The Greenhouse Gas Benefits of Substituting Wood for Other Construction Materials in New England

Photo: Mahlum Architects, courtesy W.G. Clark Construction

September 22, 2014

Prepared by Ann Gosline
for

New England Forestry Foundation
P.O. Box 1346
32 Foster Street
Littleton, MA 01460

New England Forestry Foundation

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I. New England Forestry Foundation's Project to Document the Potential of New England's Working Forest Lands

This report is part of a larger project to document both the existing value and potential of New England's working forest lands: Value – not only in terms of business opportunities, jobs and income – but also nonfinancial values, such as enhanced wildlife populations, recreation opportunities and a healthful environment. This project was undertaken by first developing a series of technical reports on the following topics:

- Assessing the Current Conservation Status of New England Forests
- Potential for Woody Biomass to Displace Fossil Fuels
- Benefits of Substituting Wood for Other Building Materials
- Forest Recreation
- The Potential for Enhancing Wildlife Habitat
- The Influence of New England's Working Forest on Air Quality
- The Influence of New England's Forest on Water Quality
- Consumption and Production of Wood in New England
- New England Nontimber Forest Products: Practices and Prospects
- The Potential of New England's Forests to Produce Wood
- The Potential of New England's Forest Products Industry

New England's forests have sustained the six-state region since colonial settlement. They have provided the wood for buildings, fuel to heat them, the fiber for papermaking, the lumber for ships, furniture, boxes and barrels and so much more. As Arizona is defined by its desert landscape and Iowa by its farms, New England is defined by its forests. These forests provide a wide range of products beyond timber including maple syrup, balsam fir tips for holiday decorations, paper birch bark for crafts, edibles such as berries, mushrooms and fiddleheads and curatives made from medicinal plants. They are the home to diverse and abundant wildlife. They are the backdrop for hunting, hiking, skiing and camping. They also provide important ecosystem services that we take for granted including clean air, potable water and carbon storage. In addition to tangible benefits that can be measured in board feet or miles of forest trails, forests and the natural world generally have been shown to be important to both physical and mental health.

Beyond their existing contributions, New England's forests have unrealized potential in each of these areas. Sustainable forest management could increase wood quantity and quality at the same time. Habitats for a wide variety of wildlife species could be enhanced. The virtues of improved forest management and buying locally produced goods are widely extolled, but what might that actually look like on the ground? More specifically, how could enhanced forest management make more locally produced forest products available to meet New Englander's own needs, as well as for export, improve the local and regional economies and provide the greatest social and environmental benefits?

The purpose of this project is to document the productive potential of the region's forests. The best available data from the US Forest Service, state forestry agencies, and universities was used to characterize this potential. To realize this potential, exemplary forest management will be required, the results can include not only increases in the quantity and quality of timber, but also concurrent improvements in wildlife habitats, recreational opportunities, and air and water quality.

This project of the New England Forestry Foundation is intended to complement other efforts aimed at not only conserving New England's forests, but also enhancing New England's agriculture and fisheries. The New England Forestry Foundation intends to complete this project this year.

II. The Opportunity – An Overview

This paper focuses on the potential to reduce greenhouse gas (GHG) emission levels by using wood in construction in place of other construction materials.[1] We recognize that this is only part of the opportunity to substitute wood for other materials but chose it because it is a substantial opportunity and has been studied more than some others. This paper describes an immediate opportunity to use conventional wood-framed construction for one- to six-story buildings in New England that have traditionally been constructed with steel or concrete. Even when the effects of harvesting wood on carbon sequestration in the forests are considered, the overall GHG levels associated with wood construction of these buildings are significantly lower, over a longer time horizon, than emissions levels associated with traditional steel or concrete building methods, especially when wood byproducts and wood wastes are used to displace fossil fuels.

This paper also discusses emerging opportunities to use new engineered wood products (EWPs) to build with wood, including using "mass timber" technologies to build skyscrapers up to 30 stories.[2] Innovations in design and in EWP product technology, including cross-laminated timber (CLT) and Laminated Strand Lumber (LSL) now make it possible to build taller buildings with wood cost effectively.

It is beyond the scope of this paper to discuss the opportunity to use wood for other materials, such as plastic in furniture or pallets or vinyl in flooring. Although this subject has received less attention by researchers than the GHG benefits of substituting wood for other materials in construction, the same factors and variables are relevant.

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[1]The author is very grateful to David Bruce McKeever, USDA Forest Products Laboratory; Blane Grann, FPInnovations; Morton Barlaz, NC State University; and Stephen Shaler, University of Maine, for providing information for this report that was not available through published sources.

[2] The term "engineered wood products" refers to composite products manufactured by binding or fixing together wood strands, particles, fibers, veneers or boards, generally with adhesives. Plywood is the best know EWP, a category that now includes a wide range of products. "Mass timber" products are engineered wood products designed to be used instead of or to complement light- and heavy-timber framing options.

III. The Opportunity to Use More Wood Products in Construction in New England

A. An Immediate Opportunity: Expanding the Use of Wood in Low-Rise (One-to Six-Story) Construction

By contrast to residential construction (discussed below), nonresidential construction is most frequently built with steel or concrete structural elements. In 2011, in new low-rise nonresidential buildings (defined as 1- to 6- story) the structural material of choice was steel (63%)[3] or concrete (25%), while wood was used in only 12% of these buildings.[4] (Adair, et al. 2013, Table ES3).

In a recent study, Adair, et al. (2013) applied a "Building Potential Filter" to approximately 10,000 nonresidential low-rise buildings in the US to determine which could readily have been built with wood-framed construction rather than steel or concrete under the International Building Code (IBC)[5] with few or no design changes (David McKeever, pers. comm., March 31, 2014). The authors concluded that wood was used as the primary structural material in a little under a quarter of the projects in which it would have been allowed. They estimated that of projects with the potential to be built with wood framing, 83.7% were built from steel and 16.3% were built from concrete (calculated by building square footage) (see Adair, et al., Table 3). The opportunity to substitute wood for concrete and steel varied by type of building, with stores and schools representing the greatest opportunities.

Using actual construction data from 2011 as a baseline, Adair, et al. quantified the potential additional wood product volume that could have been used in 2011 if wood had been substituted for steel and concrete in low-rise nonresidential construction where allowed. Because the 2011 construction level was significantly reduced by the recession (down 35% from 2008), the authors also extrapolated potential wood volumes for a more typical year. Based on a set of factors, they calculated that upon economic recovery, the potential additional use of wood in low-rise nonresidential construction would be 1.7 times the 2011 level.

In Table 1, column 1 reports actual wood usage reported in nonresidential low-rise building construction in New England in 2011 (see Adair, et al., Table ES1). Column 2 represents the additional potential wood usage calculated by Adair, et al. (see Tables ES-4 and 15) based on the "Building Potential Filter" analysis at 2011 construction levels. Column 3 presents 2011 actual wood usage adjusted to a typical year's construction volume (1.7 x column 1). Column 4 presents additional potential wood usage for New England adjusted to a typical year's construction volume (1.7 x column 2).[6]

Table 1. Actual and potential wood usage in New England low-rise nonresidential construction at the 2011 level and for a typical construction year

[3] Percentages are by floor area square footage.

[4]In 2011, 84.3% of new nonresidential construction floor space was in buildings of 4 stories or less, and 93.7% was in buildings of 6 stories or less.

[5] These did not include "mass timber" buildings, which could be allowed on a case-by-case basis under Section 104.11. (See discussion of mass timber buildings, below.)

[6] Adair et al. do not calculate potential in a typical year by region. The numbers in columns 1 and 2 were reported for New England, and columns 3 and 4 represent these numbers multiplied by the 1.7, which is the overall US multiplier Adair et al. determined to be appropriate to adjust for a typical year. (Adair et al. 2013, Table 15 and p. 7).

	New England Actual 2011 *(Th. bfe)**	New England Additional Potential at 2011 Level *(Th. bfe)*	New England Actual Adjusted to Typical Year (1.7 x col. 1) *(Th. bfe)*	New England Additional Potential in Typical Year (1.7 x col. 2) *(Th. bfe)*
Lumber	41,059	84,231		
EWP**	17,860	71,983		
Panels	21,575	42,721		
Total	80,494	198,935	136,840	338,190

**Th. bfe* = Thousand board feet equivalent. (Adair et al. 2013) use the abbreviation "Th. bfe" for thousand board feet equivalent; most reports use "MBF" to refer to thousand board feet.)
** EWP – Engineered Wood Products

Therefore, based on the analysis in Adair, et al, there is a projected immediate potential under the current IBC to substitute wood for steel and concrete framing in low-rise nonresidential construction in New England, using an **additional** 338,190,000 *bfe* annually in the walls, floors, roofs and siding of these buildings in a typical construction year.[7] This wood volume would be in addition to the estimated 136,840,000 *bfe* used to frame such buildings at the rate in which wood is currently used in such structures.

By contrast to non-residential construction, one and two family residential buildings in the US and in New England are overwhelmingly constructed with wood as their primary structural material (90 to 94%) (Ingerson 2011). Nonetheless, there is an opportunity in New England to use wood to substitute for concrete slabs and in walls in residential construction, in both single family and low-rise residential construction. Based on a report of wood usage in residential construction in 2006, there will be a potential, upon economic recovery to use an **additional** 64,750,000 bfe. annually in residential construction in floors and walls. (Adair and McKeever (2009), also David B. McKeever, Research Forester, USDA Forest Products Laboratory, pers. comm. May 5, 2014.)[8]

A separate analysis of potential wood supply in New England has been prepared as part of this project. Suffice it to say here, the potential supply of wood from New England's working forest could meet a significantly expanded demand for wood construction materials. Supplying an additional 676,280 cords (approximately equivalent to 338,190,000 *bfe)* for non-residential construction and 129,000 cords (64,500,000 bfe) in residential construction compares with the projection of an increased supply of potentially several million cords in the analysis in another paper prepared for NEFF.[9]

[7] Based on their analysis, Adair et al. (2013, Table 15) estimate that there will be the potential at the national level under the IBC to use an additional 2,640,314,000 *bfe* annually for traditional wood-framed construction in non-residential construction in a typical year.

[8] This figure is based on an estimate that 35% of potential estimated for the Northeast would be potential in New England. (McKeever, pers. Comm., May 5, 2014). Potential to substitute wood for non-wood siding is not included here, but provides an additional opportunity to substitute wood for non-wood materials, primarily vinyl.

[9] The amount of additional wood that could be produced by New England's working forests in the future depends on the acres retained as working forests and the degree to which "exemplary" forestry is practiced. Rather than producing an average only 0.29 cords/acre, production could be doubled to 0.60 cords/acre. If the current acreage of New England's working forest lands were retained (28,193,503 acres) and exemplary forestry were universally practiced it would result in production of approximately 16,916,000 cords, up from the current production of approximately 8,154,000 cords.

B. The Potential of New Engineered Wood Products to Expand Wood Use in Construction, Including in "Mass Timber" Construction

The potential described above focuses on substituting traditional wood framing for other construction materials when possible under the current IBC. The analysis in Adair, et al. (2013) also revealed that at the time the study was conducted, traditional wood framing as the primary structural material could not be readily used under the IBC in buildings representing almost half of the floor space in nonresidential buildings up to six stories, or in any buildings with more than six stories.[10]

There is growing interest in the use of new engineered wood products (EWPs), especially in "mass timber" products and technologies, in buildings in which light wood frame construction is not suitable. Mass timber technologies greatly expand the opportunity to use wood for other construction materials in low-rise (one- to six-story) buildings. While US and Canadian building codes, (which are usually based on the IBC) do not currently explicitly recognize mass timber systems, this does not prohibit their use under IBC Section 104.11, which provides that design professionals can use alternative materials, design and methods of construction if the proposed design complies with the intent of the provisions of the code (Mass Timber 2013).

Marselle's 132-unit condominium building, a wood (primarily EWP) and concrete structure in Seattle. The developer reported that building the top five stories with wood saved 30% in construction costs. Photo: Matt Todd, courtesy WoodWorks.

Moreover, mass timber technologies could be used to build taller structures, including buildings up to 30 stories (MGB Architecture & Design Equilibrium Consulting 2012). In the US in 2011, new buildings and additions of over 6 stories comprised 6.3% of total new square footage in nonresidential building construction (Adair, et al. 2013, Table 18**.** In certain categories, buildings over 6 stories comprised a higher percentage, including 33.2% of hotels and 17% of

[10] Adair et al. (2013) identified buildings that could readily be built under the IBC and therefore did not include buildings that could be built under Section 104.11 of the IBC, which provides that designers may seek to establish on a case-by- case basis that alternative designs meet the intent of the code.

health care construction (Adair, et al., Table 18). Assuming policies encouraging compact urban growth continue, the demand for medium and high-rise buildings is likely to grow over the coming decades.

CLT has received considerable attention as a mass timber building material. CLT panels are fabricated from several layers of lumber boards stacked crosswise and glued together on their wide faces, and sometimes, on the narrow faces as well. The US edition of the CLT Handbook (Karacabeyli and Douglas 2013) reports that thicknesses of individual pieces of lumber may vary from 5/8 inches to 2 inches and width may vary from approximately 2.4 inches to 9.5 inches; typical panel widths are 2, 4, and 8 feet; panel lengths can be up to 60 feet; and panel thickness can be up to 20 inches. The product is much stronger than traditional wood framing and can be substituted for steel and concrete framing in larger buildings. CLT and glulam beams can be used for spans that have traditionally been constructed with steel, including floors, load-bearing walls, and roofs. CLT can be fabricated to specification when produced and bolted together on the construction site quickly.

Tall Wood:
MGA | Michael Green Architecture
Project Lead Architect: Michael Green, Architect AIBC

CLT was developed in Europe in the early 1990s, e.g., it has been used in low- and mid-rise buildings where it has been used to construct multiple 8-story buildings and a 9-story building in England. A 10-story building has been built in Australia. Mass timber buildings built around a concrete core have been approved for a 30-story building, and proposed for a 34-story building in Sweden (Frearson 2013, Njus 2014). CLT can be used in taller buildings where traditional wood construction is not appropriate because of its strength and because of its resistance to fire (see "*The Case for Tall Wood Buildings*," MGB Architecture and Design Equilibrium Consulting 2012). Approved changes in the IBC effective in 2015 will recognize CLT products that are produced according to standard and will streamline the acceptance of CLT buildings.

The authors of both the "*CLT Handbook*" and "*The Case for Tall Buildings*" analyzed the comparative costs of constructing mass timber buildings and equivalent concrete and steel buildings and found mass timber buildings to be cost competitive, particularly in mid and high-rise buildings. (See Karacabeyli and Douglas 2013, ch. 1, section 9.2.1 – shells of 5- and 8-story mass timber buildings less expensive that equivalent nonwood building shells; and MGB 2012 – 12- and 20-story buildings slightly less or slightly more expensive (depending on building design) than nonwood alternative for buildings in British Columbia, Canada). There are currently two manufacturers of CLT in Canada, but none in the US.

Other EWPs also make it possible to expand substitution of wood products for concrete and steel. The University of Maine and research partners have received a grant to evaluate the potential to use a patented Laminated Strand Lumber (LSL) product in panels for taller buildings in some applications where CLT would be suitable. LSL is a structural composite lumber manufactured from strands of wood species or species combinations blended with an adhesive.

The strands are oriented parallel to the length of the member and then pressed into mats using a steam injection press. New England has one of the nation's only LSL mills, which produces the product from aspen and mixed hardwoods. The University of Maine's Advanced Structures and Composites Center has also developed Deltastrand, a strong EWP made from red maple, which could be used where traditional wood framing is not suitable. These products will expand the opportunity to substitute wood for other construction materials, with the potential to utilize wood species and quality that have not previously been useful in construction materials.

IV. Methodological Issues in Evaluating the GHG Impacts of Substituting Wood for Steel or Concrete in Construction

There is significant interest in evaluating the potential to reduce GHG emissions by using more wood in construction, especially in substituting wood products for steel and concrete, which require high levels of energy to produce.[11] Researchers undertaking these analyses face a number of methodological questions and have used a variety of approaches, often making comparison of studies difficult.

A. Overview of Factors Relevant to Evaluating GHG Benefits of Substituting Wood for Other Construction Materials.

The GHG consequences of building with concrete and steel include emissions associated with: 1) obtaining the raw materials; 2) manufacturing the construction material; 3) constructing the building; and 4) demolishing and disposing of the materials at the end of the building's life.[12] Analysis of GHG consequences of concrete should include analysis of both emissions associated with manufacture of concrete and the reabsorption of CO_2 overtime. When cement is produced, calcium carbonate disassociates into calcium oxide and CO_2 (a process called calcination). Concrete will, over time, reabsorb some percentage of the CO_2 emitted upon manufacture through the reverse reaction in which calcium oxide recombines with CO_2 to produce calcium carbonate (carbonation). Crushing concrete when a structure is demolished significantly increases its reabsorption of CO_2 (studies of rates of carbonation are summarized in Sathre and O'Connor 2010, p, 88).

The organic nature of wood adds additional complexity to studies of the GHG impacts of wood in construction. In addition to the emissions from obtaining the wood, manufacturing wood products, constructing buildings and disposing of wood materials at the end of life, it is also necessary to evaluate: 1) the net GHG consequences in the forest of harvesting wood for construction materials and the forest's ability to regenerate and sequester carbon over time; 2) the potential to use wood waste as a fuel source; and 3) questions concerning release of carbon and methane from discarded wood products in landfills.

The timeframe chosen is also critical. If one evaluates the GHG emissions of using wood over a longer timeframe, such as 100 years, forests have time to re-sequester much or all of the forest

[11] Energy needed to produce a material is often called "process energy." The term "primary energy" is often used to describe process energy, the energy used in the construction process and the "feedstock" energy that remains stored in the building material. The term "embodied energy" is defined differently in various contexts and in different studies. EPA's TRACI methodology includes both process energy and feedstock energy (carbon stored in the building) in the term "embodied energy" (see Robertson et al. 2012).

[12] Studies differ in their approach to allocation of energy associated with steel recycling and use of recycled steel. For a discussion of these differing approaches, see Grann (2013).

carbon removed when wood was harvested for construction products, but emissions associated with disposal of wood products at end of buildings' lives (generally by incineration or disposal in landfills, with possible methane emissions) become critical to the analysis. Forest managers concerned with sustainability are accustomed to using such longer timeframes, as they include one or more rotations of timber growth.

Policymakers concerned with GHG impacts on climate, however, are frequently concerned with reducing GHG emissions and climate impacts the next 20 to 30 years and are increasingly looking to understand – and at times create incentives for – strategies that reduce emissions in the next two or three decades. The most recent reports from the IPCC (AR5) emphasis the importance of significantly reducing over all GHG emissions from any source over the next 25 years. In analyses of climate impacts of building with wood in this shorter timeframe, the net GHG impact of harvesting on forest carbon storage plays a significant role, as does the biomass energy used in product manufacturing, as forests may not recover this biomass energy in the forests within this short timeframe, while the role of GHG emissions at the end of buildings' lives is less relevant.

B. Life Cycle Assessments Defined

Life Cycle Assessments (LCA), also called a Life Cycle Analyses, of building materials and whole buildings are the essential analytical tools for comparison of the GHG impacts of different types of buildings. The four phases of LCAs for all materials are set out in ISO[13] 14040 and 14044: 1) goal and scope definition; 2) life cycle inventory; 3) life cycle impact assessment (LCIA); and 4) interpretation. The Athena Sustainable Materials Institute, a leader in evaluating the environmental impact of different building materials and building systems, explains phases 2 through 4 as follows:

> LCI... the **life cycle inventory**... is the data collection portion of LCA. LCI is the straightforward accounting of everything involved in the "system" of interest. It consists of detailed tracking of all the flows in and out of the product system, including raw resources or materials, energy by type, water, and emissions to air, water and land by specific substance…
>
> LCIA is **life cycle impact assessment**, the "what does it mean" step. In LCIA, the inventory is analyzed for environmental impact. For example, manufacturing a product may consume a known volume of natural gas (this data is part of the inventory); in the LCIA phase, the global warming impact from combustion of that fuel is calculated. There are various methods globally for categorizing and characterizing the life cycle impact of the flows to and from the environment, which can somewhat complicate the comparability of different LCA studies. Other variables in LCIA include the system boundary (how far upstream, downstream and sidestream does the analysis go), the functional unit (what is the volume/mass/purpose of the object being assessed), and specific LCIA methods such as allocation (how are impacts assigned to the product and by-products, on what basis)…
>
> Life cycle **interpretation** [is the] phase of life cycle assessment in which the findings of either the inventory analysis of the impact assessment, or both are evaluated in

[13] International Organization for Standardization, Geneva

relation to the defined goal in scope in order to reach conclusions and recommendations. (Athena Sustainable Materials Institute 2014.)

C. System Boundaries in LCAs

As the Athena Sustainable Materials Institute notes above, to conduct an LCA of wood products or building designs, researchers must define study parameters, called "system boundaries." Studies can be grouped by systems boundaries chosen.

1. "Cradle to Gate" LCAs – Analyses of Processing Emissions up to the Construction Site

These studies consider emissions from obtaining and processing a material up to a construction site. "Cradle to gate" LCAs of forest products typically cover emissions from the following:

- Forest management, such as planting, fertilizing
- Harvesting equipment
- Transportation (to the mill, to secondary processing and to the construction site) (generally relatively insignificant, if in continent)
- Primary processing – milling and kiln drying
- Wood waste – the wood removed as excess or residue to create the product (which varies by type of wood product, condition of the wood, efficiency of the mills' equipment):
    - This wood waste may be more than 50% or the live wood. If it is converted into a short-lived products or discarded, some emissions from these products take place in a short time frame.
    - Wood waste may be used for energy to produce the wood product. Many studies do not account for these emissions, reasoning that forests will reabsorb this carbon as they regrow. Increasingly, researchers consider it important to account for these emissions, as the timing of all emissions, including biogenic emissions, has climate consequences (Searchinger 2012, Cherubini, et al. 2011).
    - Wood waste may be used for energy to perform some function unrelated to production of the wood product. (These emissions may be partially offset by the emissions from fossil fuels for which biomass energy substitutes. The offset is partial because biomass produces less energy per unit than fossil fuels.)
- Secondary processing – e.g., creating EWPs or finished products such as windows.

It should be noted that secondary processing energy for some products is significant. For example, Puettmann and Wilson (2005) reported that in comparison to the energy required to produce a cubic meter of kiln-dried lumber in the southeast (3,492 MJ/m3), it requires almost twice the energy to produce an equivalent quantity of glulam (6,244 MJ/m3) and over 3 times the energy to produce an equivalent quantity of OSB (11,145 MJ/m3). (Puettmann and Wilson reported that resin production accounts for significant amounts of energy required: 8% for glulam and 28% for OSB). While LCIs for various wood products vary by region and efficiency of processes, these results give an idea of variability of process energy requirements of different wood products.

2. "Cradle to Grave" LCAs: Expanding the Analysis to Include Emissions at the Construction Site and Emissions from Disposal of Construction Materials at the End of Building Life – Incineration or Landfilling

"Cradle to grave" studies typically include "cradle to gate" emissions listed above, as well as emissions from the construction process and from disposal of construction site waste. These studies also address the complex issue of emissions from disposal of wood products that cannot be recycled at the time of demotion of buildings.

Dry wood is approximately 50% carbon. Precisely because wood stores carbon, the fate of this stored carbon at the end of the life of a building significantly affects the analysis of climate benefits over longer time horizons. The estimated “half-lives” for single family homes has been rising and is now estimated at 100 years for a house built after 2000; the estimated half-lives of multi-family residential structures is 70 years, for non-residential structures 67 years, and 30 years for wood furniture (Smith, et al. 2006, Table D3). When no longer useful as a wood product, building materials can be: 1) burned to create energy; 2) allowed to decompose or used for a short-lived product such as mulch, releasing most of its carbon to the atmosphere; or 3) placed in a landfill.

Sathre and O’Connor (2010) analyzed 66 US and international studies of the GHG impact of wood in construction. They found that over the full life cycle (cradle to grave) of building materials, end-of-life management of wood was the single most significant variable in the carbon profile of wood products. The majority of studies reviewed assumed incineration of wooden construction materials at the end of building life.[14]

Landfilling is another common fate of wood products from buildings at the end of building life. In the US, some construction and demolition (C&D) waste is taken to municipal landfills, while some is discarded into separate designated waste sites. There is a scientific consensus that wood does not decompose in anaerobic conditions present in landfills to the same extent as other organic materials. Anaerobic decomposition of solid wood and some wood products is known to progress more slowly and be more limited than decomposition of most other organic materials because of the high lignin content in wood. Lignin in wood is highly resistant to decomposition and inhibits decomposition of cellulose and hemicellulose – also present in wood – in anaerobic settings. As a result, carbon from all of the lignin and some of the cellulose and hemicellulose from wood products remains in permanent storage in landfills. A recent study suggests that the percentage of wood carbon that decomposes in landfills (“carbon conversion”) varies between hardwood and softwoods and may be smaller than previously thought. Wang, et al. (2011) found carbon conversion in various wood products to be 2% to 8%, (with one exception[15]), lower than the 13.6% carbon conversion percentage found by Eleazer, et al. (1997) for branches and lower than the 23% carbon conversion of wood carbon reported by Skog (2008).

Although only a small percentage of wood may decompose in a landfill, the portion of wood that does decompose may contribute to production of methane. It is this pathway that is the most concern and creates significant variables. Methane is a short-lived but extremely potent GHG. Although methane breaks down in the atmosphere in approximately 12.5 years, its “global warming potential” is 72 times that of CO_2 over 20 years, and approximately 25 times that of CO_2 over 100 years (IPCC 2007). An analysis of GHG consequences of landfilling wood construction products therefore requires evaluation of a number of factors: the carbon conversion percentage of wood; the amount of methane and CO_2 produced by the landfill from the wood; the percentage a methane produced that is captured; whether the captured methane is flared and/or used as fuel (which results in substitution benefits as well as biogenic emissions). The LCAs

[14] In these studies, the incineration was considered as substituting for fossil fuels, and the GHG from the wood product were sometimes considered carbon neutral. This approach to carbon accounting, like carbon accounting of biomass in the production of wood products, is increasingly controversial, as is discussed, below.
[15] Carbon conversion for hardwood was found to be higher than for softwood products (2% compared to 8%). OSB made from hardwood exhibited 19.9% carbon conversion. Wang et al. 2011.

reviewed in Sarthe and O'Connor (2010) applied a range of assumptions, usually reflecting studies that establish that wood has a low carbon conversion percentage in anaerobic conditions. Among the LCAs reviewed by Sarthe and O'Connor, the only ones that found the use of wood in construction to be more damaging to the environment than use of concrete and steel are studies that assume wood products are discarded in landfills where they create uncontrolled methane emissions, often assuming decomposition percentages of wood carbon that are not supported by more recent studies.

3. Studies Integrating the Effects on Carbon in Forests: The Intersection of System Boundaries and Time Horizons

A full analysis of the GHG impacts of using wood in construction must include an analysis of the impact of harvesting on the forest carbon storage and sequestration levels. Forest systems store carbon in above ground living biomass (trucks, branches, and leaves), in below ground biomass (roots), in dead wood and leaf litter, and in forest soils. Growing trees sequester carbon. The rate and volume of sequestration varies with species, conditions, and the age and condition of the forest. Young trees have a higher sequestration rate than mature trees, but a stand of young trees will store a lower total volume of carbon than a healthy forest with more mature trees. Carbon dynamics are different at the stand or landscape level. Lippke, et al. (2011) and Sathre and O'Connor (2010) report that forests that are managed appropriately can remain stable or increase in carbon stock over time, while unharvested forests may ultimately reach a dynamic equilibrium, where new growth is balanced by tree death and decay. Forests damaged by drought or infestation, such as western Canadian forests damaged by the Mountain Pine Beetle, can become net carbon emitters (Robertson, et al. 2012). Other researchers have reported that net sequestration can continue for very long periods of time. (Luyssaert, et al. 2008).

The impact of harvesting on a forest will vary by forest region, forest health, intensity of harvesting, intermediate treatments such as thinning, treatment of residues, and rates of regeneration. As a threshold matter, however, the sensitivity of the LCA to effects of harvesting on forest carbon storage varies significantly based on the time horizon chosen. Studies that cover time periods exceeding one or more full rotations in a sustainably managed forest usually assume that the temporary reduction in carbon storage in the forest associated with harvesting is recovered as the forest regrows. The majority of comparative LCA studies discussed in this report use these longer timeframes and assume that effects of harvesting on forests and biogenic emissions are neutral over the timeframe of the study.

Forest harvest rotations in the Northeast average between 50 and 100 years. Therefore, studies of GHG impacts of increasing the use of wood in construction in shorter time horizons, such as 20 or 30 years, cannot rely on the assumption that forest carbon loss associated with harvest will be re-sequestered in the forest in these shorter timeframes.[16] Studies focusing on shorter time horizons must carefully compare the GHG consequences – including the biogenic GHG emissions from any use of wood waste as fuel and the drop in carbon storage and, at least temporarily, sequestration in the forest after harvesting – against the potential reduction of GHG emissions avoided when wood is substituted for fossil fuels and wood is used instead of energy-intensive alternative building materials. For this reason, results of studies that assume carbon neutrality of forest effects and biomass burning cannot be relied on to address GHG impacts over

[16] The assumption that biogenic emissions should be considered climate neutral has increasingly come under scrutiny, especially in studies addressing climate impacts in shorter timeframes (Searchinger 2012, Cherubini, et al. 2011).

a 20 or 30 year horizon. On the other hand, evaluation of complex questions concerning emissions associated with disposal of wood products upon building demolition by incineration or landfilling is generally not relevant in the 20 to 30 year time horizon.

4. Looking Beyond GHG Effects to Integrate Albedo and the Concept of Cumulative Radiative Forcing – a Significant Step Forward in Analysis of Climate Impacts of Building with Wood versus Concrete and Steel

A number of researchers have identified the need to look beyond GHG impacts to include biogeophyscial factors, especially changes in surface reflectivity (albedo) (Marland, et al. 2003, Cherubini, et al. 2012, Zhao and Jackson 2014, Montenegro, et al. 2009). When forests are harvested, particularly through clear-cutting, the ground is exposed for a period of time while the young forest regrows. In climates with significant winter snow cover, surface reflectivity during winter months is increased, reducing "radiative forcing" of the earth. In some locations, this change in reflectivity can, for a period, be so significant that it offsets the climate impacts of reduced carbon storage in the forests. Cherubini, et al. (2012) put forward a methodology for evaluating the combined effects on forest carbon of harvesting for biomass fuels and albedo change. The authors emphasize that the results vary based on local conditions and tree species. For example, they found that in Wisconsin, the climate benefits of albedo change caused by forest harvesting for biomass fuel are likely to outweigh slightly the negative climate effects of carbon loss in the first 20 years after harvest, while in the Pacific northwest, the negative carbon impact of harvesting for biomass fuels is likely to outweigh significantly the climate benefits of temporary albedo change in the first 20 years after harvest, even when harvest sites are at approximately the same latitude and altitude.

The methodology suggested by Cherubini was applied in a comprehensive LCA comparing a mass timber building constructed in Quebec Province to an equivalent steel and concrete building (Grann 2013). This thorough and comprehensive cradle to grave study: 1) analyzed biogenic as well and fossil fuel emissions; 2) accounted for carbonation by concrete in both buildings over 100 years; 3) analyzed the net effects of albedo and forest carbon dynamics; and 4) analyzed the cumulative radiative forcing of various impacts, thereby accounting for the timing of climate impacts over the 100 year time horizon of the study. This study, which is discussed further below, illustrates the potential significance of incorporating albedo in northern latitudes. It reports that in the study location in Canada, the albedo change significantly offset the loss of carbon storage in the forest immediately after harvest, reducing the global warming impact of the mass timber building. The methodology used in this study will be important in the future to a more comprehensive analysis of the climate impacts of harvesting wood, particularly in northern New England.

V. LCAs Comparing Traditional Wood-Framed Construction to Equivalent Concrete and Steel Construction over Longer Time Horizons

Any comparative analysis of uses of wood and other materials must define the "functional unit" to be compared. Functional units may compare products by mass or volume, by building component, by complete building, or by services provided. While it is possible to compare LCAs of wood products with energy needed to produce steel and concrete by product unit or weight, it is more useful to compare units that provide equivalent functions in a construction context, e.g., to compare a wood-framed structural system to steel or concrete-framed systems for buildings of equivalent size and function.

Most US and Canadian LCAs comparing equivalent buildings use the Athena Institute's "Athena Impact Estimator" (AIE), which incorporates aspects of the US Environmental Protection Agency's (EPA) "Tool for Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) methodology. AIE provides energy data for materials and building types by geographic region and provides analyses of categories of environmental impact, including a factor defined as Global Warming Potential (GWP).[17] It is important to note that the GWP measure reported in AIE does not include emissions from biomass, either when used in production of wood materials or when materials are incinerated at the end of life. LCAs using the AIE generally report total biomass and fossil energy required for materials and buildings as well as GWP. GWP reported by AIE and TRACI do not include forest carbon impacts of harvesting, carbonation by concrete, or albedo.

In the US, the best known building-to-building comparative analyses have been conducted by the Consortium for Research on Renewable Industrial Materials (CORRIM), applying a variant of AIE. Researchers compared the energy required for materials and construction of an equivalent wood-framed and concrete house in Atlanta and an equivalent wood-framed and steel-framed house in Minneapolis. While the houses being compared had many structural elements in common, the wood-framed house required 16% less total energy than the equivalent concrete-framed house and the wood-framed house required 17% less total energy than the steel-framed house. (Bowyer, et al. 2004, Module J, Table 3.3.) A more direct comparison of the above ground wall structural elements showed that concrete block required 262% more energy than wood framing, and steel wall structural elements required 43% more energy than wood framing. (Bowyer, et al. 2004, Module J, Sections 5.1.1 and 5.2.1; see also Lippke and Edmonds 2006).

Gustavsson and Sathre (2004) used the methodologies of three different studies conducted in Europe to analyze the comparative energy used in manufacturing wood and concrete materials for construction of a wood-framed and concrete-framed four-story apartment structure built in Sweden. The authors found that while the absolute results varied significantly depending on assumptions and methodologies of the three studies, the ratio of energy required to produce lumber for the wood-framed building versus the concrete for the concrete-framed building was on the order of magnitude similar to findings by CORRIM (Gustavsson and Sathre 2004, set out in Sathre and O'Connor 2010, p. 87).

Sarthe and O'Connor (2010) analyzed 66 studies of the GHG impact of wood product use from a life cycle perspective, including 21 studies addressing the comparative GHG impacts of wood and other construction materials. Sarthe and O'Connor conducted a meta-analysis of the 21 studies addressing the comparative GHG impacts of building with wood as compared to concrete and steel. Results of these studies ranged widely, in part because researchers choose different system boundaries and related assumptions. Key differences in boundaries and assumptions included: 1) whether cradle to gate or cradle to grave, 2) the timeframe considered; 2), the assumptions about "end of life" emissions from wood materials when a building is demolished; 3) decisions to include or exclude the effects of harvesting trees on carbon storage and sequestration rates in the forest; and 4) researchers' assumptions about the GHG implications of using wood residues and wastes for energy. (The significant majority of the 21 studies either

[17]AIE also provides information on impacts on acidification, eutrophication, smog, and ozone depletion drawn from the data developed in the LCI phase of the LCA.

used a longer time horizon, or did not include forest impacts of harvesting.)

Sathre and O'Connor noted that while study results varied, the studies yielded an average mid-range "displacement factor" (amount of GHG emissions avoided over the long term when using wood instead of another material) of 2.1. The authors concluded that this mid-range estimate of a 2.1 displacement factor is a reasonable estimate of GHG mitigation efficiency of using wood products to substitute for a range of other products in construction. They converted the 2.1 displacement factor as follows: "for each tC in wood projects substituted in place of nonwood products, there occurs in average GHG emission reduction of approximately 2.1 tC. Expressed in other units, this value corresponds to 3.9 t CO_2eq emission reduction per oven–dry t of wood product, or roughly 1.9 t CO_2eq emission reduction per m3 of wood product assuming a dry weight density of 500 kg/m3" (Sarthe and O'Connor 2010, p. 114).[18]

In the absence of a study based specifically on the conditions in New England forests, this rule of thumb provides a useful measure of the approximate GHG benefits of substituting traditional wood for other construction products *over the long term*. Indeed, it is the displacement factor built into ForGATE, a forest sector GHG assessment analytical tool for Maine developed in a collaboration between the Manomet Center for Conservation Sciences, University of New Brunswick, and the USDA Forest Service (Hennigar, et al. 2013).

Based on the information concerning the immediate opportunity to use more wood in buildings set out in Section II, above, there will be an immediate opportunity to increase the use of wood in construction, upon economic recovery, by an additional 402,940,000 bfe (approximately 805,880 cords). (This is the total of 338,190,000 board feet (approximately 676,380 cords (estimated at 1 Th. bfe = 2 cords), substituting wood for concrete and steel construction in non-residential construction, and 64,750,000 bfe (approximately 129,500 cords) in residential construction floors and walls. Based on the displacement factor provided by Sarthe and O'Connor (2010), this would yield an annual CO_2e savings of nearly 3.5 million metric tons of CO_2e[19] (based primarily on studies with longer term horizons). (Based on the EPA Greenhouse Gas Equivalencies Calculator, this is the equivalent of annual emissions from nearly three quarters of a million cars [US EPA 2014a.)[20] There are additional opportunities, although less quantifiable at this time, of substituting wood in buildings in which "mass timber" systems can be used.

[18] It should be noted that this meta-analysis does not apply to GHG impacts over a 20 to 30 year time horizon, or to buildings built with "mass timber" systems, which present different variables. See Sections V and VI, below.

[19] Assumptions for calculating the CO2e emissions avoided based on Sarthe's displacement factor: 1 Th. bfe = 2 cords; 1 cord = 1.25 U.S. tons (US) dry weight, of which 50% = C (.62 tons C per cord); 805,880 cords x .62 C = 499,646 tons (US) C; 499,646 tons C x 2.1 (Sarthe displacement factor) = 1,049,257 tons (US) C avoided; 1,049,257 tons (US) C x 3.67 (converting C to CO2e) = 3,,850,773 tons (US) CO2e avoided; 3,,850,773 US tons x .9 (conversion to metric tons) = 3,465,695 metric tons CO2e avoided. Note: The same result may be reached by converting directly from Th bfe, if one assumes 1 Th. bfe = 2.5 tons dry weight, of which 50% C, or 1.25 T carbon per 1 Th bfe.

Note: Adair et al, 2013) use the abbreviation "Th. bfe" for thousand board feet equivalent; most reports use "MBF" to refer to thousand board feet.

[20] Based on the EPA Greenhouse Gas Equivalencies Calculator, this is the equivalent of annual emissions from 729,620 cars.

VI. The Need for Studies to Compare GHG Impacts of Traditional Wood-framed Construction to Equivalent Concrete and Steel Construction over Shorter Time Horizons

As is discussed above, to evaluate the relative merits of building with wood versus concrete and steel in a 20 to 30 year horizon, it is critical to analyze the impact of harvesting on forest carbon dynamics in this time horizon. While a New England-based study has been done on the impact of harvesting for biofuels (Walker, et al. 2010), no equivalent study has yet been undertaken to analyze the comparative GHG impacts of building with wood harvested in New England versus building with concrete and steel in this shorter time horizon. A significant study conducted in the UK provides useful information and provides a sense of the magnitude of potential benefits.

Matthews, et al. (2014) analyzed the GHG impacts of options for the use of wood harvested from UK forests to address these questions, among others:

- Is it better to leave wood in the forest or harvest it for timber, other wood products (e.g. panel boards) and/or fuel?
- Is it better to use harvested wood to provide materials or fuel?
- Are there particular options involving the use of (UK) wood that clearly offer the biggest benefits? Are there other options that should be avoided?

Matthews, et al. (2014) found potential GHG benefits of harvesting softwood forests already under active management. Figure 1 (next page) represents some of the results from the Matthews study for a Sitka spruce forest managed by rotating harvesting of stands on a 56-year rotation.

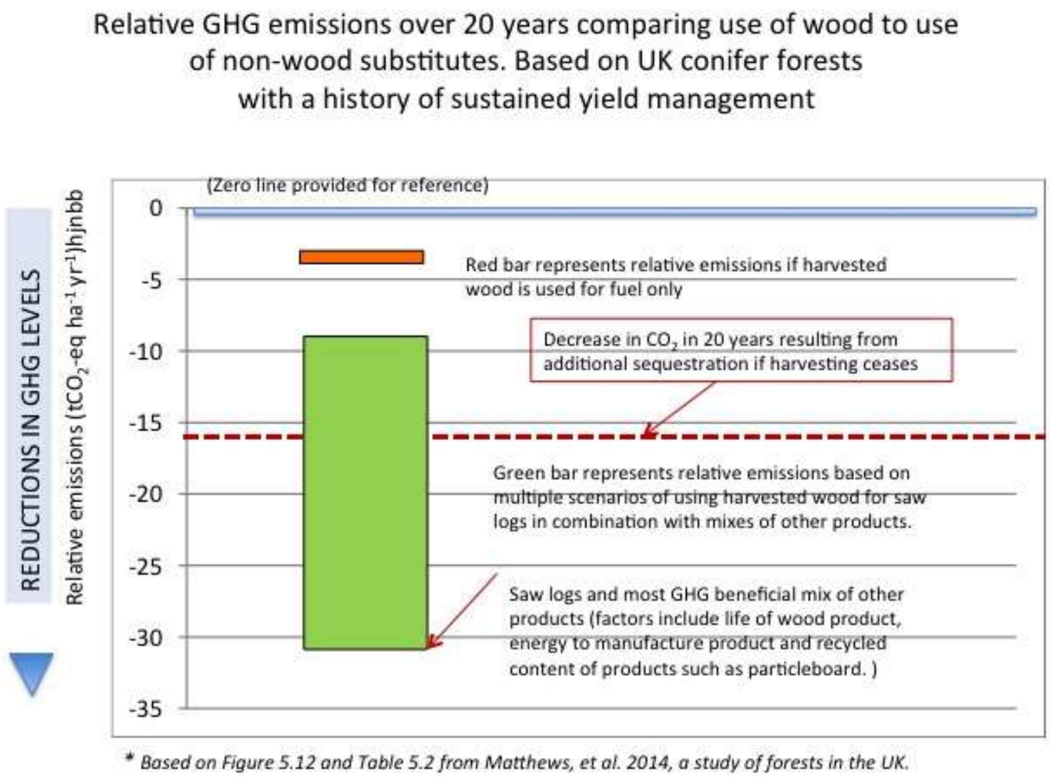

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Figure 1. Relative GHG emissions over 20 years comparing use of wood to use of nonwood substitutes. Based on UK conifer forests with a history of sustained yield management

Matthews, et al. found the greatest GHG benefits to result from use of harvested wood as follows: sawlogs used for timber; sawlog offcuts and small roundwood used for particleboard (also using 70% recycled wood); and bark and 50% of branchwood used for fuel. Matthews, et al. found that while burning harvested wood for fuel only results in a slight decrease in absolute CO_2 emissions when substitution for fossil fuels is taken into account, the GHG impact is less beneficial than ceasing harvesting. These results show that use of wood primarily for long-lived products is far more GHG beneficial than 1) using wood for primarily for fuel, or 2) in some scenarios, ceasing harvesting and manufacturing products from nonwood alternatives.

Matthews, et al. also analyzed comparative GHG impacts of restoring harvest to broad-leafed forests planted in the 1950's with the intention of harvesting, but left unmanaged (see Figure 2). Matthews, et al.'s results indicate that under certain circumstances, initiating management and harvesting in these forests can be GHG beneficial, but that GHG reductions depend heavily on the type of long-lived products produced from the harvested wood.

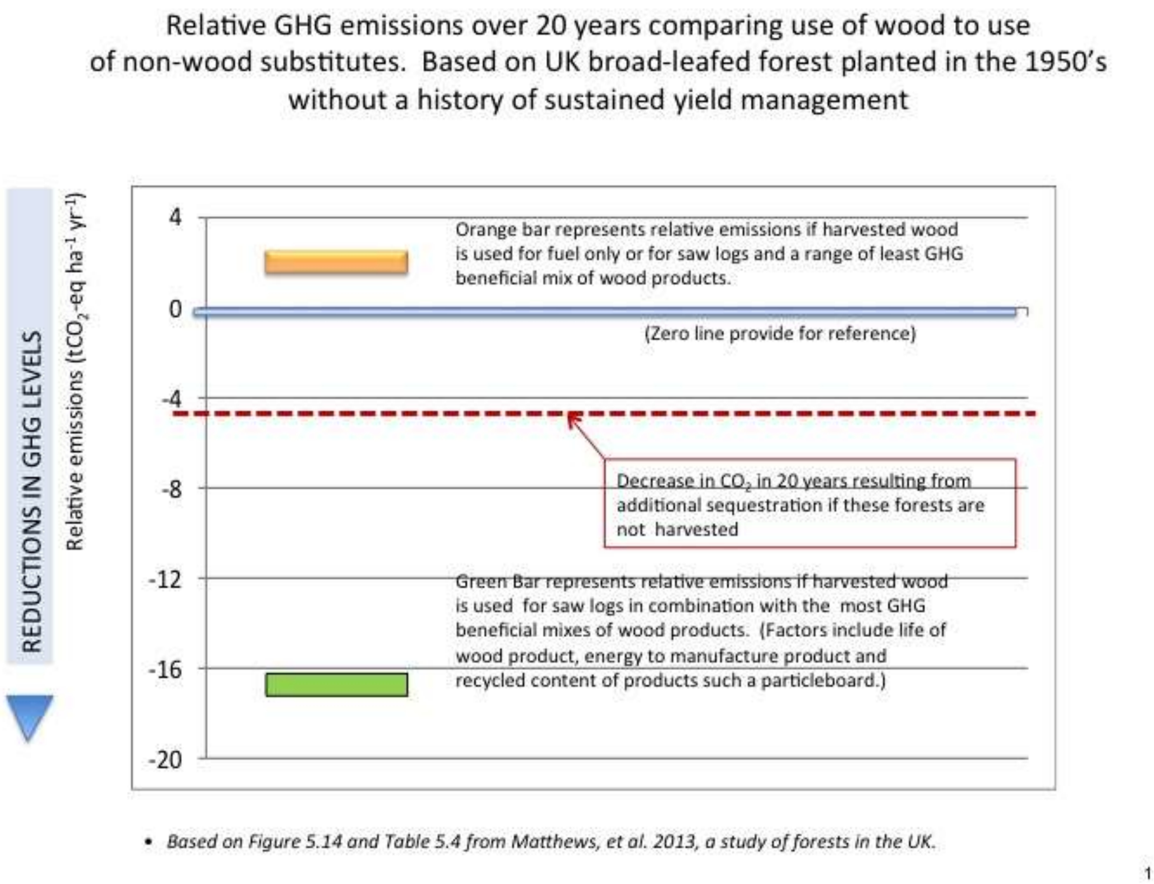

Figure 2. Relative GHG emissions over 20 years comparing use of wood to use of non-wood substitutes. Based on UK broad-leafed forest planted in the 1950's without a history of sustained yield management

Matthews, et al. found the greatest GHG benefits to result from use of harvested wood for sawlogs used for timber; sawlog offcuts and small roundwood used for particleboard (also using 70% recycled wood); and bark and 50% of branchwood used for fuel. If wood is used for fuel only or a less beneficial combination of products, GHG impacts would be less beneficial in 20 years than allowing these forests to continue without active management.

Although the Matthews study concerns forest management and wood product use in the UK, it is possible to extrapolate its results to New England. The results of the Matthews study suggest that careful management and use of harvested wood for timber and other long-lived products, with use of forest residues for fuel, can be GHG beneficial in the near term, especially if combined with efficient wood processing and use of recycled wood. The results of the Matthews study for a managed softwood forest are relevant to some working forests in northern New England, while the results concerning broad-leafed forests are more applicable to other parts of New England. It would be extremely useful to replicate the methodology of the Matthews study based on New England forests. Such a study could also incorporate an analysis of albedo for Northern New England (see next section).

VII. Comparing Mass Timber Buildings to Equivalent Concrete and Steel Buildings

Results of studies of energy and GHG impacts of building with traditional wood framing should not be assumed to apply to mass timber buildings. The CLT Handbook (Karacabeyli and Douglas 2013) emphasizes that construction with CLT has different attributes than light wood construction. The authors note that CLT construction uses more wood and requires more energy-intensive manufacturing than traditional wood-framed construction, but that CLT construction provides more potential opportunity to store carbon. As with construction with traditional wood framings, this potential benefit must be weighed in light of the climate impacts of harvesting, and evaluated in light of the time horizon of interest.

There are relatively few LCA studies comparing mass timber buildings with equivalent concrete and steel buildings. Two North American studies comparing climate impacts of CLT buildings and equivalent concrete and steel buildings are of interest. Robertson, et al. (2012) performed a cradle to gate[21] LCA comparing an existing 5-story CLT office building (Discovery Place) built in 2009 on Vancouver, BC with an equivalent concrete and steel building. The authors applied the AIE and also calculated total process emissions. The authors found that process emissions for a CLT building were almost equivalent to the process emissions for an equivalent building constructed with reinforced concrete, but the percentage of process emissions supplied by biomass energy was much higher for the CLT than the reinforced concrete building. As a result, the CLT building had a lower GWP than the concrete and steel building under the AIE, which does not include biomass energy in calculation of GWP. The authors pointed out that their study did not take into account the shorter construction time for the CLT building, the potential for recycling CLT, and the potential inefficiency of the small pilot CLT manufacturing facility the provided data for CLT manufacturing emissions. The authors noted that the carbon storage in the building was of significance in the Pacific Northwest, which is experiencing the worst Mountain Pine Beetle outbreak in recorded history, causing the forests of western Canada to become net emitters of GHG. The outbreak is expected to kill one billion cubic meters of standing wood in BC alone and more throughout the western US. This dynamic makes CLT carbon storage potentially very beneficial in this region and other regions with catastrophic die-offs or affected by insects infestation (including spruce budworm), extending carbon storage for decades from trees that would otherwise decompose or burn.

Grann (2013), as is noted above, conducted a more comprehensive cradle to grave LCA, comparing the climate impact of a 24 unit, 4-story CLT apartment building with an equivalent concrete and steel building. The buildings were modeled for construction in Quebec City to be constructed with wood harvested Chibougamou, in Quebec Province. (The CLT building design was based on a CLT building actually constructed in Chibougamou, but the LCA sited the building in Quebec City, the nearest location for which AIE had data.) This study evaluates the climate and other environmental impacts over a 100-year time horizon. Grann applied AIE, which applies US EPA's TRACI tool for the assessment of environmental impact (see Section IV, above for description of the AIE and TRACI). Grann also included an analysis of biogenic carbon fluxes, including analysis of effects of harvest in the forest, the effects albedo change in the forest during the decades immediately after harvest, and the impact of carbonation of concrete in both structures over time. The study assumed a 60-year life span for both buildings,

[21] Like other cradle to gate studies, this study did not include impacts on the forest, construction emissions, or GHG impacts of demolition, landfilling, or incineration at end of life.

with landfilling thereafter. Grann also performed sensitivity analyses of 50% recycling of CLT and end-of-life incineration of CLT.

In analyzing forest carbon impacts, Grann applied the Carbon Budget Model of the Canadian Forest Sector based on growth and yield curves for black spruce in Quebec that was assumed to be carbon neutral over a 90 year rotation. Grann did not, however, consider biogenic emissions to be carbon neutral, but instead applied the approach put forward by Cherubini, et al. (2012). Grann assumed no recovery of forest residues for fuel use. Grann found:

> For every kg of CO_2 in harvested round wood removed from the harvest site roughly 0.81 KG of CO_2 is released from harvest site to the atmosphere during the first 12 years. From year 13–45, the initial flux of CO_2 from the harvested site is resequestered by regrowth of the forest, and from years 46-90, an additional one KG of CO_2 equivalent to the CO_2 in the harvested roundwood removed from harvest site is removed from the atmosphere through additional tree regrowth (Grann 2013, p. 40).

Grann applied the methodology put forward in Cherubini, et al. (2012) for evaluating the combined effects of harvesting on forest carbon and albedo. In climates with significant winter snow cover, surface reflectivity during winter months is increased for a period after harvest, reducing "radiative forcing" of the earth. The effects of albedo are significant at the harvest location site, which is at approximately 50 degrees latitude and is snow covered over many months of the year.

Grann (2013) analyzed carbonation of concrete in both the CLT building and the concrete and steel building. He concluded that for the concrete and steel building modeled, approximately 10% of CO_2 emitted in the manufacture of concrete in the building would be reabsorbed in the first 20 years, approximately 20% of CO_2 could be expected to be reabsorbed up to the end of its 60-year building life, reabsorption of approximately 40% additional CO_2 would take place in rapidly after demolition if the concrete is crushed for filler, ultimately resulting in a total of approximately 70% of CO_2 emissions at the end of 100 years (Grann, p. 49, Figure 18). (Grann noted that the crushing process itself produced some emissions.) Concrete components in the CLT building also reabsorbed CO_2, but carbonation had less effect in the CLT building because less concrete was used.

Grann found that over a 100 year time frame, although the primary energy consumption[22] associated with building the CLT building was higher (8.06 terajoules for the CLT building vs. 7.49 terajoules for the concrete and steel building),[23] when GWP is determined and adjusted to include biogenic impacts, carbonation and albedo, the CLT building has an adjusted net GWP over 100 years that is lower than the equivalent concrete and steel building, and is lowest when 50% of CLT panels are reused. Grann (2013, figure 48).

Grann reported the following numbers for GWP and net GWP adjusted for biogenic carbon, carbonation, and albedo:

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[22] It should be noted that primary energy consumption is limited to the energy from resource extraction, material production and energy used in the construction process, and is a different measure than GWP or net GWP over 100 years.

[23] The actual CLT building analyzed in this study used significant quantities of energy-intensive rockwool in floor separations for acoustic performance. Grann noted that the LCA study highlighted the opportunity to use of alternative materials or technology innovations to reduce or substitute for rockwool to improve the environmental performance of CLT buildings (Grann 2013, p. xii).

Table 2. Global warming potential and net global warming potential over 100 years for CLT and concrete and steel buildings, with landfilling of CLT at end of building life (measured in kg CO_2e)

	AIE GWP*	Biogenic Carbon**	Carbonation	Albedo	Net GWP
CLT buildings	440,366	105,333	*-6,377*	*-499,508*	39,812
Concrete and steel buildings	772,388	2,262	*-77,890*	*-10,630*	636,130

* AIE Global Warming Potential does not include biogenic carbon fluxes or emissions from biomass, carbonation, or albedo effects.

**Biogenic Carbon includes: carbon emissions at the harvest site and from biofuels used in manufacture of wood products for the building, plus emissions from the landfill, offset by carbon sequestration at the harvest site over 100 years.

Source: Grann (2013), Table 2, page v, updated by personal communication with author April 16, 2014

Grann's analysis finds that over 100 year timeframe, the Quebec City CLT building would have a significantly lower net GWP than an equivalent concrete and steel building, spectacularly so when albedo change at the harvest site is taken into account. An alternative scenario of recycling 50% of CLT produces even lower net GWP for the CLT building over 100 years, such that net GWP would dip below zero. If albedo were not considered, the CLT would have approximately 20% lower net GWP over 100 years than the concrete and steel building.

This study demonstrates the sensitivity of LCA results including forest carbon fluxes and albedo changes at the harvest site. This study also demonstrates that there is significant potential to maximize substitution benefits over a 100-year timeframe through optimization of CLT building design, choice harvest location and approach, use of wood residues for fuel, CLT recycling, and effective management of wood materials at the end of their useful life. The study suggests that albedo may be an especially important variable in such analyses in areas of New England with significant, persistent snow cover.

Although Grann looked at some impacts over a 20 years, the study does not report net, adjusted GWP impacts of the CLT and concrete and steel buildings over this shorter timeframe. As albedo has such a significant effect at this far northern site, it appears that this building would have a GHG benefit or be neutral during a 20 to 30 year period,[24] but further analysis of underlying data would be required to reach this conclusion. Analyses of mass timber buildings that provide information about 20 or 30 year GHG impacts will

[24] Grann reports that in 20 years, biogenic C will be 925,000 kg CO_2e and albedo counter-effects will be minus 1,210.000 kg CO_2e (Table 65, page 90). Albedo effects are therefore estimated to be larger than biogenic emissions during this initial 20-year period, when biogenic carbon emissions are highest from both decaying residues at the harvest site and from manufacture of CLT and other building materials. (It is also the period when albedo has the largest counter effects.) It is not possible to say whether total albedo effect in 20 years would exceed the larger total energy required to build the CLT building, but it appears likely that it would. This would make it likely that the CLT building would be GHG beneficial in this shorter terms.

clearly be very important in the future, as the focus of policy-makers continues to intensify on near-term GWP impacts of various measures.

It should be noted that mass timber design and use in the US is in its infancy. CLT and EWP technology used in mass timber systems will continue to evolve. Moreover, less energy-intensive mass timber technologies may soon be available. The University of Maine recently received a grant to evaluate the potential of using LSL in mass timber applications.[25] LSL is now being produced in Maine in a retooled mill set up for LSL manufacturing. Although an LCA has not been released for LSL at the time of this report, the production of LSL is expected to require less energy than CLT (Dr. Stephen Shaler, Director of the School of Forest Resources, University of Maine, pers. comm., April 2, 2014.). Moreover, these evolving EWP technologies and uses will provide uses for trees that otherwise have little economic value and may therefore be part of a broader forest-management strategy. LSL uses a formula of 85% aspen and 15% mixed low quality hardwoods.

VIII. Summary Concerning Climate Impacts of Building with Wood as Compared to Concrete and Steel

A. Traditional Wood Construction and GHG benefits

We have sufficient information to determine that **there is a significant opportunity to gain GHG benefits over a 100-year timeframe by substituting traditional wood-frame construction for concrete and steel in non-residential buildings in New England**. Based on the information concerning the immediate opportunity to use more wood in buildings set out in Section II, there will be an immediate opportunity to increase the use of wood in construction, upon economic recovery, by an additional 402,940,000 bfe (approximately 805,880 cords). (This is the total of 338,190,000 board feet substituting wood for concrete and steel construction in non-residential construction, and 64,750,000 bfe in residential construction floors and walls. Based on the displacement factor provided by Sarthe and O'Connor (2010), this would yield an **annual CO_2e savings of nearly 3.5 million metric tons of CO_2e** (based primarily on studies with longer term horizons). Based on the EPA Greenhouse Gas Equivalencies Calculator, this is the equivalent of annual emissions from nearly three quarters of a million cars. There are additional opportunities, although less quantifiable at this time, of substituting wood in buildings in which "mass timber" systems can be used.

It is more difficult to reach conclusions about the GHG impacts of building with traditional wood construction over the shorter 20 to 30 year time horizon that is of increasing concern to policy-makers. LCAs comparing GHG impacts of buildings do not generally evaluate comparative GHG impacts over these shorter time horizons. **Extrapolating from the Matthews study, discussed in Section VI, it appears that traditional wood construction using timber and other wood products from New England's sustainably managed softwood forests is likely to have GHG benefits in this short time frame**. These benefits could be more significant when wood is harvested in areas with persistent snow cover, where surface reflectivity will be increased for a period after harvest. It is also possible that efficient use of wood products from hardwood forests can have GHG benefits in this shorter time horizon, with highly efficient manufacture and use of wood products. Moreover, use of wood for long-lived products from trees that are dying because of insect damage (e.g., spruce budworm,

[25] In the same grant, the researchers will be evaluating the potential to use Norway spruce for CLT.

hemlock and balsam woolly adelgid, emerald ash borer) or disease (e.g., beech bark disease) can always be presumed to be GHG beneficial in the shorter time horizon, as the long-loved products store carbon that would otherwise be released into the atmosphere. This applies to both wood-framed construction and use of wood for construction components, such as hardwood flooring.

B. Mass Timber Construction and GHG impacts

Mass timber designs using advanced EWPs, including but not limited to CLT, are expanding the opportunity to use wood in construction of low-, mid- and high-rise buildings where wood could not previously have been used. Mass timber designs use more wood and require more total processing energy than traditional wood frame construction. The few available LCAs of North American CLT buildings have found that with today's CLT manufacturing efficiencies, **the total energy required to build CLT buildings can be equivalent to the energy needed to build a comparable concrete and steel building**. In a longer time horizon (100 years), regrowth in a sustainably harvested forest will recapture CO_2 over time, offsetting a significant portions of emissions from manufacture and construction. Extrapolating from the Brann study discussed in this report, it appears that if wood from the CLT building is disposed of responsibly in well-managed landfills or incinerated to substitute for fossil fuels, **these buildings will have a long-term GHG benefit over building with concrete and steel**, even taking into account concrete's reabsorption of CO_2 through the carbonation process. Recycling CLT will provide additional long-term GHG benefits.

There is a question of whether and when building with mass timber designs will increase GHG emissions in the short term before providing longer-term GHG benefits. While more study of this question is required, it appears that this could be the case in some circumstances. On the other hand, **the use of wood in mass timber buildings provides an opportunity to store carbon from forests where trees are dying from insect infestation or other causes, storing carbon that would otherwise be released** into the atmosphere. (It is no surprise that interest in CLT is especially strong in western Canada, where forests are dying from an unprecedented infestation by Mountain Pine Beetle.) It is also possible that mass timber buildings could provide benefits in shorter time horizons if wood is harvested in locations with persistent snow cover and in a manner that increases surface reflectivity in the winters. Again, more study is required.

As is noted in the report, mass timber technologies and designs are relatively new in North America. New, less energy intensive EWPs, evolving mass timber designs, and emphasis on recycling are likely to reduce GHG impacts of mass timber designs in the coming decades.

C. Cascading Uses

Studies make it clear that the **greatest GHG benefits of using wood arise from "cascading use," using wood in long-lived forms first and repeatedly if possible, then using the wood for short-live products or biofuels, or by placing wood at the end of multiple uses in landfills with efficient handling of the limited methane produced by wood product**. The cement and steel industries are very active in promoting reuse and recycling of materials upon demolition of buildings, while the recycle rate for wood from buildings is lower. Optimizing the GHG benefits from long-lived wood products could be enhanced by partnerships with municipalities, communities and the construction industry to encourage reuse and recycling of wood where possible, and when not possible, to ensure responsible use of wood in biofuel programs or disposal of wood in well-managed landfills.

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X. Appendix A. Solid Wood in Landfills – GHG Emissions and Carbon Storage

Assumptions about the release of carbon at the end of a wood products' useful life significantly affect the results of analyses of the GHG benefits of substituting wood products for other materials. Research on this topic has evolved over the past few decades.

In considering GHG consequences of wood in landfills, four variables are important:

- The amount of GHG released from wood products in landfills and – the other side of the same coin – the amount of carbon reabsorbed by soils or permanently stored in the landfill;
- The nature of the GHG release – whether it is in the form of CO_2 or methane, which is 20 to 25 CO_{2e} greater global warming potential in 100 years and 72 times CO_{2e} global warming potential in 20 years;
- The timing of the GHG release (the question of timing of release is significant both because of concern over the timing of releases of GH's on climate systems, and because timing can effect efficiency of methane-capture systems);
- Whether and how much methane resulting from decomposition of wood products in landfills is captured for fuel use, fossil fuel substitution benefits.

In order to understand the literature on the GHG consequences of disposing of wood in landfills, it is useful have an understanding of landfills generally.

In the developing world and in some developed countries, including the US, it is common to dispose of solid waste by landfilling (Laner, et al. 2012, Matthews 2012a).

Waste that goes into landfills in the US is categorized as municipal solid waste (MSW) (food waste, paper products, yard waste and trimmings, and other household and commercial trash) or C&D waste. The EPA reported that in the US in 2012, the US produced 251 million tons of MSW (US EPA 2014b). Approximately 54% of this waste (136 million tons) was discarded into landfills; approximately 12% (29 million tons) was incinerated for energy recovery; approximately 26% (over 65 million tons) was recycled; and approximately 8% (21 million tons) was composted.

For wood products, the following quantifies were recycled or composted on one hand or landfilled on the other:

	Total MSW (million tons)	**Land-Filled (million tons)**	**Recycled or composted (million tons)**	**Recovery rate (recycling and composting)**
Solid wood	16	13	2	15%
Yard trimmings	34	14	20	58%
Paper & paperboard	69	24	44	65%

Source: US EPA (2014b).

US EPA reported that post recycling and composting, wood products made up the following percentages by weight of materials deposited in landfills in 2012:

Wood:	8.2%
Yard Trimmings:	8.7%
Paper and paperboard:	14.8%

EPA does not keep national statistics on C&D waste. Falk and McKeever (2012) estimated that in 2010, 130 million tons of C&D waste was generated in the US, containing approximately 36.4 million tons of wood, of which 80% was from demolition and 20% from construction. The percentage of wood in C&D waste varies by region, ranging from approximately 25% to 55%. (Staley and Barlaz 2009), averaging approximately 40% for C&D waste. Falk and McKeever (2012) estimated that 17.3 million tons of this C&D wood waste was recoverable. This volume or somewhat more[26] presumably was taken to landfills or specialized disposal sites.

Landfills in the US and other developed countries are heavily regulated. In the US, regulations require the following:

- **Location restrictions.** Ensure that landfills are built in suitable geological areas away from faults, wetlands, flood plains, or other restricted areas.
- **Composite liners requirements.** Include a flexible membrane (geomembrane) overlaying two feet of compacted clay soil lining the bottom and sides of the landfill, protect groundwater and the underlying soil from leachate releases.
- **Leachate collection and removal systems.** Sit on top of the composite liner and removes leachate from the landfill for treatment and disposal.
- **Operating practices.** Include compacting and covering waste frequently with several inches of soil help reduce odor; control litter, insects, and rodents; and protect public health.
- **Groundwater monitoring requirements.** Requires testing groundwater wells to determine whether waste materials have escaped from the landfill.
- **Closure and post closure care requirements.** Include covering landfills and providing long-term care of closed landfills.
- **Corrective action provisions.** Control and clean up landfill releases and achieves groundwater protection standards.
- **Financial assurance.** Provides funding for environmental protection during and after landfill closure (i.e., closure and postclosure care).

(Source: www.epa.gov/osw/nonhaz/municipal/landfill.htm, accessed April 5, 2014)

Regulations require that landfills continue to be actively monitored and must continue collecting leachate for a minimum of 30 years after closure.

The decomposable materials in landfills are the organic materials – plant and animal products. Other materials in landfills may affect the chemistry of decomposition, but are not themselves decomposable. Carbon that enters landfills can be emitted as CO_2, emitted as methane, emitted as a volatile organic compound, become dissolved in leachate, or remain stored in the landfill (US EPA 2012, p. 2).

[26] Falk and McKeever (2012) estimated that of 36.4 million tons of wood, 19.1 million tons were recovered, combusted, or “not usable.” It is unclear whether some of the “not usable” wood was placed in landfills. Also, some percentage is placed in designated C&D disposal sites. Falk and McKeever also noted that wood diversion has not yet reached the rates of recycling report for steel (98%) or concrete (82%).

Decomposition in landfills, by contrast to decomposition of materials in the woods or composting systems, is largely anaerobic, meaning that decomposition takes place in the absence of oxygen. If certain conditions are met, this produces a combination of methane and CO_2 (the two primary components of "landfill gas"), as well as other chemicals in smaller quantities.

When refuse is initially buried, oxygen in the refuse is consumed, typically over a few days, producing CO_2 and an increase in temperature from waste heat of aerobic metabolism. When oxygen is depleted, anaerobic fermentation results in an accumulation of carboxylic acids, causing a decrease in pH. In the third phase, methanogenic bacteria convert accumulated acids into methane and CO_2. In this phase, cellulose and hemicelluluse decomposition begins. In later phase, methane production becomes stable and then decelerates until methane production ceases (Barlaz 2004, , p. 6-7). Methane escapes from many landfills, which are the third largest source[27] of methane emissions in the US (US EPA 2013).

The rate and extent of decomposition in landfills is affected most by moisture level. EPA divides landfills into two broad categories: "traditional" landfills and "bioreactors." In dry tomb landfills, waste is buried and kept as dry as possible to limit the amount of leachate formed as moisture percolates through the waste collecting decomposition intermediates byproducts and contaminants (Micales and Skog 1997). Landfills collect the leachate so that it does not contaminate soil and groundwater, and vent, flare or collect landfill gas. EPA estimates that decomposition rate in these landfills in different categories based on amount of annual rainfall, which affects decomposition rate.

In bioreactor landfills, the goal is to accelerate decomposition, usually to produce and use landfill gas as a fuel. There are several types of bioreactor landfills, according to the EPA (US EPA 2014c):

- **Aerobic.** In an aerobic bioreactor landfill, leachate is removed from the bottom layer, piped to liquids storage tanks, and recirculated into the landfill in a controlled manner. Air is injected into the waste mass, using vertical or horizontal wells, to promote aerobic activity and accelerate waste stabilization.
- **Anaerobic.** In an anaerobic bioreactor landfill, moisture is added to the waste mass in the form of recirculated leachate and other sources to obtain optimal moisture levels. Biodegradation occurs in the absence of oxygen (anaerobically) and produces landfill gas. Landfill gas, primarily methane, can be captured to minimize GHG emissions and for energy projects.
- **Hybrid (Aerobic-Anaerobic)**. The hybrid bioreactor landfill accelerates waste degradation by employing a sequential aerobic-anaerobic treatment to rapidly degrade organics in the upper sections of the landfill and collect gas from lower sections. Operation as a hybrid results in the earlier onset of methanogenesis compared to aerobic landfills.

[27] Many studies of landfills cite earlier EPA reports that landfills are the second largest source of methane emissions. The EPA reported that by 2011, however, methane emissions from petroleum and gas systems had overtaken both agriculture and landfills to become the largest methane source.

Numerous advantages have been attributed to the operation of a landfill as a bioreactor including: (1) more rapid settlement maximizes airspace utilization and ultimately reduces land requirements for waste disposal; (2) higher rates of gas production improve the feasibility of recovering the gas for beneficial use; (3) in-situ leachate treatment as degradable organic matter in the leachate is consumed within the waste mass, thus reducing reliance of off-site or external wastewater treatment facilities; and (4) the enhanced rate of decomposition may reduce the time and intensity of post-closure monitoring and maintenance. While the major focus of bioreactor landfills has historically been on accelerated settlement and gas generation, reductions in the need for off-site leachate treatment are becoming more important as landfill owners in some regions are facing limits on leachate acceptance from publically owned wastewater treatment facilities (POTWs). (M. Barlaz, personal communication, 4/7/2014.)

Even at bio-reactor landfills with methane capture, however, methane escapes before and after methane levels are reached and sustain levels at which methane can be captured for fuel.

In 2011, the EPA reported on data from 1990-2009 the following breakdown of the fate of percentage of methane produced by landfills in the US: 28% was released from landfills with no landfill gas recovery; 38% was produced by landfills where landfill gas was recovered and flared; and 34% was produced by landfills with landfill gas recovery for energy (US EPA 2012, p. 1). This does not mean, however, that all methane was captured in the second two categories of landfills, as the capture rate at these landfills is not complete. EPA estimates that where methane is collected, capture percentages of methane for lumber in landfills range from 50% to 91% (US EPA 2012, p. 12; M. Barlaz, personal communication, 4/7/2014).

The primary areas of research relevant to the issue of emissions from landfills are: 1) the rate at which various waste products decompose ("decay rate");[28] 2) the efficiency of capture of methane from landfills, 3) is the methane released, flared (converting emissions to CO_2) or captured for fuel; 4) and the uses of captured methane fuel and what other fuel it displaces.

In recent decades, researchers have begun to quantify the potential for landfills to function as permanent carbon sinks. This is a function of the extent to which a material releases carbon upon completion of decomposition. The fraction of carbon that remains in the landfill is considered to be functionally permanently stored. The fraction that is not released is termed the material's "carbon storage factor." Different types of materials have significantly different percentages of carbon storage.

In WARM version 12, the US EPA (2012) summarizing the results of Eleazer, et al. (1997) and Barlaz (1998), reported the following carbon storage percentages for various materials:

Material	Percent of Carbon Stored
Corrugated containers	55
Newspaper	85
Office Paper	12
Coated Paper	75
Food Scraps	16

[28] Decay rate is important, both as it relates to the timing of emissions, and because of timing of capture of methane. Because methane capture is generally not put into place immediately, methane capture is less efficient for rapidly decomposing materials such as food wastes (WARM 2012, p. 12).

Grass	53
Leaves	85
Branches	77
Mixed MSW	19
Gypsum Board	64

A. Solid Wood in Landfills

Anaerobic decomposition of solid wood and some wood products is known to progress more slowly than decomposition of most other organic materials because of the high lignin content in wood. Cell walls in plant materials are made up of cellulose, hemicellulose, and lignin. Cellulose is relatively simple in structure. Hemicelluloses are branched polymers that bind bundles of cellulose fibrils to form microfibrils, helping stabilize cell walls. They also cross-link with lignin, creating a complex web of bonds that provide structural strength and also challenge microbial degradation (Ladisch, et al. 1983; Lynch 1992).

Lignins are a class of highly complex organic polymers. Lignin in wood is highly resistant to decomposition and inhibits decomposition of cellulose and hemicellulose in anaerobic settings. Lignin in wood products interferes significantly with degradation of cellulose and hemicellulose degradation both chemically and mechanically (Micales and Skog 1997).[29] As a result, carbon from all of the lignin and some of the cellulose and hemicellulose from wood products remains in permanent storage in landfills. This percentage of carbon in a material that does not decompose and is therefore stored is its "carbon storage factor" (Wang, et al. 2011, p. 6868).

A number of studies have been conducted on the rate and extent of decomposition of wood in landfills. Studies have approached the question from a variety of different perspectives, including: 1) comparative decay of solid wood, other wood products such as paper, and other MSW materials; 2) decay rates as related to landfill gas production efficiencies; and 3) rate of carbon sequestration potential of wood products in landfills. This last focus is of the most interest here, as it reflects the carbon that remains locked in landfills after the decomposition process is complete.

Studies fall into two categories: 1) laboratory studies simulating conditions in landfills; and 2) excavation of materials from landfills.

A few carefully controlled studies of decomposition of materials in landfill conditions conducted in the laboratory of Dr. Morton Barlaz at North Carolina State University have formed the basis of much of the US literature analyzing decomposition of wood in landfills, including the results cited above from US EPA's WARM Version 12 document (2012). Eleazer, et al., (1997) analyzed carbon conversion percentages of a range of materials in laboratory conditions simulating decomposition in conditions in bio-reactor landfills concluding that 13.6% of carbon in branches decomposed (Wang, et al. 2011, p. 6868, Eleazer, et al. 1997). This finding has been used as a proxy for carbon conversation percentages of wood products generally in several subsequent studies (see, e.g., Staley and Barlaz 2009).

[29] The variability of degradability of paper products is related to it lignin content. Newspaper is high in lignin and degrades significantly more slowly than other paper products such as office paper, from which most lignin is removed in the manufacturing process.

A recent study from the same laboratory has raised questions as to whether the 13.6% carbon conversion for branches found in Eleazer, et al. (1997) is broadly applicable to solid wood products. Wang, et al. (2011) conducted a targeted laboratory study of decomposition of different types of wood in landfills. In this study the researchers evaluated carbon conversion percentages of major wood products in laboratory conditions designed to obtain the maximum decompositions and resulting landfill gas yields in bioreactor landfills. Triplicate reactors were set up to obtain results for each of different types of wood products. The types of hardwood and softwood products and percent of carbon converted are set out in the table below. The researchers also measured carbon remaining after decomposition was completed, which revealed that except in the hardwood products, very little of the cellulose and hemicellulose was converted via decomposition.

Wood Product	Carbon Converted (%)	Standard Deviation
Hardwood – red oak	8.0	(3.3)
Hardwood - eucalyptus	0.0	(0.0)
Softwood - spruce	1.8	(0.2)
Softwood – radiate pine	0.1	(0.1)
Plywood	1.4	(1.1)
Oriented Strand Board –Hardwood	20.9	(2.2)
Oriented Strand Board –Softwood	0.0	(0.0)
Particle Board	1.3	(0.8)
Medium Density Fiberboard (MDF)	1.1	(0.2)

Adapted from Table 2, Wang, et al. (2011), results of tests not completed at time of publication (red oak and oriented strand board) provided by M. Barlaz (personal communication 4/7/2014.

The researchers made several observations about these results. They addressed the findings that red oak and oriented strand board (OSB) manufactured with hardwood had a higher carbon conversion rate than softwood products, including OSB manufactured with softwood. They noted that in general, hardwoods have a lower lignin content than softwoods and that hardwood lignin polymers have somewhat lower structural integrity than softwood lignin and are somewhat more susceptible to chemical degradation. The researchers explained that while eucalyptus is a hardwood, the toxins from eucalyptus inhibited methane production.

The researchers discussed the effects of petroleum-based adhesives in the EWPs. They found that the presence of the adhesives did not significantly affect carbon conversion of the wood products. They found that urea formaldehyde in medium density fiberboard and particleboard did not itself show methane-producing potential, but resulted in release of elevated levels of ammonia. They found that the phenol formaldehyde adhesive in OSB and plywood exhibited some methane producing potential, but that the overall contribution to methane yield of these products was not significant.

The results in the table above show that Wang, et al. found carbon conversation rates to be less than 2% for most softwood products, approximately 8% for red oak, and approximately 20% for OSB made from hardwood. The researchers found no explanation for the difference in carbon conversion percentage for branches (13.6%) found in Eleazer, et al. (1997) and the results in their study. Wang, et al. indicated that the comparative results of the two studies and the varying carbon conversion percentages for different wood products found in their study indicated that it may not be appropriate to apply a single carbon conversion percentage to wood products as a whole. They noted that all results indicate that carbon conversion of wood in landfills is far lower than the 50% carbon conversion rate applied by the IPCC for organic materials in landfills (Wang, et. al 2011).

While wood products produce relatively little methane in landfills, because methane is such a powerful GHG, it is critical to capture as much of this methane as possible, for fuel use or flaring.   The best approach is to keep wood products in service through the cascading uses as long-lived wood products that can be recycled for maximum life spans, and then either dispose of the products in a well-managed landfill or use it for biofuel when it can be burned efficiently and cleanly.