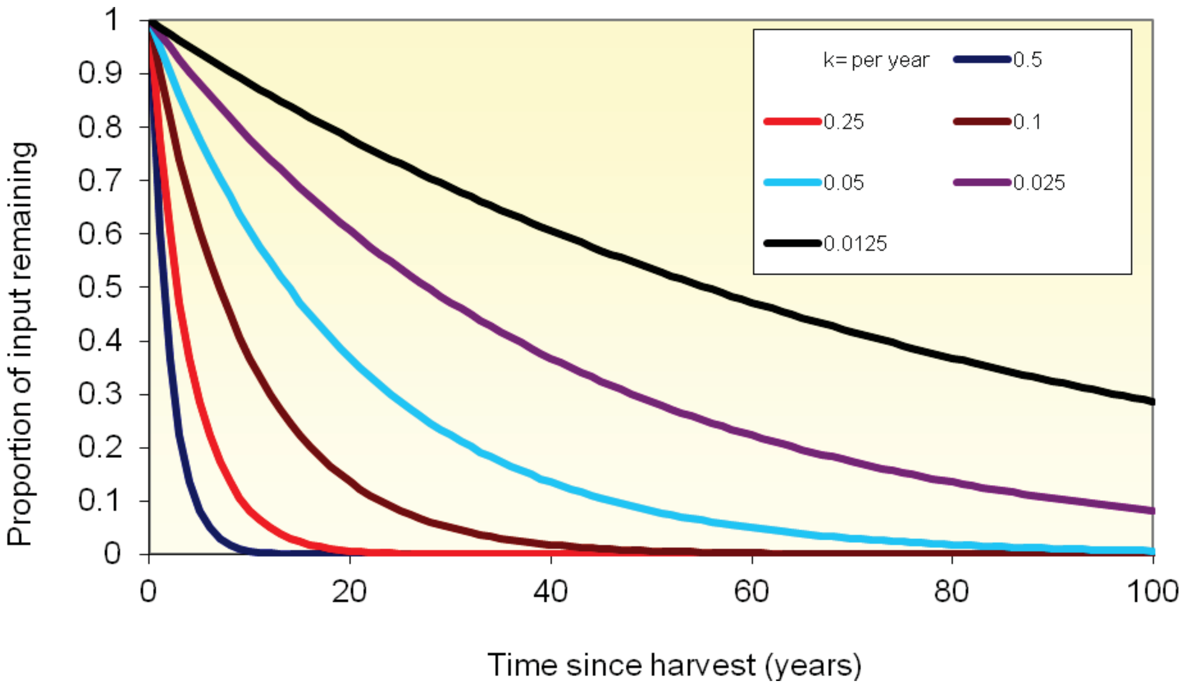


Figure C-1: Fate of residue/slash left after harvest as function of k and time since harvest.

The amount of carbon stored on average in a forest system or fuel-shed comprised of units or stands that generate equal amounts of residue or slash is given by: I/k, where I is the average forest input of residue or slash. To create a relative function independent of the amount of residue or slash created, the input of each harvest unit or stand can be set to either 1 (to give the proportion of the input) or 100 (to give a percent of the input). The average forest input (I) would therefore be equal to $1/R_H$ or $100/ R_H$ where R_H is the harvest return interval. Using this relationship to solve the average store relative to the input is presented in Figure C-2 for the most likely range of decomposition rates for leafy to woody material in North America. This indicates that there are a wide range of possible cases in which the store of residue or slash can exceed the initial input (shown by the horizontal line indicating storage of 1). This means that combusting this material will cause the store to drop by the amount indicated, and this amounts to the net flux of carbon to the atmosphere. To a large degree there is a negative relationship between the harvest interval and k; materials with high values of k (i.e., leafy) are typically harvested with short intervals between harvests and material with low values of k (i.e., large wood) are typically harvested with long interval between harvests. This suggests that the effect of harvesting residues and slash is largely independent of the loss rate-constant.

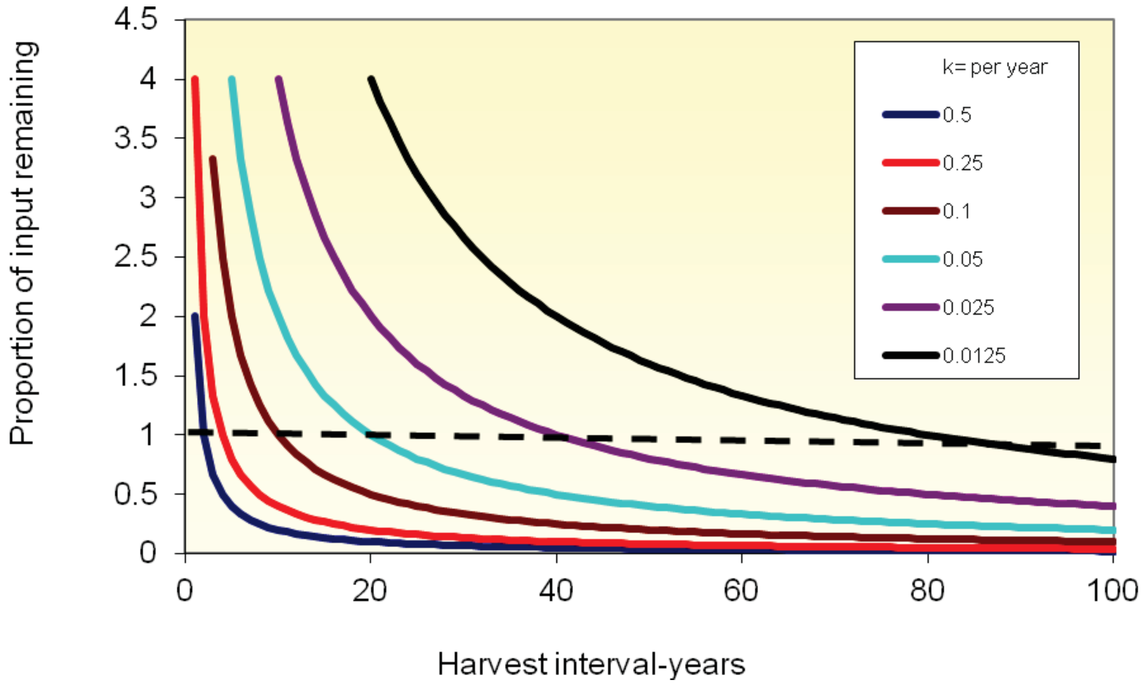


Figure C-2: Landscape average store of residue/slash as function of k and harvest interval.

APPENDIX D: Carbon Balances over Time in an Existing Forest System

To determine whether a forest harvest system for existing forest acreage creates a carbon debt, or alternatively, a gain it is appropriate to examine this problem at the landscape-level (or in the context of biogenic carbon a fuel-shed basis). Note the discussion that follows refers only to existing managed forests (and their stored carbon) and not broader landscape effects such as the expansion or contraction of forest area. At the forest system level there are three possible cases: (1) a relatively constant, steady-state store of carbon if the harvest system is continued unchanged, (2) an increase of carbon stores to a higher steady state if the intensity of harvest declines, and (3) a decrease of carbon stores to a higher steady-state if the intensity of harvest increases. These cases are illustrated in Figures 4-6 which are based on the online Forest Sector Carbon Calculator used in the forest system landscape mode (http://landcarb.forestry.oregonstate.edu/default.aspx) .

In Figure D-1, a 50-year clear-cut harvest rotation was practiced until 2010 and then continued for 500 years. This resulted in no carbon debt. If tracked at the stand scale one would see carbon levels rising and falling, but over time the net balance is zero. In contrast, if one converted the 50-year clear-cut harvest rotation system to a 25-year clear-cut harvest rotation system as in Figure D-2 there would have been a decline in carbon stores in the ecosystem. This decline would be considered a carbon debt and while not permanent (i.e., forever), it would remain as long as the 25-year management system persists. If the 50-year clear-cut harvest rotation was replaced by a 100-year clear-cut system at year 2010, then there would have been a gain carbon stores (Figure D-3). That gain would remain as long as that 100-year clear-cut system of management was maintained. All these simulations all assumed that soil productivity is maintained regardless of harvest interval.

At the existing forest level (as opposed to the stand level), live, dead, and soil stores all acted the same. Each of these pools either remained in balance (i.e., no net gain) or could increase or decrease depending on how the interval of harvest changes. The steady-state store of all three pools is controlled by the I/k relationship developed by Olson (1963), where I is the input of carbon to the pools and k is the proportion lost from the system in respiration and harvest (the live also has a loss related to mortality of trees). As the harvest interval decreases the input to the pool (I) decreases and the proportion lost via harvest (k) increases. This explains why the ecosystem stores decrease when the harvest interval is shortened and why they increase when the harvest interval is increased. A similar response happens when one takes a larger share of the carbon stores away when there is a harvest.

These dynamics have several important implications that need to be considered in the context of biogenic carbon: (1) long-term carbon debts, gains, and balances are best examined at the forest system-level (not to mention the broader agriculture-forest landscape level), (2) all forest carbon pools can exhibit either debts, gains, or remain relatively constant, (3) most systems of forest management will reach a steady-state if maintained over a long enough period and this steady-state can be maintained as long as the management system is continued, and (4) ultimately reaching a steady-state does not determine if there has been a loss or gain in carbon as this depends on how harvest management changes from one steady-state to the next.

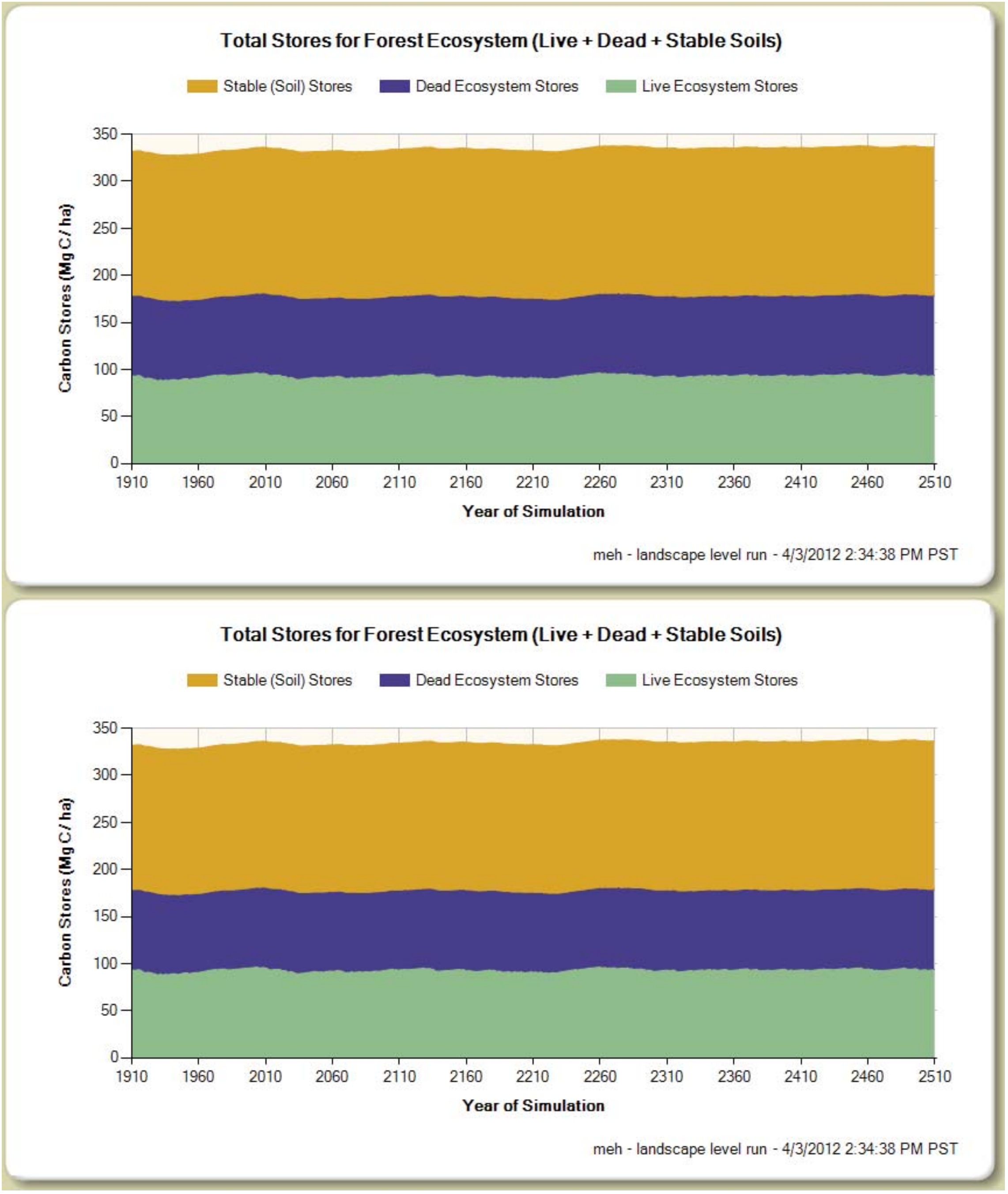
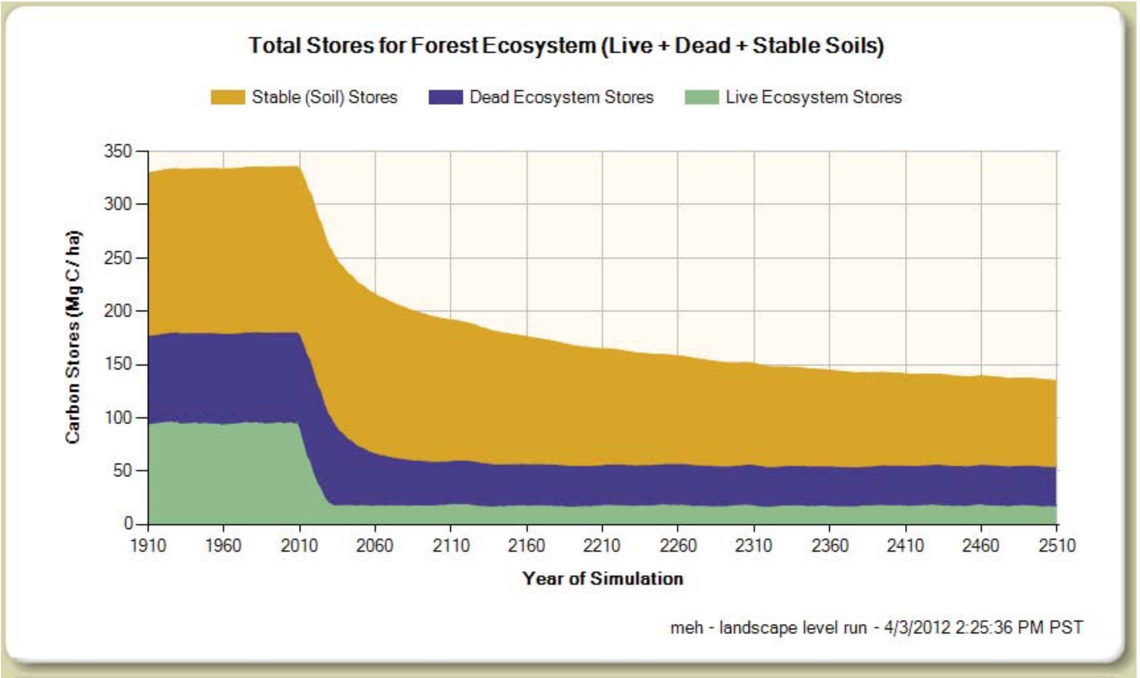


Figure D-1: Changes in carbon stores of major forest ecosystem pools when a 50 year clear-cut harvest system is established and continued. The result is a continued carbon balance.



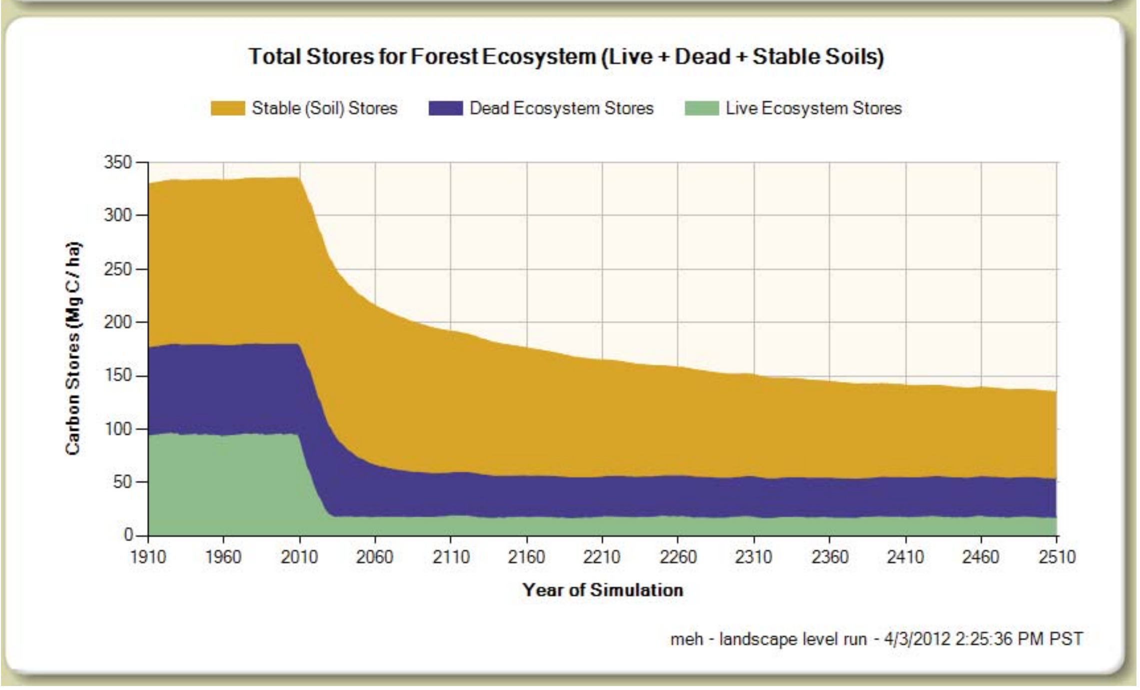


Figure D-2: Changes in carbon stores of major forest ecosystem pools when a 50 year clear-cut harvest system is replaced by a 25 year clear-cut harvest system in 2010. The result is a carbon debt.

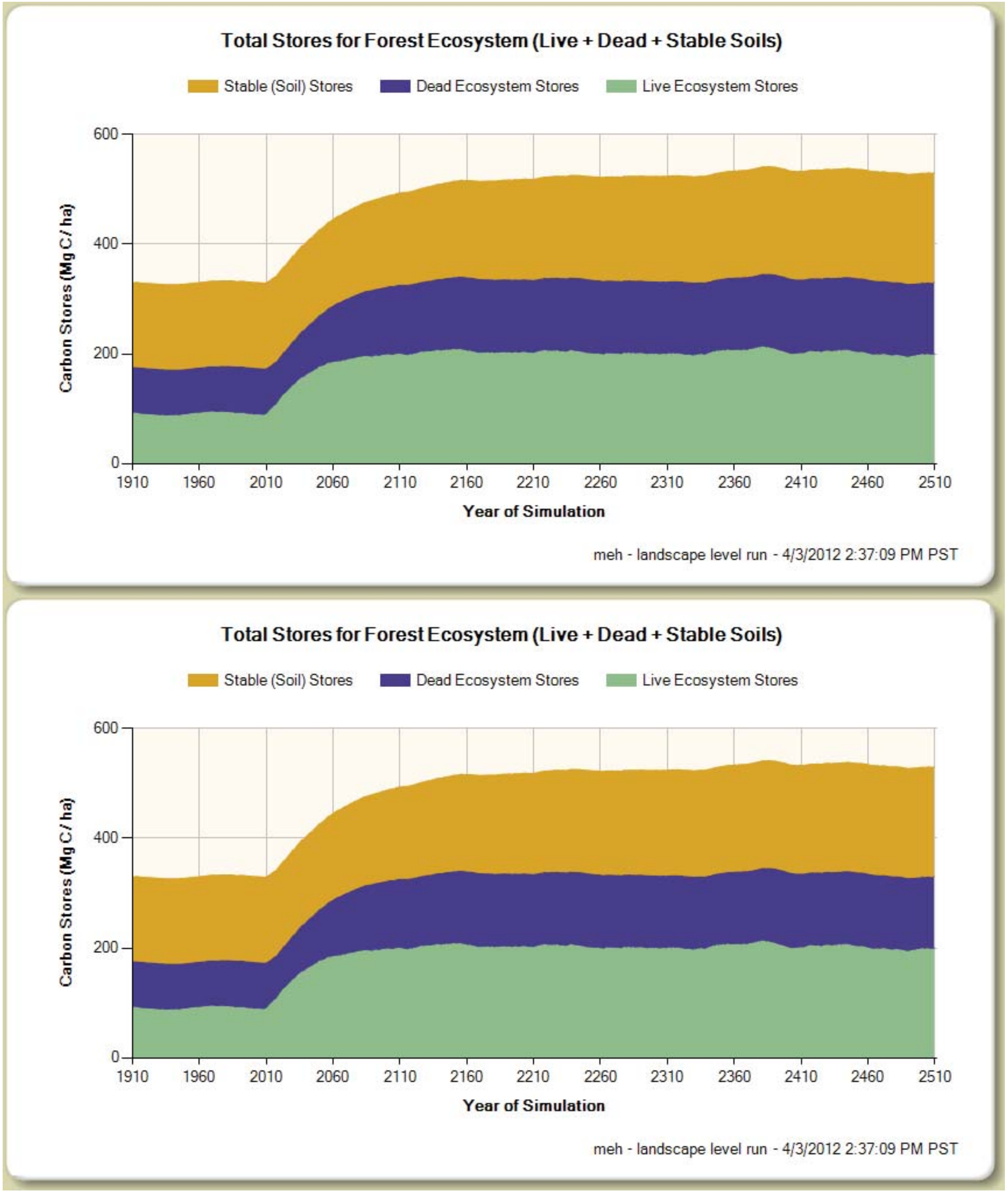


Figure D-3: Changes in carbon stores of major forest ecosystem pools when a 50 year clear-cut harvest system is replaced by a 100 year clear-cut harvest system in 2010. The result is a carbon gain.

APPENDIX E: Dissenting Opinion from Dr. Roger Sedjo

Introduction

EPA's Science Advisory Board (SAB) was asked to review and comment on the EPA's Accounting Framework for Biogenic CO2 Emissions from Stationary Sources (Framework September 2011). The motivation for the Accounting Framework "is whether and how to consider biogenic greenhouse gas emission in determining thresholds … for Clean Air Act permitting" (p. 4). To my knowledge the SAB Advisory has been completed and is being submitted to the broader SAB process. The comments below (and page numbers cited) relate to the SAB Advisory draft of 6-15-12 (SAB 2012).

I take fundamental issue with many of the elements of the SAB Report. Although I largely agree with the Advisory's criticisms of the absence of supporting science for many of the Framework's suggested approaches, I find unconvincing and unscientific much of the Advisory's attempt to salvage large elements Framework's approach. My comments focus largely, but not entirely, to forest issues in the Report not only because that is the area of my greatest expertise but also because the defects in the Framework approach are most egregious in forestry.

The EPA considered whether to categorically include biogenic emission in its greenhouse gas accounting or whether to categorically exclude biogenic emissions (p 6-7). The agency rejected both extremes and asked the SAB whether it supported their conclusion that categorical approaches are inappropriate for treatment of biogenic carbon emissions. However, I do not believe that this issue was properly vetted within the SAB process. Although the statement that "carbon neutrality cannot be assumed for all biomass energy a priori" (p 7) is correct, it misrepresents the serious position developed by the Intergovernmental Panel on Climate Change (IPCC 2006) and commonly used included a critical qualification regarding the condition of land cover generally and forest stock specifically. This requirement is missing from the simplistic evaluation statement. This position is supported in the Appendix to this piece, (USDA appendix by Hohenstein, 2012), which notes that the major IPCC rationale does not claim "a priori" neutrality. The IPCC, which suggested this approach, makes carbon neutrality contingent on an aggregate monitoring approach that focuses on the changes in aggregate land use and forests. Thus, the definitive development of the wide spread exclusion of biogenic and wood does not, in fact, involve an a priori assumption of neutrality. Rather it involves a qualification (for wood) that the forest stock be constant or expanding. I should note here that consideration of that important qualification was largely absence from the evaluation by the SAB and, in my judgment, aggressively discouraged by the organizers from the SAB discussion.

Finally, if the proposed Accounting Framework were capable of providing reliable accounting, one might give it serious consideration as an alternative to the IPCC approach in achieving the EPA objectives. However, as is acknowledged by the Advisory (e.g., p. 15), the proposed Accounting Framework is replete with problems as are the calculations of the elements necessary for calculating the Biological Accounting Factor (BAF). The acknowledged scientific weaknesses in the EPA document are identified throughout the SAB Advisory.

This paper demonstrates below that the SAB Advisory has not adequately addressed some of these issues and has not found ways to estimate in a scientifically acceptable way the values of some of the requisite components of the BAF.

Defects in the Accounting Framework

Questions raised in the Advisory about the Framework run from the appropriateness of the proposed use of the same accounting framework for the various feedstocks, which are different, to issues dealing with the appropriate baseline and questions concerning the relevant timescale. The SAB Advisory essentially embraces a variant of the BAF approach, which was developed in the Framework, even though the Advisory points to numerous important weaknesses of the BAF approach. The BAF is a simple accounting model that tries to identify and measure the various components and impacts of carbon emissions and accumulations from biomass energy sources. Ultimately, the Advisory essentially embraces the general BAF approach but applies it differently to individual biogenic feedstocks. However, the Advisory acknowledges throughout that a number of the components of the BAF cannot be adequately measured.

For example, the Advisory acknowledges that for important major elements of the Framework, e.g., leakage, there is no satisfactory monitoring or measurement system. Leakage, which can be either positive of negative, may involve the deflection of deforestation and associated emission out of woodshed under consideration or it may involve sequestration associated with offsetting forest management outside of that woodshed. Thus, the values of these major elements are essentially empirical, could be either positive or negative, but have their impacts outside of the area of direct observation. But, without accurate leakage values, the BAF approach proposed cannot accurately estimated for carbon changes. It cannot even determine the sign of the changes with any great accuracy. Thus, although the Advisory states that "it is important to have scientifically sound methods to account for greenhouse gas emission caused by human activities" (p 13), it acknowledges that the it is widely acknowledged in the literature that leakage cannot to be readily measured with any accuracy (Murray et al. 2004; Macauley et al. 2009). Nevertheless, in contradiction of this finding the Advisory suggests that "the agency … try to ascertain the directionality of net leakage … and incorporate that information into decision making." (p 9-10). This suggestion flies in the face of the concept of "scientifically sound methods."

Indeed, the application of the proposed framework would either need to leave these elements of the BAF empty, as suggested in the USDA letter posted on the SAB website, or nonscientific guesses would need to be imposed, as suggested in parts of the Advisory. In either case large errors in measurement appear almost inevitable and, rather than providing the regulators with accurate information, would provide misinformation to regulators and would likely redound to errors in the application of regulations. The idea introduced in the Advisory of default BAFs does not do anything to address their fundamental lack of scientific rigor.

Other thorny issues involve questions of the boundaries of a woodshed and/or a region, which relate to the leakage question, the intermixing of industrial wood and biomass so that significant portions of any harvest are used for each, and the export of biomass for energy, e.g., the large flow of wood pellets to Europe, where their emissions for the production of bioenergy will not be captured in the accounting. Finally, any accounting approach that tries to monitor each biomass using unit is surely going to be time consuming and expensive, perhaps too expensive to justify the use of the biomass for energy (Sedjo and Sohngen 2012).

An important defect is that the Advisory embraces a carbon-debt framework. However, this framework is an artifact of an arbitrary decision of how the accounting system is applied. If the forest is sustainability managed, then there is no carbon-debt. Withdrawals equal growth for both biomass and carbon. Accounting debts can occur in some circumstances, however. For a mature forest stand, if the accounting period begins with the harvest of the stand, as in the Manomet Study, a debt is incurred for that stand. Note that net carbon sequestration could be occurring in that forest but on different stands. Most forests are multi-aged and hence will have net growth occurring on some stands while stock reductions occur on other stands.

An additional source of confusion regarding carbon debt is related to the accounting period. If the accounting focuses on a stand and the accounting period begins with the harvest, a debt will be establishment for the forest stand. However, if the accounting begins with the forest establishment, e.g., at tree planting, then the initial post planting growth is building up a stock of carbon that will be released at harvest. Thus, any future debt from that stand will have been offset in advance of the harvest and no intertemporal net carbon debt is incurred.

Thus, although an accounting debt can be found for mature stands, the debt is an artifact of the time period selected and the choice of how narrowly to define the relevant forest stands. Furthermore, a carbon debt will not be occurred for sustainably managed forests. In the aggregate, the U.S. forest system is more than sustainable as demonstrated by the FIA's data going back to a least 1952. Thus, a fully accounting of the entire managed US forest does not find a carbon-debt.

In summary, the Advisory identifies a host of problems with the proposed Accounting Framework, and reports that "the SAB did not find the Framework to be scientifically rigorous" (p 30). Indeed, although the Framework is said to "include most of the elements that would be needed to gauge changes in CO2 emissions," the problems with the effective of monitoring, measurement and verification of several of the components are daunting.

Alternative Approaches for Accounting for Biogenic Carbon

One wonders why the SAB exerted so much effort to try to save the Accounting Framework, containing as it does, such fundamental defects. It is my understanding that the SAB was asked to review and comment on the Framework, but not necessarily to save it. Indeed, as noted above, EPA's change included the question of "whether … to consider biogenic greenhouse gas emission in determining thresholds … for Clean Air Act permitting" (p. 4).

Nevertheless, despite the identification of very serious defects in the approach, there is a considerable attempt in the SAB process to downplay the problems and ignore the lack of scientific bases for measuring some of the elements, apparently in order to preserve a variant of the approach, no matter how defective.

There are at least two basic ways that one might approach the problem of estimating the net emissions associated with biogenic energy. The highly regarded scientific organization, Intergovernmental Panel on Climate Change (IPCC) has suggested an aggregate approach that would focus on the changes in aggregate land use and forests to determine whether, for example, aggregate forest stocks are expanding or contracting. This approach has been supported by the USDA (Hohenstein 2012) in a response to an earlier draft Advisory by the SAB.

In the context of measuring the total aggregate forest the issue of leakage and anticipatory management within the US does not arise since to total system is evaluated. Where the aggregate is subdivided into a few large international regions, these issues are more easily captured since flows in forest biomass are measured in the international trade statistics and individual woodshed monitoring is not necessary. Indeed, for the US this approach can easily be put in place at low cost since the Forest Service has been undertaking Forest Inventory Assessments (FIA) for over fifty years.

The alternative to the IPCC approach, suggested by the Accounting Framework, involves the individual audit of each separate woodshed associated with a facility and an attempt to estimate the impact of each individual operation on net emissions. Such an approach would be a monitoring nightmare complicated by the fact that wood feedstock could, and likely would on occasion, be brought into one region from other small regions as required, this situation would involve leakage. Leakage could be replete since more regions would almost surely involve more leakage. Not only is the individual wood shed audit approach much more expensive, it also is inadequate since wood sheds are not always well defined and wood will undoubtedly flow across various woodsheds and leakage will occur. However, such detail is entirely unnecessary for purposes of the broad monitoring of biogenic facilities and their effects on atmospheric carbon. The relevant consideration is not the infinitesimal impact of each individual facility. Rather, the concern is with the grand aggregate impact of the bioenergy system on net emissions. If this approach does not properly account for the effects of leakage and anticipatory forest management (reverse leakage), the BAF estimates will have basic errors.

The Framework approach and the SAB Advisory appear to accept the notion that the Framework Accounting approach is superior to the IPCC approach. However, no evidence of this is provided either in argumentation or in analytical studies. Nevertheless, it is probably indisputable that the costs of the Accounting Framework approach with its estimated BAFs are far higher than those associated with the IPCC approach.

Five Summarizing Points

First, the guidelines provided by the EPA for the SAB Advisory essentially accept the Framework view and dismisses the IPCC suggested approach with regard to biogenic feedstocks within the land use sector, including forests. This was done despite that fact that there was no serious discussion by our SAB group of the adequacy or viability of the IPCC approach. Indeed the IPCC approach was dismissed by the EPA as inadequate on rather flimsy grounds. I note that my position is supported in the letter by William Hohenstein, Director of the Climate Change Program Office posted at the SAB website. The letter states that USDA "prefers the IPCC accounting framework" approach and takes issue with the rationale used by the SAB Advisory and its dismissal of the IPCC approached. USDA differs with the assertion of the SAB Advisory and maintains "the IPCC approach is not equivalent to an a priori assumption that these feedstocks are produced in a carbon neutral manner or an assertion that land use activities contributing feedstocks to the energy sector can be managed without consideration of atmospheric outcome."

Second, an attempt to assess the carbon debt of individual stands fundamentally misses the point since it is the entire forest, not individual stands that are relevant to the carbon footprint as seen by the atmosphere. As such, the attempt to imperfectly apply the BAF to individual forests is costly and irrelevant to the aggregate U.S. carbon footprint.

Third, although the Advisory acknowledges the dynamic nature of market driven supply systems that would be providing the biogenic energy feedstock, it essentially uses a static approach that largely ignores various market responses and adaptations to changing circumstances. Although the Advisory acknowledges that investment decisions for trees must predate their utilization by years and indeed decades, this reality is not incorporated into any BAF calculation. Indeed, while investment decisions must be driven by the anticipation of the existence and size of future markets, these considerations are acknowledged for wood biomass in parts of the Advisory and then disregarded in the application of the approach for regulatory purposes. Thus, the actual approach suggested is essentially static, missing the essential dynamic nature of the supply process. Despite these basic defects, the Advisory recommendations are treated as if they are scientifically sound.

Fourth, the Advisory erroneously states that incentives for producing replacement bioenergy crops are absence. Such a result would occur in viable markets only if there were no anticipation of increasing future demand. However, a variety of signals, including requirements of renewal portfolio standards and forecasts of dramatic biomass energy demand increases over the next couple of decades by various authoritative organizations, e.g., EIA.

Fifth, the Advisory tends to support a very expensive and onerous regulatory accounting system rather than a much more efficient system such as suggested by the IPCC. This support is given without any apparent serious assessment or rationale that the regulatory results of the BAF system will be equal to or superior to those that would result from a much less expensive and less onerous IPCC type approach.

In summary, I find that although the SAB Advisory provides a useful critique of the Accounting Framework and the BAF approach. However the Advisory falls into the trap of trying to make a basically defective system functional and tends to support many aspects of that flawed system. In the end the Advisory largely ignores its own criticisms and supports a fundamentally flawed approach. Thus, since the motivation for the Accounting Framework "is whether and how to consider biogenic greenhouse gas emission in determining thresholds … for Clean Air Act permitting" (p. 4), it can rationally be concluded that biogenic greenhouse gas emission are best not considered in determining thresholds or perhaps considered only of the forest and land use conditions as such that they do not meet minimal IPCC conditions.

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