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Accounting for forest carbon pool dynamics in product carbon footprints: Challenges and opportunities

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ABSTRACT

Modification and loss of forests due to natural and anthropogenic disturbance contribute an estimated 20% of annual greenhouse gas (GHG) emissions worldwide. Although forest carbon pool modeling rarely suggests a 'carbon neutral' flux profile, the life cycle assessment community and associated product carbon footprint protocols have struggled to account for the GHG emissions associated with forestry, specifically, and land use generally. Principally, this is due to underdeveloped linkages between life cycle inventory (LCI) modeling for wood and forest carbon modeling for a full range of forest types and harvest practices, as well as a lack of transparency in globalized forest supply chains. In this paper, through a comparative study of U.S. and Chinese coated freesheet paper, we develop the initial foundations for a methodology that rescales IPCC methods from the national to the product level, with reference to the approaches in three international product carbon footprint protocols. Due to differences in geographic origin of the wood fiber, the results for two scenarios are highly divergent. This suggests that both wood LCI models and the protocols need further development to capture the range of spatial and temporal dimensions for supply chains (and the associated land use change and modification) for specific product systems. The paper concludes by outlining opportunities to measure and reduce uncertainty in accounting for net emissions of biogenic carbon from forestland, where timber is harvested for consumer products.

1. Introduction

Forests cover approximately 65% of the total land surface and play a vital, yet complex role in the global carbon cycle. Holding 90% of the plant biomass carbon and 80% of soil carbon found in all terrestrial ecosystems, they also assimilate 67% of the total carbon dioxide removed from the atmosphere by these ecosystems (Landsberg and Gower, 1997). Annual loss of forests due to disturbance (harvesting, conversion, fire, insects, pathogens, and wind) contributes as much as 20% of total global greenhouse gas (GHG) emissions each year—rivaling emissions from the global transportation sector (Denman et al., 2007).

Gower (2003) divides the carbon cycle into two interrelated phases: 1) initial disturbance effects on carbon pools and 2) changes in carbon cycle processes during forest ecosystem recovery or succession. These can be 'natural' disturbance events (e.g., fire, pest outbreak, etc.) or anthropogenic events (e.g., timber harvest, road construction, mining, etc.).

The life cycle assessment (LCA) community and associated guidance methodologies, such as product carbon label protocols, have struggled to adequately account for GHG emissions associated with direct and indirect land use change in consumer products made from forest resources. A major reason for this is the spatial complexity of land use, as geography and industrial practices vary widely. Advances in terrestrial carbon modeling reveal a complex dynamic of the carbon pools in forests, but rarely a 'carbon neutral' flux profile as has been modeled traditionally in LCA (Johnson, 2009). The work by Searchinger et al. (2008) reminds us of the potential for 'accounting' error if biogenic carbon emissions related to land use are ignored.

Starting in 2008, procedures and methodologies to account for direct land use change were developed in several major product carbon footprint protocols. Within these methodologies, land use modification, where the category of the land (e.g., forest) remains the same, but the carbon pools in that land have been fundamentally disturbed, is also at issue (viz., 'old-growth' forest converted into a working forest, with short-term harvest rotations and intensive cultivation practices, including management of plantation seedlings).

The IPCC has issued a well-developed methodology for calculating emissions in the same land use category (e.g., forestland remaining forestland) and for shifts from one land use category to another (e.g. forestland to cropland), but this guidance is designed for GHG inventories at the national level, not the product level (IPCC, 2006).

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Protocols recently issued by the World Resources Institute (WRI), the International Organization for Standardization (i.e. ISO Draft International Standard 14067), and a consortium led by Carbon Trust (i.e., PAS 2050) are essentially trying to adapt IPCC guidelines for products and are advancing efforts to incorporate the effects of direct land use change and land use modification. But the instructions require further development, perhaps in supplemental guidelines, to handle a number of critical issues related to forest carbon pools and products made from forest resources.

As we point out in this paper, a major reason for this slow development stems from underdeveloped linkages between two academic communities—forestry carbon modelers and life cycle inventory (LCI) modelers. In both communities, however, a growing number of researchers are aware of the need to bridge this gap. For example, Gower (2003) calls for integrated research to model the complex interrelationship between the biological carbon cycle (i.e., forest ecosystem) and the industrial carbon cycle (i.e., forest products).

A second major reason stems from the increasingly globalized nature of our product supply chains, which can involve a dizzying array of producer–retailer–wholesaler–broker arrangements spanning multiple continents. Unweaving these spatially diverse and often temporally contingent arrangements to collect data for forests in specific supply chains can be a time-consuming, difficult task. But such practical difficulties do not obviate the fact that some consumer wood products made in some supply chains, for example, may carry a very high share of the climate change burden created by disturbance to forest carbon pools. Others may carry far less depending on their specific geographies and production processes.

The production and consumption of paper take place in these sorts of complex and extended globalized supply chains, and account for about 40% of the world's industrial wood harvest, rising to about 50% by 2050 (Abramovitz and Mattoon, 1999). Multinational suppliers manage a dynamic web of fiber sourcing, pulping, paper production and converting operations all over the world. The world's two largest producers of paper are the U.S. and China.

Since 1990, China has accounted for more than 50% of the world's overall growth in paper and paperboard (Barr and Demawan, 2005). But the supply chain geographies of these two industries are very different. The Chinese paper sector once consisted of many small-scale pulp mills dotted across the local landscape and heavily reliant on non-wood fiber, but is now dominated by huge modern mills that import large quantities of pulp and wastepaper from Siberia to Indonesia (Sun and Canby, 2010). Despite subsidies and land allocated for plantations, China's industry will rely heavily on imported pulp for decades to come (Barr et al., 2005). In the U.S., there is far more domestic supply, limiting the need for sourcing high-quality pulp from around the globe.

In the comparative case study presented here, we seek to understand how the geography of global supply chains for coated freesheet paper may vary, and the implications of this variation for measuring the product carbon footprint. Specifically, we explore one phase of the paper life cycle: anthropogenic emissions and removals from forest carbon pools due to timber harvest. The study seeks to clarify the impact of temporal and spatial system boundaries in accounting for emissions and removals from forest carbon pools in wood and paper products.

Specifically, we compare the production of coated freesheet paper in the U.S. with that of China. With reference to PAS 2050 and the WRI Protocol, we develop a methodology to rescale Intergovernmental Panel on Climate Change (IPCC) forestry guidelines for the product level. An important conclusion from our comparative study is that with respect to land use impacts for forests, these protocols remain underdeveloped, and results may be quite variable depending on assumptions about geographic origin and temporal scale.

To develop the context necessary to suggest how forest carbon models and LCI might be more closely coupled, we provide the reader with two sections of background literature. Section 2 overviews key aspects of forest carbon science, forest harvest practices, and IPCC national GHG inventory methodologies for forests. Section 3 discusses LCA modeling to date in the forest product sector and how emerging product carbon footprint protocols–PAS 2050: 2011 (BSI British Standards Institution et al., 2011), WRI Protocol (WRI and WBCSD, 2011), and ISO 14067 (ISO/DIS, 14067, 2012)–address carbon accounting for issues such as direct land use change and land use modification, soil carbon, geographic specificity, and time. Section 4 details the scope, data, and method of the comparative coated paper case study, while Section 5 presents the results. In concluding, Section 6 offers pathways to improve how land use change and modification are incorporated into carbon footprint LCA models and related protocols for products made from forests.

2. Forest carbon dynamics and IPCC guidelines

Forests have five primary carbon pools–above-ground biomass, below-ground biomass, deadwood, litter, and soils–that simultaneously accumulate and release carbon (IPCC, 2006). Many factors interact to affect the flux dynamics of these carbon pools, including the type of forest ecosystem, the age of the forest, and if harvested, the length of stand rotation cycles and the forestry practices used. In this section, we briefly profile major forest types, discuss how harvest practices can affect forest carbon pools, and introduce readers to common approaches to measuring forest carbon, including guidance from the IPCC for national-scale GHG inventories.

2.1. Forest types

Our planet has a wide-variety of forest ecosystem types, with the major divisions being tropical, temperate, and boreal forests. These forests, at a variety of geographic scales, each have their own 'disturbance' history. In addition to the tropical–temperate–boreal nomenclature, scholars often categorize forests according to their anthropogenic disturbance patterns and forestry management regime. In a global study of forests that supply the pulp and paper sector, Subak and Craighill (1999) divided them into wood supply by forest type in just three land use categories: 'original' converted forests, plantations, and regrowth forests. In a follow-up study, Wood Resources International and Seneca Creek Associates (2007) divided forests into six categories: 'original' forest, mixed tropical hardwoods, managed natural regeneration, unmanaged natural regeneration, native plantation, and exotic plantation.

This delineation is important because an emerging body of research suggests that 'original' forests (also referred to as 'primary' or 'frontier' forests), across a range of geographic regions and ecosystem types, hold significantly more carbon in above-ground and below-ground biomass than do managed forests or plantations (Dean et al., 2003; Harmon et al., 1990; Thornley and Cannel, 2000). Contrary to the long-standing view that they are in equilibrium with respect to their carbon balance, recent research by Luyssaert et al. (2008) concluded that these forests (up to 800 years old) can continue to accumulate carbon, throwing into question the notion that they generally reach a known point of carbon stasis (i.e., incorporating net accumulation and net release).

Therefore, modeling the carbon dynamics associated with timber harvest depends on the type of forest in question. Significantly disturbing or converting some types of forests can negatively impact the global carbon cycle. Logging these forests, in a sense, represents an *opportunity cost* (Ford, 2010), as the time necessary for a harvested forest to regain its carbon sink capacity can take many decades, and if left undisturbed some types of forest, as shown in Fig. 1, would have gone on to expand its carbon pool or at least remain in relative stasis over time.

2.2. Harvest practices

The types of forest practices utilized, ranging from indiscriminate clear-cutting to sustainable forestry, can also affect the degree of carbon

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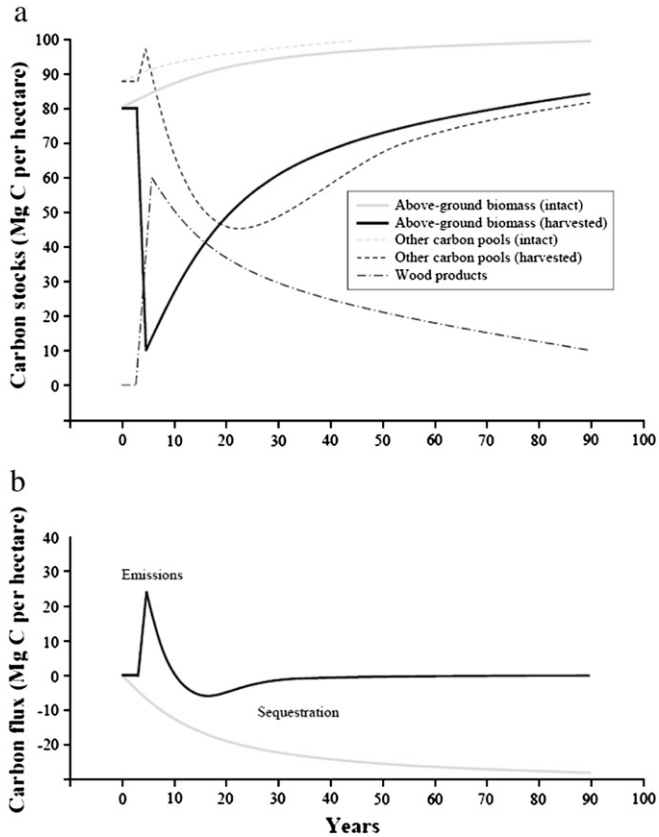

Fig. 1. (a) Idealized carbon stock changes in above-ground biomass, wood products, and other carbon pools (e.g., litter, dead biomass, soil organic matters, etc.) for intact (in gray) and harvested (in black) scenarios in a temperate forest after clear-cutting. (b) The corresponding net carbon exchanges between terrestrial ecosystems and the atmosphere, positive values indicate carbon emissions.
Adapted from "Aboveground forest biomass and the global carbon balance" by Houghton, (2005), Global Change Biology, 11, p. 946. Copyright 2005 by Houghton (2005). Adapted with permission.

loss as well as the ability of a forest ecosystem to recover its carbon sequestration capacities post-harvest (Magnani et al., 2007). Scientific understanding of carbon dynamics in above-ground and below ground biomass is better than with soils—where most of the world's biotic carbon is actually stored, particularly in the northern hemisphere's boreal forests (Pan et al., 2011). Schlesinger (1997) found that when frontier coniferous and deciduous temperate forests are converted to secondary forests, the soil carbon declines by about 10%, and this loss is a final rather than an annual value. Guo and Gifford (2002) performed a meta-analysis on the effects of land use change on soil carbon stocks and found that conversion of natural forests to plantations decreased soil carbon content by about 13%. Finally, in soil in particular, there can be the release of methane as a result of anthropogenic activity, such as logging. This is true for many forest biomes, from the boreal forests in the northern hemisphere whose soils release methane when the permafrost melts as a result of anthropogenic activity (Newell, 2004), to the tropical forests of Indonesia when the methane-rich peat soils are drained to plant fast growing plantations (Uryu et al., 2008).

However, research also indicates that in some geographic regions, industrial timberlands may be managed both to provide forest products and some degree of carbon sequestration (Birdsey et al., 2006). This reportedly can be done by increasing the sequestration of the below-ground biomass, as well as leaving logging slash, litter, and deadwood on the forest floor. Predominant forest practices, however, generally disregard these carbon pools in favor of sustainable wood yield and site quality (Houghton et al., 1999). Intensive management practices need to be quantified, however, to ensure that fossil carbon burdens associated with the production of fertilizer and other energy-intensive cultivation and harvest activities do not offset the gains (White et al., 2005).

Carbon-related impacts due to harvest practices need further study (Eriksson et al., 2007). The Forest Stewardship Council (FSC), which sets forth principles, criteria, and standards for sustainable forestry, still does not have a stated position on carbon loss due to forest practices, although they do have a working group to explore the role that FSC and forest certification can play in frameworks and projects to mitigate climate change.

2.3. IPCC GHG inventories for forests

There are a number of models to estimate changes in forest carbon pools due to direct land use change (dLUC), as well as the *modification* of land use or land cover. In IPCC terminology, dLUC is 'land converted to a new land use' (e.g., forestland to cropland). Land use modification refers to 'land remaining in the same land use' (e.g., natural forest to plantation forest) under various harvest practices. IPCC requires estimating change in carbon stock in both categories and provides detailed guidelines and equations for calculating these changes, primarily at a national level (IPCC, 2006). Fundamental to the IPCC methodology is the concept of hierarchical tiers (Tiers 1, 2, 3). The Tier 1 approach uses default emission factors and basic equations. Country-scale activity data are necessary but this information can often be gleaned from global-scale compilations. Tier 2 tends to use the same approach as Tier 1, but with country or region specific emissions and activity data. Tier 3 involves "higher order" methods and draws on a variety of models and inventory systems that are often very site-specific, rather than at a country or broad regional level. Tier 3 guidelines provide much more detailed, spatially-explicit accounting of carbon dynamics in a particular forest ecosystem, as shown in Fig. 2. Selecting the appropriate Tier depends on factors such as data availability and geographic scope of the inventory. Despite its broad scope, changes in above-ground biomass, below-ground biomass, and soil carbon pools are calculated. Only dead organic matter carbon pools (e.g. litter, etc.) are excluded.

Also fundamental to the IPCC methodology is a forest stratification scheme based on climate (e.g., boreal, temperate moist, or tropical), ecosystem zone (e.g., polar or tropical rainforest), soil type (e.g., sandy, wetland, or organic) and management practice (e.g., perennial tree crop). This scheme is used to apply emissions and carbon stock factors to estimate carbon stocks in a particular geographic region and forest type. Principal sources for Tier 1 data include datasets compiled by the UN Food and Agriculture Organization (FAO). However, not all of these datasets are harmonized with IPCC guidelines (FAO, 2001). For example, the FAO's Forest Resources Assessment (FRA) datasets have information at the national level for many countries for above-ground biomass disaggregated by forest and other wooded land. However, these datasets contain specific information neither on managed land, nor on wood removal volumes by land use category (such as forestland).

In the IPCC guidelines there are two primary, equally valid methods to determine changes in carbon stocks. The first is the gain–loss method, which underpins the Tier 1 level and essentially subtracts forest carbon loss from carbon gain. It is a process-based modeling approach that draws on ecological theories of how trees grow and how photosynthesis sequesters carbon, taking both natural and anthropogenic processes into consideration (IPCC, 2006). The second is the stock-difference method, which uses large-scale carbon stock inventories for a specific land area, and measures the difference between two or more time periods. This method requires more data and is usually used in Tier 2 and 3 approaches. Gain–loss and stock-difference methods can be combined to provide a more complete estimate of carbon flux. Examples of Tier 3 approaches that combine these two methods include the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) and biomass energy research by the Commonwealth of Massachusetts (Kurz et al., 2009; Manomet Center for Conservation Sciences, 2010).

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2.4. Temporal considerations

The effect of time is perhaps the most difficult variable to include when calculating changes in carbon stock. Forests grow back at different rates, some are more prone to natural disturbance (e.g., fire, pest outbreaks, etc.) and the harvest practices deployed affect the rate of regeneration. The IPCC guidelines are primarily designed to calculate national-level carbon loss or gain for 1 year, although within the gain–loss method there are some assumptions about regrowth rates built into the equations.

To incorporate temporal consideration, scholars have taken varying approaches. Subak and Craighill (1999) incorporated carbon loss due to logging of frontier forests by assuming that average storage after harvest was about half of the original level. Appreciated for its simplicity,

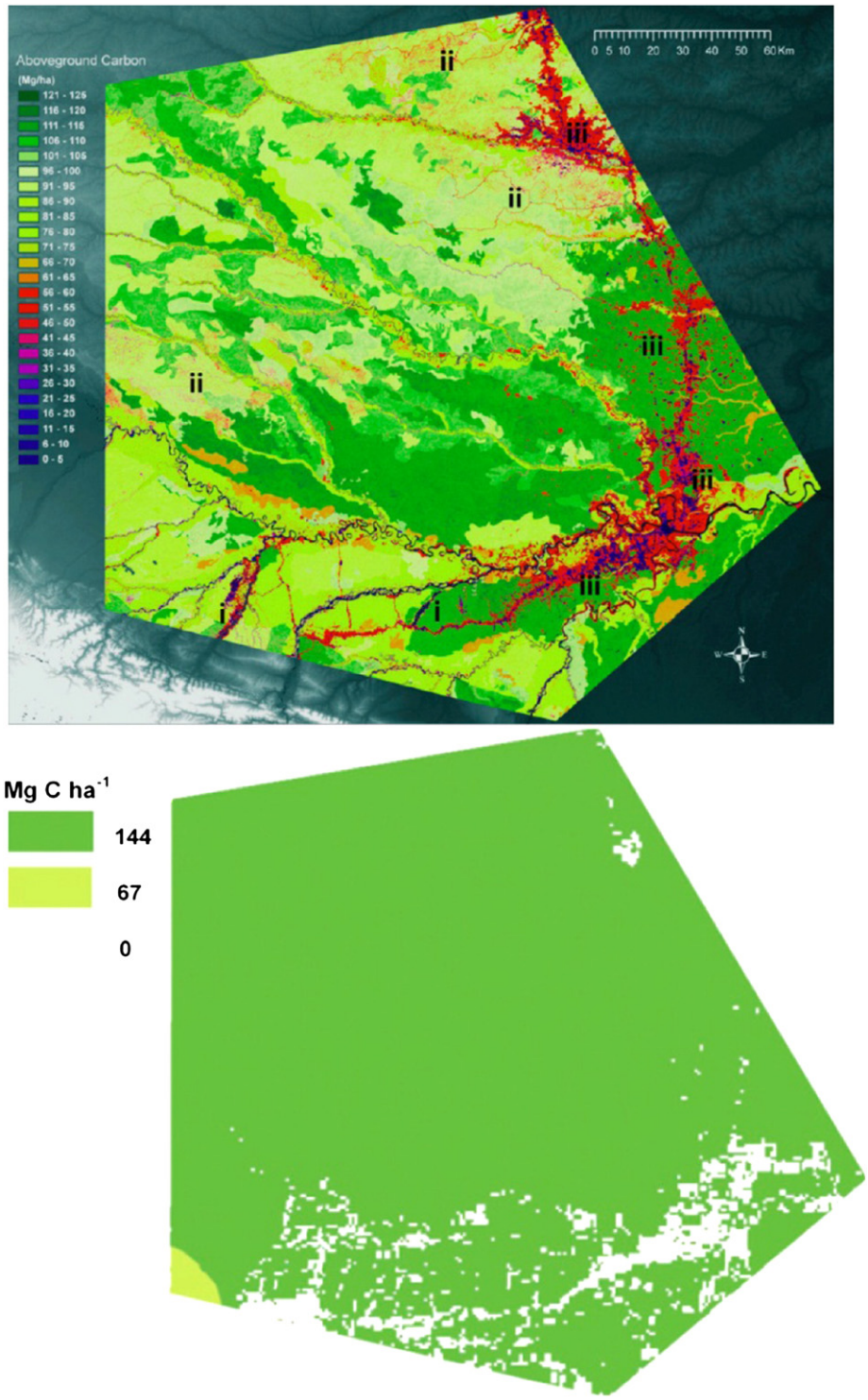

Fig. 2. (a) Tier 3 spatially disaggregated carbon stocks of AGB (in mega-gram per hectare), by Asner et al. (2010). The shades of green and yellow indicate variation of the carbon stocks, while the red and blue indicate natural and anthropocentric disturbances. (b) Tier 1 resolution of the carbon stocks (in mega-grams per hectare) in the same region. Adapted from "Reply to Skole et al.: Regarding high-resolution carbon stocks and emissions in the Amazon" by Asner et al. (2011), Proceedings of the National Academy of Sciences, 108(4), p. E14. Copyright 2011 by Asner et al. (2011). Adapted with permission.

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it has proved to be an influential approach. Essentially, it tries to factor in science that indicates permanently diminished sequestration capacity, but avoids consideration of specific regrowth rates, harvest practices, and other variables. 'Plantations' were assumed to be a net carbon sink. For 'regrowth' forests, they assumed that forest regrowth offsets harvest pulpwood—thus a net zero balance. The authors essentially eliminated the time scale question by making assumptions for three very broad forest types.

3. Forest product carbon footprints in LCA practice and international protocols

Typically, LCA practitioners rely upon professional LCA modeling software with associated life cycle inventory (LCI) and life cycle impact (LCIA) models (e.g., SimaPro of Pré Consultants, GaBi of PE International) (Graedel and Allenby, 2010). LCIs are the key component in terms of incorporating changes in forest carbon pools into wood product carbon footprints. However, the current wood LCI models lack sufficient coverage to allow for modeling of the broad array of forest types (and wood fiber) found in globalized supply chains. In part due to these limitations, carbon footprint protocols struggle to establish pragmatic rules, while incorporating the carbon dynamics associated with dLUC and land use modification globally. In addition to reviewing LCA studies of forest products, this section focuses on the rules and procedures emerging from product carbon footprint protocols, specifically from the standpoint of system boundary choices for forests in life cycle assessment of wood-based products.

3.1. Forest carbon dynamics in LCA modeling

Scholars and LCA practitioners have used primary case study data to complete LCAs for a range of forest products, including paper (González-García et al., 2011; Leach and Givnish, 1996; Lopes et al., 2003), furniture (González-García et al., 2009; Rivela et al., 2006), Christmas trees (P.E. Americas, 2010), and construction products (Bergman and Bowe, 2011; Nebel et al., 2006; Perez-Garcia et al., 2005). Most of these studies generate or rely upon forest LCI models for a narrow band of secondary, temperate forest types, which demonstrate complete uptake and regeneration of carbon from harvested wood (i.e., above-ground biomass) over limited time scales. Because biogenic carbon is treated as carbon neutral (except perhaps for small amounts released in landfills as methane at end of life), the studies generally conclude that dominant life cycle phase in the product carbon footprint is the manufacturing phase. Not surprisingly, fossil carbon from plantation management (e.g., fertilizers, seedlings, and thinning and harvest activities (e.g., equipment fuel) plays only a minor role relative to manufacturing.

In contrast, a few studies using life cycle methods link to carbon models for other forest types, forest management activities, and harvest practices over longer time scales and for more carbon pools (i.e., below ground biomass and soils) (McKechnie et al., 2011; White et al., 2005). These studies indicate that carbon footprints may vary significantly due to land use modification, and, by implication, dLUC as well.

Studies of wood-based products typically rely on life cycle inventory (LCI) models for wood provisioning found in three sources: Ecoinvent (Ecoinvent Centre, 2011), the U.S. LCI (NREL, 2011), and GaBi databases (P.E. International, 2011). In these three modeling software packages, wood LCI subset databases are the fundamental building blocks for many products and processes (e.g., sulfate pulp in paper, plywood, oriented strand board etc.). Such LCIs may in fact be an appropriate match for product systems where supply chains rely exclusively on the same forest types and forest management practices studied in the LCIs. However, lacking LCI models for other forest types, practitioners sometimes rely on these models to assess products made from globalized wood supply chains where there may be significant mismatches.

Wood LCIs applied in models derived from all three major global datasets make an implicit claim of carbon neutrality for biogenic (forest) carbon. Biogenic carbon from above-ground biomass enters the model in elementary flows of wood, where the uptake of carbon dioxide is accounted for as a raw input from nature. This biogenic carbon, which is not characterized in the climate change impact models, is contrasted throughout the LCI with fossil carbon (e.g., fertilizers, harvest equipment, etc.). Leftover wood scraps (i.e., hog) are a fuel source for many forest-based products (e.g., paper, plywood, etc.) and so the biogenic carbon dioxide emissions from hog fuel and from end of life processes (e.g., incineration, landfill, etc.) are tracked. But based on the rationale that they will be offset by subsequent forest regeneration (Bergman and Bowe, 2011; Früwald et al., 2001; Johnson et al., 2004; NCASI, 2005; Werner et al., 2007), these emissions are considered carbon neutral.

The forest models that underpin this assumption of carbon neutrality in these three LCI software packages only look at above-ground biomass and focus on a narrow range of forest types and harvest practices. The wood provisioning LCI models use specific forest models (often Tier 3 methodological approaches), but they are based on studies of temperate plantations or intensively managed natural forests in just five geographic locations, three in Europe and two in the U.S. For example, Ecoinvent models for hardwood and softwood production are based on studies of Swedish and European Union (EU) plantations (Bauer, 2007; Werner et al., 2007).

The documentation cited for the GaBi wood product models indicates a method developed for central European plantation forests (Früwald et al., 2001). In the photosynthesis sub-module, biogenic carbon values for "typical" central European species of hardwood and softwood are supplied (i.e. above-ground biomass), along with the claim that it is not practical to distinguish various wood types. The study method includes detailed accounting for plantation processes like care of seedlings, thinning, pesticide application, and liming.

The U.S. LCI models for hardwood and softwood are from studies of privately owned temperate forests, one in the Pacific Northwest and one in the Southeastern U.S. (Johnson et al., 2004; Perez-Garcia et al., 2004). As partially acknowledged by Werner et al. (2007), these studies give a narrow (and optimistic) set of possibilities for disturbances to forest carbon pools. The forestry rotations are relatively short, much shorter than commonly practiced in unmanaged forests. In the U.S. LCI models, the system boundary includes above-ground biomass, but excludes downed material and foliage from harvested trees (i.e., litter) and assumes that below-ground biomass grows and decomposes at the same rate (i.e., carbon neutral). Even though these tend to be Tier 3 studies, the soil carbon pool is excluded entirely. In sum, with no unmanaged natural forests and no tropical or boreal forests in the models, a significant proportion of global wood production is unrepresented.[2]

As a result, most wood product LCA studies rely on these LCIs in the three software packages and focus on above-ground biomass only. They make the same assumptions of carbon neutrality for biogenic carbon as the 'global' LCI datasets without specifically identifying the time required to regenerate the above-ground biomass. A particular forest study is often applied across varying geographies. Lacking primary data on forest management practices for 25% of wood coming from the Baltic region, for example, González-García et al. (2011) assume that a Swedish plantation is representative. In

[2] The publishing organization for the studies used in the U.S. LCI is the Consortium for Research on Renewable Industrial Materials (CORRIM). For the German studies used in GaBi databases, it is the *Bundesforschungsanstalt für Forst-und Holzwirstchaft* (Federal Center for Forestry and Forest Products). Both organizations are enthusiastic supporters of the idea that plantation forestry can bring increased carbon storage in products or on land restored to forestation, along with other ecological benefits. Often, the broader point of studies is the favorability of carbon footprint from wood in construction or furniture products versus competing materials (e.g., steel or plastic).

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an LCI for wood flooring products, Bergman and Bowe (2011) report a wide-array of wood species from across the Eastern U.S. (around 30 states), but there was no tracking or reporting for carbon dynamics in specific forests. A comparative LCA for natural and artificial Christmas trees helpfully discloses that carbon loss from land use change, as well as carbon stored in soil and litter, is excluded in modeling plantations (P.E. Americas, 2010).

Several LCA studies do track carbon pools in specific forests by coupling forest models and LCA methods to account for land use modification (McKechnie et al., 2011; Perez-Garcia et al., 2005; White et al., 2005). These studies pay close attention to the time carbon is stored in products versus the time it takes to sequester carbon in forest re-growth. Variation in forest type and harvest practices requires analysis at disaggregated spatial scales for above-ground, below-ground, and soil carbon pools. One study finds that for a 100-year period, biofuels made from a Canadian natural temperate forest would create significant net emissions of carbon dioxide relative to gasoline over the full life cycle (McKechnie et al., 2011). When coupling an 'industrial carbon budget' and a 'forest carbon budget' for three separately managed temperate forests in the U.S., White et al. (2005) report a wide variation, from net negative to net positive carbon flux. The magnitude of carbon pools in the forest is much larger than in the products. Thus, uncertainty in the forest model drives this variation, and many parameters in forest carbon flux need to be better understood.

None of the LCA studies mentioned accounts for dLUC, but the challenges are similar to those outlined for land use modification above. Ecoinvent has one model for dLUC, a 'stubbed land' model (i.e., land with only stumps remaining) covering annual cropland for Malaysia in hectares. This LCI carries very high uncertainty and has a number of limitations. First, it transposes one study of Brazilian tropical forest to quantify dLUC for tropical forests in Malaysia. Second, it excludes soil carbon and assumes that the carbon pool loss includes only biogenic carbon from the 20% of above-ground biomass hypothesized to be burned during the conversion process. Finally, it is unclear whether this is a realistic allocation of biogenic carbon emissions (i.e., the portion that does not regenerate as annual cropland biomass and the portion left unused by other product systems). In any case, in the Ecoinvent model, the stubbed land LCI serves as an input to agricultural (food) products and requires further development if it is to be used to account for release of biogenic carbon due to dLUC in harvesting for wood products

3.2. Emerging product carbon footprint protocols

There are two finalized international product carbon footprint protocols–Publicly Available Specification 2050 (hereafter "PAS 2050") (BSI British Standards Institution et al., 2011) and the Greenhouse Gas Protocol, Product Lifecycle Accounting and Reporting Standard (hereafter 'WRI Protocol') (WRI and WBCSD, 2011)–and a third, the draft ISO standard (ISO/DIS, 14067, 2012). These protocols are based on foundational LCA principles and methods (e.g., ISO 14040 and ISO 14044) and both mandate the IPCC 100-year impact model. As ISO/DIS 14067 is the least specific in terms of methodology and guidance, in this paper we focus principally on the implications of PAS 2050 and the WRI Protocol for wood product carbon footprints. Table 1 compares and contrasts how these three protocols address land use change, geographic variation, and temporal boundaries.

Relative to what is typical for LCA wood product studies, the WRI Protocol and PAS 2050 are fairly rigorous. Emissions due to indirect land use change (iLUC) are excluded, but in the case of direct land use change (e.g., forest to cropland) both require biogenic carbon accounting. The WRI Protocol requires accounting for the carbon impacts due to land use modification (i.e., 'conversion within land use categories,' p.37), but PAS 2050 does not. Therefore, timber harvest in natural forests should be accounted for, including cases of conversion of natural forests to plantations. ISO/DIS 14067 requires consideration of direct land use change 'when significant' but does not define a specific threshold for significance (i.e., a cut-off rule for the proportion of the footprint that must be covered).

To perform calculations, all three protocols suggest using the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). The WRI Protocol provides a hypothetical case of conversion from forest to cropland and PAS 2050 provides an emissions factor table based on scaling IPCC Tier 1 results down to the product level. To reduce uncertainty, both suggest developing specific supplemental guidance.

IPCC Tier 1 provides formulas and emission factors for soil carbon. PAS 2050 requires accounting for soil carbon (change) in dLUC, but does not provide specific guidance. In the WRI Protocol, accounting for soil carbon is optional. ISO/DIS 14067 is less clear on this point. Section 6.3.9.4 calls for including dLUC in the footprint in accordance with IPCC Tier 1 guidelines. But later it reads, "*Unless* calculated as part of LUC, the GHG emissions and removals occurring as a result of soil carbon change *should* be included in the life cycle inventory and *should* be assessed and *shall* be documented separately in the CFP study report" (emphasis added) (Section 6.3.9.5 ISO/DIS 14067). Thus, in the case of dLUC, including soil carbon change in the carbon footprint appears to be required. But, in instances of soil carbon change where there is no dLUC (e.g., land use modification), inclusion is optional and must be reported separately (e.g., soil management practices at tree farms).

In the case of globalized supply chains, both PAS 2050 and the WRI Protocol recognize the practical difficulties companies face in identifying exactly where materials come from. In its previous version, PAS 2050:2008 required a worst-case scenario assumption if locations, timing, or types of dLUC are unknown. This incentivized companies to gather this information (BSI British Standards Institution et al., 2008b). The updated PAS 2050:2011 still requires the worst-case assumption when the timing of dLUC is unknown (BSI British Standards Institution et al., 2011). But if the type of dLUC is unknown, it now requires an average of PAS 2050 factors for various types by country. When both the type and location are unknown, it requires a weighted average of emission factors for all the countries (i.e., global wood production). The WRI Protocol is even more flexible. Companies can report based on best, average, or worst-case scenarios. As seen in our comparative case study below, this flexibility lends itself to either giving widely divergent results for forest products, or leading to footprints based on the global best-case scenario. In sum, these protocols need further development to handle spatial complexities in forest carbon product footprint accounting.

3.3. Temporal considerations

In the protocols, the temporal considerations are not that clearly specified. Table 1 lists four temporal components–carbon storage in products, weighting for delayed emissions, and how far back or forward in time to account for dLUC or land use modification–that impact wood product carbon accounting. The protocols strive to separate biogenic carbon emissions from biogenic carbon stored in product. In ISO/DIS 14067, biogenic carbon stored in the product must be included in the carbon footprint (CFP), but the impact may not be reduced due to a delay in emissions, as noted in the following language: "Where GHG emissions and removals arising from the use stage or from the end-of-life stage occur over more than ten years after the product has been brought into use, these GHG emissions and removals shall be included in the CFP without the effect of timing of the GHG emissions and removals" (Section 6.3.8, ISO/DIS 14067). Nevertheless, the timing of emissions and removals relative to the year of production still needs to be specified in the life cycle inventory. Furthermore, the effect of timing for emissions longer than 10 years may be calculated and documented separately from the CFP (Section 6.3.8 ISO/DIS 14067).

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Table 1
Dimensions of Product Carbon Footprint Protocols that Impact Accounting for Forests.
Sources: ISO/DIS 14067 (2012), BSI, British Standards Institution et al. (2011), WRI and WBCSD (2011).

Attribute	ISO/DIS 14067	PAS-2050 (final)	WRI Protocol (final)
Land use and carbon pools			
Direct land use change (dLUC)	Required when 'significant'	Required when purpose of land is changed	Required Also required for change *within* a land use category
Indirect land use change (iLUC)	Not required	Not required, for future consideration	Not required, may be reported separately
Soil carbon	Recommended	Required, for dLUC only	Optional, contingent on data availability
Geographic specificity/material origin	Not addressed	Not required, but sub-country level preferred If country of origin is unknown, use weighted average of all countries where feedstock is produced	No requirements for geographical specificity Encouraged to perform scenario uncertainty analysis for likely growing regions. Range of reporting options
Temporal considerations			
Carbon storage and release in use phase or at end-of-life (EoL)	May report separately if >10 years	Report if >100 years Document sources used to calculate product lifecycle	If no guidance, use >100 years If guidance, using specific sector for time frame
Delayed emissions weighting	Not addressed	Required formulas provided	Not allowed, may be reported separately
Backward time period for dLUC	Vague	20 years or a single harvest period (whichever is longer)	20 years or a single harvest period 12(whichever is longer)
Forward time period for dLUC	Not specified	Equal allocation over 20-years	Equal allocation over 20 years

For PAS 2050, carbon stored in wood products within 100 years is excluded from the footprint, but may be separately documented as emitted in longer time periods.[3] For the WRI Protocol, the time frame is to be determined by any available sector guidance. This will be critical for wood products, as previous LCA studies find that product carbon pools are significant relative to forest carbon pools (Lippke et al., 2004; Perez-Garcia et al., 2005).

Another important temporal consideration is how far backward do we need to go to track dLUC or land use modification, and how far forward to account for removals and the forgone opportunity to store additional carbon in disturbed forests (i.e., opportunity cost). Unless it is within a single harvest period, both PAS 2050 and the WRI Protocol exclude dLUC or land use modification older than 20 years. Since many forests have harvest rotation cycles longer than 20 years, most land producing forest products will require assessment.[4] To make forward-looking assessments, both protocols instruct practitioners to allocate *single* removals of biomass (at the year of dLUC) equally over 20-year time periods, again a relatively short period for forest carbon cycles (BSI British Standards Institution et al., 2008b; WRI and WBCSD, 2011, p.117–125).

Finally, there is the question of whether or not to apply weighting factors for delayed emissions for carbon stored in products or for delayed sequestration due to forest re-growth. PAS 2050 has a formula where delayed emissions have less impact, while the WRI Protocol does not allow such weighting (i.e., assumes all emissions or removals occur in the year the product is made). With the long time cycles of some forests and wood products, this distinction between PAS 2050 and the WRI Protocol is an important one.

4. The case of coated freesheet paper: scope, data, and method

To better understand how these protocols address spatial and temporal considerations, land use issues, supply chain complexity, and system boundary decisions, we compared coated freesheet paper supply chains for about 12 factories in the U.S. operated by the largest U.S. producer (NewPage Corporation with about 35% of North American production) with 10 of the largest factories in China. Our reference flow was an average production of 1 metric ton of paper, cradle to gate (i.e., no use or disposal phases) for 2007. We looked solely at the wood sourcing and the main manufacturing processes, excluding other primary materials such as clay and treatment chemicals, additives and other processes such as wastewater treatment, harvesting equipment emissions, fertilizer, etc. We only looked at carbon dioxide emissions, excluding other GHGs. We focused on three key areas where we thought the carbon dioxide emissions were likely to differ most based on underlying geographical variation: transportation, energy use in pulp and paper production, and carbon loss due to timber harvest. In this paper we focus on the issue of potential carbon loss to timber harvest from dLUC, results of the broader study can be found in Newell and Vos (2011).

4.1. Wood fiber supply sources

The virgin fibers required for coated freesheet paper production are pulped in two basic commodity types: bleached hard kraft pulp (BHKP) and bleached soft kraft pulp (BSKP). Data found in Cornerstone produced by RISI (2007) provides flow sheets for production of coated freesheet paper for facilities throughout the world, including fiber input ratios of BHKP to BSKP. In both supply chains, for the facilities we analyzed, almost no recycled fiber was used to make coated freesheet paper. Although Cornerstone does not include a complete list of every paper or pulp producer, it is the most thorough data available for the industry on a global scale. Using Cornerstone, we determined the amounts of wood inputs to make BHKP and BSKP at pulp mills and the amounts of pulp and wood needed to make paper at integrated mills or the amounts of pulp needed to make paper at non-integrated mills in the two supply chains.

The wood supply structure for NewPage's facilities is primarily locally sourced. Using Cornerstone and based on personal interviews, we mapped out the fiber supply structure and production processes. Like most U.S. manufacturers, NewPage has integrated pulp and paper production at the same facility and logs are sourced by harvesting wood from managed natural forests within approximately a 100-mile radius of each facility. A small amount of pulp is also imported from Canada to supplement local wood supplies (about 10% of the required fiber). To create even data resolution in comparison with the

[3] In addition to excluding carbon from the footprint when stored in products, PAS 2050:2008 offers a credit for carbon storage that is "additional to what would have been stored anyway" (BSI British Standards Institution et al., 2008a, p.28). According to the 2008 guidance, this applies to harvests from managed forests but not 'natural' ones.

[4] A case may be made that this rule reflects a policy judgment about what historical practices to hold accountable for degrading forest carbon pools. It, therefore, places the "onus" for maintaining forest carbon pools on industrializing countries that do not have long histories of deforestation.

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supply chain for China, we did not track specific forests or forest management practices for the NewPage supply chain, but rather placed the harvest locations by country into general U.S. and Canadian categories.

To create country-level harvest locations for China's supply chain, we mapped out the fiber supply structure for the industry as a whole. China currently imports more than 90% of the pulp that it uses to produce paper. Trade data (Global Trade Information Services, 2008) reveals that for China's industry, just over 75% of the imported supply of BHKP in 2007 came from three countries (i.e., Indonesia, Brazil, and Chile), and about 71% of the BSKP comes from four countries (i.e., Canada, Chile, Russia, and the U.S.). The balance comes from a large number of countries, including New Zealand, Finland, and Thailand (Fig. 3).

We then developed weighted averages for the locations of fiber supply harvest and production for the fiber supply for the coated paper industry in both China and the U.S. From these import percentages, the pulp for coated freesheet production can be estimated to the level of the pulp mill based upon the global distribution of major pulp mills producing BHKP and BSKP (Fig. 4). However, to apply IPCC Tier 1 dLUC figures, we only need to estimate the country scale. This aspect of our approach conforms to the revised PAS 2050:2011 'weighted average by country of production method' for unknown land use locations.

4.2. Scaling IPCC guidelines to product-level

To investigate carbon loss from timber harvest in the two supply chains, we customized the methodology used to calculate national GHG inventories specified in the IPCC's Good Practice Guidance for Land Use, Land-Use Change and Forestry, especially Chapter 4 (Forest Land) and Chapter 12 (Harvested Wood Products) (IPCC, 2003, 2006). This guidance document, extensively referenced in all three protocols, provides equations and methods at a national level for accounting for biomass, dead organic matter and soil carbon stock changes in all land-use categories as well as default factors to convert from forest product unit (e.g. roundwood, paper, etc.) to carbon. The terminology used and methods in this IPCC guidance document are also consistent with many FAO emission factors and activity data for forest categories. These data generally are derived from the broadest levels of forest type. For example, to calculate carbon loss due to the draining of organic soil, emission factors are categorized broadly for tropical forests, temperate forests, and boreal forests. Obviously, there is tremendous variation depending on forest type and management, and harvest practices—local conditions masked by this approach. However, customizing the IPCC approach allows for an estimate of carbon loss, which is narrowly scoped in this study to include above-ground biomass growth only (i.e., excluding below-ground biomass and soil).

4.3. Carbon loss from timber harvest

According to the 2007 global assessment of the pulp and paper industry by Seneca Creek Associates and Wood Resources International, approximately 38% of the wood fiber used to make pulp globally in 2004 came from plantations. About 49% of the fiber came from 'managed' natural forests, which is a broad category used to describe active management of forestlands, including planting seedlings, use of fertilizers, and thinning practices. The remaining 15% or so came from unmanaged natural stands, where little effort is made to ensure biomass re-growth or long-term forest productivity. Many portions of these stands are harvested without regard for regeneration. Other sources of the 15% include mixed tropical hardwood forests, and 'original' forests.

To illustrate the importance of assumptions about forest type and land use change in the supply chain, our model includes two scenarios for carbon loss from timber harvest. Scenario #1 takes the weighted average assumption for the country of pulp production, but assumes that the dLUC (if any) from timber harvest in that country is unknown. In Scenario #1, we model with a conservative assumption that all lands required for production in each country undergo dLUC from forest-to-annual cropland.[5] In Scenario #2, we make a less conservative assumption. By applying estimates from Seneca Creek Associates and Wood Resources International (2007) to develop 'natural' versus plantation forest ratios for BHKP and BSKP production in each country, we reduce the amount of land assumed to undergo dLUC. This approach conforms to PAS 2050 by separating potential land use change (i.e., natural forest to cropland) from land use modification (i.e., forests converted to plantation forest, or biogenic carbon cycles in varying plantation types and management practices). In Scenario #2, we also assume that we are in the second harvest cycle of the plantation forests, which puts dLUC out of scope for the protocols, rendering plantations carbon neutral for biogenic carbon.[6] Essentially, in both scenarios the model accounts for the release of biogenic carbon from biomass fuel combustion during pulp and paper production, from wood residue left at logging sites, and from solid residuals due to production, based on two different estimates for the amount of land undergoing dLUC.

Using an IPCC-based approach (similar to the approach described in the protocols), we estimate the biogenic carbon related to dLUC as a single loss, by subtracting the above-ground biomass forest carbon stocks from the annual increase in above-ground carbon stocks for the first year of annual cropland production across the total area used to meet fiber input requirements from each country in the weighted average. Data on forest ecosystem types (e.g., tropical, temperate, or boreal) in each country and for average yields per hectare come from FAO (2001). The model then averages IPCC Tier I estimated biomass values for each forest type to develop a per hectare average biomass profile.

In the case of hardwood from native tropical forests in Indonesia, for example, the model assumes an average above-ground biomass of 240 tonnes of dry matter per hectare, a timber yield of 59 m^3/ha, and average basic wood density–ratio between oven dry mass and fresh stem-wood volume without bark–of .51. Therefore, for each country, the model develops per hectare biomass profiles using averages of above-ground biomass, timber yield, and basic wood density (i.e., accounting for moisture content). These averages vary considerably depending on the average for wood species harvested in each country.

The customized formula is:

$$(B_{before} - B_{after}) * (C_{fd}) = C_{removal} \quad (1)$$

$$C_{removal}/Y_{yield} = C_{loss} \quad (2)$$

where,

B_{before} Biomass stocks before conversion in tons dry matter/hectare (average for each country)
B_{after} Biomass stock for annual cropland (IPCC factor of 5 tons dry matter/ha)
C_{fd} Carbon fraction of dry matter (IPCC factor of .5)
$C_{removal}$ Carbon removal in kg of carbon/hectare
Y_{yield} Yield given in m^3/hectare (FAO factor for each country)
C_{loss} kg of carbon per m^3 of wood (for each country).

[5] This aspect of our approach does not conform precisely to the revised BSI British Standards Institution et al. (2011), which would assume this change as an average of the factors for forest-to-annual cropland and forest-to-perennial cropland (i.e., estimated carbon loss would be slightly lower). It would be one of several acceptable approaches when lacking geographic specificity under the WRI Protocol.

[6] In terms of timing the dLUC, this approach does not conform precisely to the revised PAS 2050:2011 because when the timing of the land use change is unknown, we are supposed to assume that there was dLUC in the year the assessment was carried out. However, in terms of plantation forests, the approach still conforms because for most plantation forests, the relevant change would be conversion of a natural forest to plantation forest. Under PAS 2050, this type of conversion does not appear to count as dLUC (i.e., there are no emission factors for such a conversion in Appendix C of PAS 2050).

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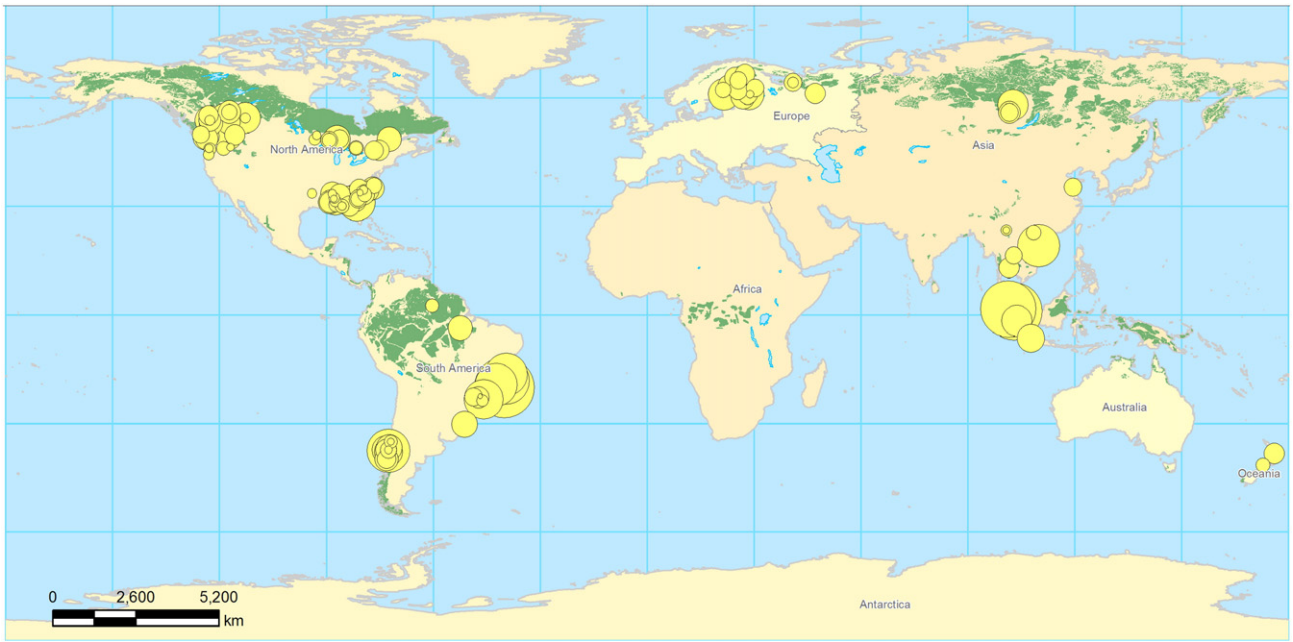

Fig. 3. Major pulp mills across the globe, 2007. Note: Frontier forests refer to forests that are largely intact natural forests as defined by the World Resources Institute. Sources: Map by authors/data from World Resources Institute (1997), RISI (2007), and ESRI (2007).

The amount of fiber required for production (mill efficiency) in the two supply chains is remarkably similar, and corresponds closely to the FAO's standard conversion factor (FAO, 2001). Using our weighted average approach, we calculate biogenic carbon emissions from dLUC by placing the carbon loss per cubic meter for BHKP and BSKP in each country in both supply chains against the amount of wood required for 1 metric ton of paper production. The formula is:

$$P * C_{loss} * W_{ef} * 44/12 = \text{kg of } CO_2 \text{ per finished metric ton} \quad (3)$$

where,

P	Percent of pulp or wood supply for each country (supply chain weighted average)
W_{ef}	Wood efficiency factor of 3.65 m^3/finished metric ton
44/12	Conversion of elemental carbon to CO_2.

Biogenic carbon is embedded in the coated paper until it is released during disposal or remains embedded in the form of recycled paper products. As our study is only cradle to gate, we estimate and subtract embedded biogenic carbon in the product from the direct land use change CO_2 emissions using the following formula:

$$D_{average} * C_{fd} = C_{density} \quad (4)$$

$$C_{density} * W_{ef} * 44/12 = \text{kg of } CO_2 \text{ embedded per finished metric ton} \quad (5)$$

where,

$D_{average}$	Average carbon density of wood species, IPCC (2006), in oven dry tons/m^3
$C_{density}$	Carbon density in kg/m^3 of wood.

The full model estimates the total biogenic carbon lost from storage in above-ground biomass for a *single* removal (at the year of dLUC), less the amount of biogenic carbon stored in the coated paper product itself.

5. Results: comparative case study of coated paper: China and the U.S.

The results from modeling Scenario #1–taking a conservative approach to unknown wood production locations–indicate much higher carbon loss from timber harvest for the Chinese supply chain (9210 kg) than for the U.S. (3517 kg) (Fig. 5). This model essentially assumes that all forests are 'natural.' As such, the Chinese industry imports more pulp from tropical forests, which (based on IPCC methodology) carry a larger carbon dioxide emission conversion burden than do temperate and boreal forests. But if we estimate the amount of plantation forest in use, and thus reduce the amount of land assumed to be undergoing dLUC in each country, the results change so much as to invert the comparison between the supply chains. In Scenario #2, the Chinese industry generates less carbon dioxide during the timber harvest phase (1368 kg) than does the U.S. industry (2671 kg) due to greater sourcing from countries that rely on plantations to produce pulp (e.g., Brazil and Chile).

The key point is that there are likely significant differences in the carbon footprint depending on the forest type and type of dLUC or land use modification that may be occurring in a given, specific supply chain. Moreover, as noted in the broader partial LCI study of the China and U.S. coated paper supply chains, the values for potential carbon loss in these scenarios for dLUC are quite large relative to fossil carbon emissions in other life cycle phases (Newell and Vos, 2011). The large magnitude and uncertainty in the values identified here for biogenic carbon emissions from dLUC correspond to other studies that look in detail at carbon cycling and land use modification in natural forests (McKechnie et al., 2011; White et al., 2005). And the relative significance of these values compared with fossil carbon emissions in other life cycle phases is supported by data in paper LCA studies that look only at plantation forests and thus treat biogenic carbon as climate neutral (Dias et al., 2005; González-García et al., 2011; Lopes et al., 2003). Thus, for any coated freesheet paper product, the carbon footprint can only be determined by the potential land use change or land use modification in the precise forest supply chain used for that product.

It is important to note that neither scenario is particularly realistic. For example, it is clearly not the case that all the natural forests harvested

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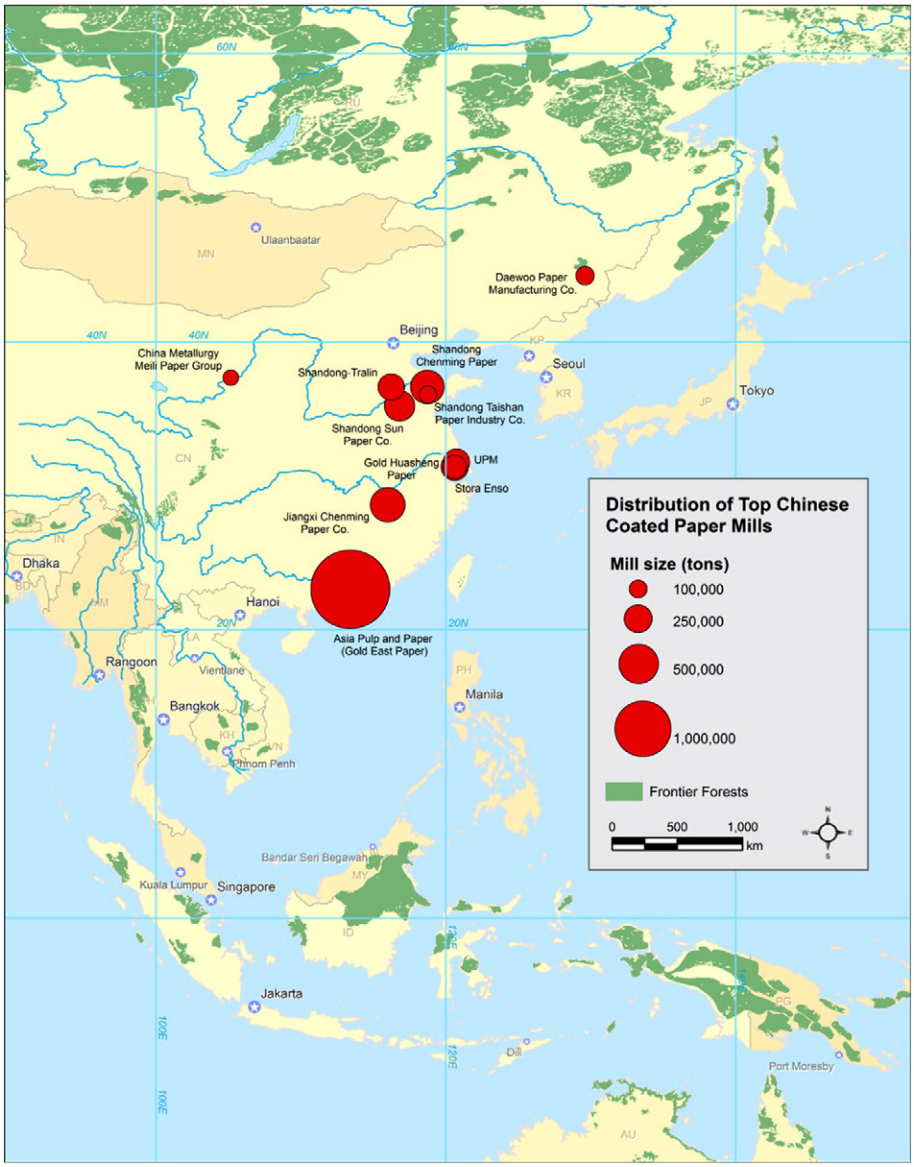

Fig. 4. Major coated paper manufacturers in China, 2007. Note: Production figures are in metric tons. Frontier forests refer to forests that are largely intact natural forests as defined by the World Resources Institute.
Sources: Map by authors/data from World Resources Institute (1997), RISI (2007), and ESRI (2007).

to make paper in the U.S. supply chain are then converted to annual cropland. An earlier study tracking plantations in a U.S. coated paper supply chain similar to NewPage finds that they are carbon neutral (Gower et al., 2006). However, in the general case of U.S. paper supply chains, there is likely some biogenic carbon storage lost from harvesting natural forests, even if they remain natural forests, depending on harvest management practices (i.e., land use modification) (Harmon et al., 1990).

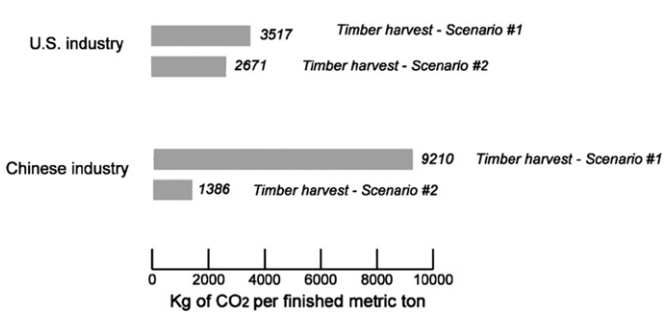

Fig. 5. Comparative coated-paper carbon footprints for timber harvest life cycle phase, U.S. and Chinese industries.

The situation is even more indeterminate for China's supply chain. On the one hand, specific companies in China are known to source widely from tropical forests in areas undergoing extensive dLUC, such as in Indonesia. On the other hand, given that China's industry is also heavily reliant on tropical forest plantations, some of which regrow rapidly, it would be important to have robust data on carbon fluxes over time in those forests. In the absence of more precise modeling of forest type and potential land use change within each supply chain, the overall footprint of coated paper is impossible to discern.

Scenarios #1 and #2 are neither best-case nor worst-case scenarios with respect to disturbances to forest carbon pools. Generally, they are biased-high for estimation of carbon footprint in the sense that they assume that dLUC–forest to cropland–is occurring for all harvested wood (Scenario #1) or all wood harvested from natural forests (Scenario #2). But they are not the worst case because the analysis here does not take into account the impact of timber harvest on soil and below-ground carbon pools. Also, the analysis assumes a permanent loss of carbon storage potential from a single dLUC event based on the difference

between the above-ground biomass at the time of harvest and above-ground biomass post-harvest, but it does not include the opportunity cost of future carbon storage in the impacted forests.

Conversely, scenarios biased-low could be created by accounting for changes from natural forest to plantation, or even denuded land converted to plantations, where there might be a net carbon gain over time. Unfortunately, adequate data on land use change and modification do not exist at the country level to make estimations (on average) for rates of such conversions or resulting carbon dynamics for each country in the two supply chains. In reality, a range of scenarios could be constructed with more specific modeling based on forest type and species and harvest practice.

6. Discussion and conclusion

This article analyzes forest carbon dynamics, LCI models, and LCA practice in the context of emerging product carbon footprint protocols. It also provides a comparative study of coated paper production to highlight the uncertainty and structural biases in current forest product carbon footprint accounting. Carbon footprint results for wood products will vary widely depending on harvesting scenarios (i.e., locations, practices, etc.) and temporal boundaries. Improving these models for wood-based products faces three primary challenges: 1) how to account for geographic variation inherent in the globalized nature of supply chains; 2) how to account for the land use changes and modifications associated with securing wood fiber; and; 3) how to account for the temporal complexities associated with changes in forest carbon pools, particularly post-harvest. Addressing these challenges has to be done in a way that is flexible enough to incorporate forest carbon science as it emerges, while avoiding making the supplemental guidance and methodology so contingent and complex that it is unusable. Toward this end, we provide specific recommendations for how this might be achieved in two key areas of focus: LCI models and product carbon footprint protocols. Finally, we suggest establishing a working group as a mechanism to discuss and standardize proposed improvements.

6.1. Improving LCI models

Current LCI models are based on a small set of studies that tend to deploy Tier 3 methodological approaches. As these offer only a narrow subset of forest types and management practices, the system boundaries are narrowly scoped and the geographic coverage is limited. IPCC criteria call for neither over- nor under-estimates of removals or emissions and for uncertainty to be reduced as far as practicable (IPCC, 2006). Applying these LCI models in LCA studies that have globalized supply chains, therefore, does not qualify as good practice. Using these LCIs is likely to lead to under-estimates for emissions and removals from forest carbon pools (i.e., more biogenic carbon is released than is currently accounted for in some specific supply chains), as illustrated by the uncertainty exposed by our coated paper case study. This reflects a broader issue in current LCA practice, where uncertainty around biogenic carbon in footprints for products derived from forests is poorly characterized.

LCI models can be improved by using an IPCC Tier 1 approach to build forest carbon profiles for major timber producing countries and thus better represent the global diversity of forested land from which we derive wood fiber. This approach would enable country-level information for a range of forest types at appropriate geographic and temporal scales for forest growth cycles, and including many of the carbon pools (i.e., above and below-ground biomass and soil carbon). As discussed in Section 2.3, IPCC already provides equations, decision rules, and emission factors for Tier 1 methods and some activity data at the country-level and for major forest types. In the future, Forest Resource Assessments (FRAs) for major countries will be developed with IPCC guidelines and equations in mind. These data and others regularly published by the FAO will provide a considerable tableau from which to develop solid carbon change profiles due to timber harvest for many countries.

A truly global suite of wood LCI models at the Tier 1 level would allow LCA practitioners to run systematic uncertainty analysis for removals and emissions of biogenic carbon due to disturbances to forest carbon pools. The resulting LCI models should represent a range of scenario assumptions regarding land use change and modification for various forest types in all countries that produce wood. If specific harvest locations are unknown, practitioners could test various supply chain allocations for countries where the wood may be sourced, ranging from a weighted average of global production to best and worst case scenarios.

To further reduce uncertainty in product carbon footprints, however, eventually site-specific data on land use management practices and forest types for supply chains need to be combined with appropriate LCIs. As the Tier 1 approach lacks sufficient detail in terms of geographic and temporal specificity, incorporating Tier 2 and Tier 3 studies of timber harvest will also be necessary to fully capture the forest carbon pool dynamics for specific locations at the sub-country level. When harvest locations are known, these models would form the basis for an expanded suite of wood LCIs that would allow LCA practitioners to match site-specific data on land use management practices and forest types for supply chains.

To foster data analysis and synthesis, two modeling frameworks hold particular promise for developing a suite of wood LCIs based on the highly specific forest conditions. The first is the so-called forest carbon 'book-keeping' approach, an input–output model often deployed for a specific administrative region that is widely used by forest carbon modelers (Houghton, 2003, 2005; Houghton et al., 1983, 1985). Essentially accounting for carbon dynamics through addition and subtraction, book-keeping forest models have been developed for regions in many countries, including Russia and Canada and they include forest carbon pools ranging from above-ground biomass to soil carbon. The book-keeping approach can also be used to test how different assumptions in carbon flux modeling (e.g., whether or not forests in 1980 were in equilibrium) affect carbon emission outcomes (Ramankutty et al., 2007; Zaks et al., 2009).

Geographic information systems are the second promising modeling framework as they allow the user to manage spatial data that contain variables such as land use and land cover, elevation, climate, soil, population, etc., over multiple time periods (Goodchild, 1992, 2003). GIS modeling software also employs remote sensing to uncover histories of land use change and modification and to provide predictive models for dLUC for sensitive regions (Eastman, 2007). Some scholars, such as Geyer et al. (2010a,b) have loosely coupled LCA with GIS for both inventory and impact analyses. While Geyer et al. focused on land use impacts of biofuel crop production, one could similarly couple LCA–GIS to better understand the spatio-temporal dynamics of carbon emissions and sequestration. Eventually, a suite of wood LCI models could be integrated with GIS models, allowing LCA practitioners to identify and select an appropriate set of wood LCI models when forest locations in the supply chain can be mapped.

6.2. Furthering development of carbon footprint protocols

The carbon footprint protocols have made considerable efforts to incorporate land use change and modification. For carbon footprint protocols, necessary inclusions and considerations include how to deal with geographic specificity, how to increase supply chain transparency (and how to model it), and what decision rules to establish on the temporal aspects of forest carbon pools. For example, given the impacts of disturbing long forest growth cycles, the 20-year forward-looking period is too short to account for 'opportunity costs' in forest carbon pools. Indeed, it would not incorporate re-sequestration of carbon even in the short-rotation plantation models that dominate current wood LCIs. Requiring a 100-year time period would parallel the required 100-year IPCC climate change impact model. Also, it would adhere to Sections 5.1–2 of PAS 2050, which requires accounting for emissions and

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removals in a 100-year time period for land use change and other life cycle phases, including carbon storage in forest *product* pools. Thankfully, in the forest ecology community, temporal dimensions of carbon dynamics are fairly well modeled by numerous methodologies at multiple spatial scales, including at the national and biomass levels (Kurz et al., 2009; Running and Gower, 1991; Running and Hunt, 1993; Thornton et al., 2005).

We suggest developing supplemental guidance for wood products specifically, and making corresponding revisions of the protocols to better account for emerging forest science and the more comprehensive LCIs. This guidance would reduce scenario uncertainty in coordination with product category rules (PCRs). PCRs are specific rules, guidelines, and requirements aimed at developing quantitative, LCA-based accounts of the environmental impact of a good or service (ISO 14025, 2006). The Confederation of European Paper Industries in collaboration with the European Commission is in the process of developing a PCR for paper products (CEPI and EC, 2011). Ideally, these PCRs would have specific rules for carbon storage by product that parallels rules for carbon storage in forest carbon pools.

Finally, we identify three improvements for both the PCRs and the carbon footprint protocols. First, both should mandate systematic uncertainty assessment for biogenic carbon removals and emissions from forests due to direct land use change and modification. Second, both need to unify in terms of which forest carbon pools to include and how to account for them. For example, as noted in this paper, there is considerable difference in the three protocols in terms of soil carbon accounting. Third, all the protocols could be worded so as to incentivize companies to get more specific about tracking forest resources in their supply chains. In the 2008 iteration of PAS 2050, for example, there was built-in accounting "penalty" when the supply chains were unknown, as the methodology required applying the "worst-case" country-level direct land use change scenario from the protocol's own estimation table.

6.3. Establishing a working group

Developing dynamic forest carbon models to link the world's forests to LCA models is a steep challenge. The supply chain for paper is among the most complex in the family of forest products, but others may be more easily traced to the source forests. Establishing a working group that cuts across key disciplines, combining the expertise of LCI developers with forest modelers, seems necessary to develop effective and inclusive supplemental guidance. This effort could be within the Forest Stewardship Council's (FSC) forest carbon working group, thereby providing access for other stakeholders, including industry and NGOs.

This working group would address some of the modeling and carbon accounting needs that we have highlighted in this paper. This working group could use the standard IPCC objectives for estimation (see Sections 1.2 and 1.4 of the 2006 Guidelines), including guidance for ensuring transparency, completeness, comparability, and accuracy. More specifically, this working group could develop a process to link the IPCC Tier 1 approach described above with the LCI models. One key agenda item would be to construct a matrix that encompasses countries and regions, forest types, management, and harvest practices, and in turn link dLUC and land use modification values for the key forest carbon pools for each category. Meta-analysis of existing studies, using clearly defined system boundaries, could be done for Tier 2 and Tier 3 type studies and gradually incorporated if consistent. This guidance could inform and be informed by the requirements in the protocols. It may be that these studies would reveal inconsistencies in the protocols with respect to requirements about temporal scale and opportunity costs, for example. Finally, this forest carbon-land use classification matrix could be linked to chain-of-custody and certification schemes to aid companies in applying appropriate average values for land use change and modification.

This matrix would enrich the body of LCI available sensitivity analysis for wood-based product carbon footprints. When the supply chain is unknown, a global worst-case scenario could be applied to see if biogenic carbon emissions exceed a threshold value (e.g., 5%) of the total product carbon footprint. This could trigger deeper investigation of the supply chain. Furthermore, this matrix could help product designers make sound decisions about alternatives to wood (e.g. steel, plastic, etc.). Sensitivity analysis will allow one to understand tradeoffs between fossil carbon emissions from recycled products (that use more fossil fuel) and biogenic carbon emissions from virgin content products (that use more biomass fuel). If wood is sourced from forests with high carbon losses, an appropriate carbon footprint method may reveal recycled content to be of even greater value across the full life cycle than seen in current models.

Product footprint protocols, such as PAS 2050, have the potential to greatly enhance sustainable global forest policy, and link directly to the growing number of the UN's Reducing Emissions from Deforestation and Forest Degradation (REDD) programs underway worldwide. In general terms, the principles and guidelines for modeling of forest carbon sequestration in the product footprint protocols should be brought into alignment with those employed for offsets under REDD's carbon market program. Given the global importance of forests in mitigating climate change, LCA models need to expand efforts to trace carbon footprints and other impacts back to their origins, thereby developing a spatially-explicit LCA practice that understands the complexity and importance of the land where products are born.

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