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Damaged forests provide an opportunity to mitigate climate change

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Abstract

British Columbia (BC) forests are estimated to have become a net carbon source in recent years due to tree death and decay caused primarily by mountain pine beetle (MPB) and related post-harvest slash burning practices. BC forest biomass has also become a major source of wood pellets, exported primarily for bioenergy to Europe, although the sustainability and net carbon emissions of forest bioenergy in general are the subject of current debate. We simulated the temporal carbon balance of BC wood pellets against different reference scenarios for forests affected by MPB in the interior BC timber harvesting area using the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3). We evaluated the carbon dynamics for different insect-mortality levels, at the stand- and landscape level, taking into account carbon storage in the ecosystem, wood products and fossil fuel displacement. Our results indicate that current harvesting practices, in which slash is burnt and only sawdust used for pellet production, require between 20–25 years for beetle-impacted pine and 37–39 years for spruce-dominated systems to reach pre-harvest carbon levels (i.e. break-even) at the stand-level. Using pellets made from logging slash to replace coal creates immediate net carbon benefits to the atmosphere of 17–21 tonnes C ha^{-1}, shortening these break-even times by 9–20 years and resulting in an instant carbon break-even level on stands most severely impacted by the beetle. Harvesting pine dominated sites for timber while using slash for bioenergy was also found to be more carbon beneficial than a protection reference scenario on both stand- and landscape level. However, harvesting stands exclusively for bioenergy resulted in a net carbon source unless the system contained a high proportion of dead trees (>85%). Systems with higher proportions of living trees provide a greater climate change mitigation if used for long lived wood products.

Keywords: British Columbia, carbon break-even, carbon debt, carbon parity, CBM-CFS3, climate change mitigation, forest biomass, Mountain Pine Beetle, temporal carbon analysis, wood pellets

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Introduction

Bioenergy has been promoted via various policies around the globe as a promising option to reduce the dependence on fossil fuels (REN21, 2012). In many of these policies, e.g., the European Union's Emission Trading Scheme, forest bioenergy has been considered carbon neutral in the energy sector, since forest carbon stock changes were reported as part of the forest ecosystem, an assumption criticized in recent analysis and debate (Fargione *et al*., 2008; Johnson, 2009; Melillo *et al*., 2009; Searchinger *et al*., 2009; Searchinger, 2010; Schulze *et al*., 2012). Apart from neglecting emissions connected to potential (indirect) land-use changes and fossil fuel use along the value chain, the carbon neutrality argument was implicitly related to short carbon cycles, i.e., an almost immediate uptake via plant re-growth of the initially released plant carbon. While such assumptions may hold true for some (multi-) annual cropping systems, the removal for burning of large amounts of woody biomass from decades- to centuries-old forest takes many years to regrow. For these reasons, it is critical to understand regional differences in forest growth and related natural cycles to define efficient mitigation policies in a carbon-constrained world.

While forests are important both as carbon sinks and for the large amounts of carbon they store (Pan *et al*., 2011), they are also subject to widespread mortality induced by environmental factors (e.g., Anderegg *et al*., 2013), and natural disturbance (Kurz *et al*., 2008a,b). In western North America, an infestation with mountain pine beetle (MPB), *Dendroctonus ponderosae Hopkins* has killed an estimated 710 Mm^3 of pine volume across a 17 Mha region in British Columbia (BC) (BC-MoF, 2012c). This unprecedented outbreak has affected current and future harvest practices, potential use for forest products, and the carbon balance. As a result of the MPB outbreak, harvesting in interior BC nowadays includes extensive salvage of standing dead trees

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(BC-MoF, 2010a), as long as the stem snags are still considered 'merchantable' for timber. This 'shelf-life' (Thrower *et al.*, 2007) of the standing dead trees decreases at varying rates depending on, among other factors, stand type, soil moisture conditions, and sawmill technology (Lewis *et al.*, 2006; Trent *et al.*, 2006; BC-MoF, 2008). The scale of the salvage area and the lower level of established forest infrastructure across Canada, as compared to Europe, mean that salvage operations are unable to access stands across the large area affected to target specific, individual cut blocks where merchantability as sawlogs (the predominant standard use of these forests) would drop fastest (Nelson *et al.*, 2012). Thus, the proportion of non-merchantable (for sawlogs) standing dead trees which contribute to the pool of harvesting residues is bound to increase over time (Dymond *et al.*, 2010; Nelson *et al.*, 2012).

In BC, primary (harvesting) residues, i.e., tops and branches of merchantable plus whole non-merchantable trees (including deadwood), are typically piled and burnt at the roadside to remove a source of fuel for forest fires (FPB, 2008; Hall, 2011; Nelson *et al.*, 2012; Sutherland *et al.*, 2012), as well as to allow the area where residues are piled to be regenerated. Slash pile burning contributes notably to the annual greenhouse gas (GHG) emissions associated with the forest sector (Dymond & Spittlehouse, 2009; Hall, 2011). From a climate mitigation perspective, it is therefore interesting to investigate the temporal carbon balance of using slash (including insect-damaged wood) instead as pellet feedstock and how this impacts the overall carbon balance in MPB affected forests.

Several valuable studies have modeled forestry related carbon cycles in relation to bioenergy use starting with Marland & Schlamadinger (1995) and Schlamadinger *et al.* (1995). Some, including those at the centre of the recent policy discussion in Europe (MANOMET, 2010; Zanchi *et al.*, 2010; Greenpeace, 2011), only consider forest carbon dynamics of single harvested stands. Some studies indicate that carbon dynamics over several harvest plots provide a more realistic appraisal of harvest effects on landscape level (Schlamadinger & Marland, 1996b; Marland *et al.*, 1997; Mitchell *et al.*, 2012; Eliasson *et al.*, 2013). Only a few take carbon stored in end-products into account (Schlamadinger & Marland, 1996a,b, 1999; Marland *et al.*, 1997; Hennigar *et al.*, 2008; Ingerson, 2011; Earles *et al.*, 2012; Mitchell *et al.*, 2012) and even fewer consider GHG emissions along the connected value chains (Hudiburg *et al.*, 2011; McKechnie *et al.*, 2011). None has yet analyzed the impact of varying insect-damage on temporal carbon cycles against different baselines, i.e., regular timber harvest or site protection. Mitchell *et al.* (2012) modeled temporal carbon for pine beetle infested stands harvested for bioenergy, however, their model calibration reflects US harvest practices, and do not cover a specific supply chain or distinguish impacts by feedstock. McKechnie *et al.* (2011) provided an analysis for wood pellets from and co-fired in Ontario, Canada, i.e., for forests significantly different than those in BC and used in end-markets with different life-cycle emissions than those in Europe.

Our present study goes beyond previous forest carbon analyses and life-cycle assessments of BC wood pellets (Damen & Faaij, 2006; Magelli *et al.*, 2009; Sikkema *et al.*, 2010), to analyze the temporal carbon balance of BC wood pellets against varying baselines, and simulates the impact of different levels of insect-damage on carbon break-even times at the stand and landscape levels. Our carbon analysis considers forest growth, harvest, storage in end-products, additional fossil fuel emissions along the wood pellet supply chain from BC producers to large-scale users in North-Western Europe, and the displacement of fossil fuels in Europe. We do not model the existing forest landscape or recent and projected changes in harvest rates.

Materials and methods

Carbon break-even and parity points

Forest carbon dynamics, long-lived wood product stocks and emissions, and supply chain emissions are used to calculate carbon break-even and parity points. We define the carbon break-even point as the amount of years the carbon level of the harvested forest area and related wood and bioenergy products is lower than the pre-harvest carbon level ($C_{S,\text{break-even(t)}} < 0$), taking into account forest carbon plus carbon stored in end-products, displaced fossil fuels, and supply chain emissions:

$$\begin{aligned} C_{S,\text{break-even(t)}} &= \sum_{i=1}^{n} C_{S,\text{pool(i)},t} - \sum_{i=1}^{n} C_{S,\text{pool(i)},0} \\ &+ \sum_{h=1}^{m}\sum_{p=1}^{u} C_{S,\text{harvest(h)},p} \times \eta_{\text{biomass},p} + \sum_{h=1}^{m}\sum_{p=1}^{u} C_{S,\text{harvest(h)},p} \\ &\times \mu_p - \sum_{c=1}^{v} E_{S,\text{LCA},c} \end{aligned} \quad (1)$$

$C_{S,\text{break-even(t)}}$: carbon balance in year t for the scenario S
$C_{S,\text{pool(i)},t}$: above- and belowground forest carbon of pool i of n in year t for the scenario S
$C_{S,\text{pool(i)},0}$: above- and belowground forest carbon of pool i of n in year 0 i.e., pre-harvest for the scenario S
$C_{S,\text{harvest(h)},p}$: carbon mass taken out during harvest h of m for product p of u for the scenario S, taking into account carbon losses from harvested log to finished wood product
$\eta_{\text{biomass},p}$: displacement factor for product p
μ_p: half-life factor for product p (takes into account biogenic GHG emissions for long-lived wood products) for the scenario following the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006)

$E_{S,LCA,\ c}$: fossil fuel emissions along the supply chains for the scenario S
t: simulation year
Note: Break-even times are also calculated for protected, unharvested areas. In this specific case however, $C_{S,harvest(h),p}$ and $E_{S,LCA,\ c}$ equal zero.

Mitchell *et al.* (2012) refer to this point as the carbon debt repayment time, although their analysis does not cover life-cycle emissions. By referencing the stand to its original condition, we take into account an initial 'carbon debt', occurring through harvest. This, however, does not entail a land-use change, i.e., varies from the definition in Fargione *et al.* (2008), since the land in BC will remain forested. It also differs from the definition by MANOMET (2010) who refer to the differential between bioenergy and fossil fuel reference combustion (see Lamers & Junginger, 2013, for an extended discussion on terminology).

In contrast to most aforementioned temporal carbon analyses in forestry, Mitchell *et al.* (2012) acknowledge the importance of comparing carbon levels in harvested sites to a reference case. The authors applied this, 'sequestration parity' concept, to an unmanaged, defined stretch of forest that is subjected to naturally occurring wildfire. The use of such a reference was also relevant for our analysis (called '*Protection*') especially as carbon stocks in the unmanaged forest are changing with time due to the pine beetle infestation. However, we also included a reference to a timber harvesting baseline (called '*Business-As-Usual*'). The time to reach carbon parity is defined as the amount of years the carbon level of the harvested forest area and related wood and bioenergy products in the scenario of interest is lower than the carbon level of the reference scenario ($C_{S,R,parity(t)} < 0$):

$$
\begin{aligned}
C_{S,R,parity(t)} = {} & \sum_{i=1}^{n} C_{S,pool(i),t} - \sum_{i=1}^{n} C_{R,pool(i),t} \\
& + \left(\sum_{h=1}^{m} \sum_{p=1}^{u} C_{S,harvest(h),p} \times \eta_{biomass,p} \right. \\
& \left. + \sum_{h=1}^{m} \sum_{p=1}^{u} C_{S,harvest(h),p} \times \mu_p - \sum_{c=1}^{v} E_{S,LCA,c} \right) \\
& - \left(\sum_{h=1}^{m} \sum_{p=1}^{u} C_{R,harvest(h),p} \times \eta_{biomass,p} \right. \\
& \left. + \sum_{h=1}^{m} \sum_{p=1}^{u} C_{R,harvest(h),p} \times \mu_p - \sum_{c=1}^{v} E_{R,LCA,c} \right)
\end{aligned} \tag{2}
$$

$C_{S,R,parity(t)}$: carbon balance in year t for the scenario S to reference case R
$C_{S,\ or\ R\ pool(i),t}$: above- and belowground forest carbon of pool i of n in year t for the scenario S or reference case R
$C_{S,\ or\ R\ harvest(h),p}$: carbon mass taken out during harvest h of m for product p of u for the scenario S or reference case R
$\eta_{biomass,p}$: displacement factor for product p
μ_p: half-life factor for product p (takes into account biogenic GHG emissions for long-lived wood products) following the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006)
$E_{S\ or\ R,,LCA,c}$: fossil fuel emissions along the supply chains for the scenario S or reference case R
t: simulation year

Study area

The pellet operations in focus, located in the cities of Prince George and Vanderhoof, BC, draw their feedstock, i.e., predominantly sawdust, from surrounding timber mills. The respective mills currently source timber within the Timber Supply Areas (TSA) of Quesnel and Prince George (BC-MoF, 2011a) (Fig. 1). While allowing for future supply base variations, we assume that timber will not be sourced beyond 300 km distance from either Prince George or Vanderhoof, or elevations above 1300 m. This limits the study area to the biogeoclimatic regions (areas with similar climate and vegetation) of the Sub-Boreal Spruce (SBS) and Sub-Boreal Pine-Spruce (SBPS) zone (Wong *et al.*, 2003); both within the Montane Cordillera, a fire-dominated ecosystem. The forests are dominated by coniferous trees and are of relatively low tree species diversity (Fig. 2).

Modeling tools and assumptions

Carbon dynamics. Forest carbon dynamics were simulated via the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) (Kurz *et al.*, 1992, 2009; Kull *et al.*, 2011). Its key advantages over other temporal carbon model options (Schlamadinger & Marland, 1996b; Masera *et al.*, 2003; Woodbury *et al.*, 2007; Chen *et al.*, 2010; Tuomi *et al.*, 2011) in the context of this study are that CBM-CFS3 explicitly allows the simulation of MPB infestation (see *Forest management assumptions* Section for details), and has a high regional accuracy for the study region. It applies sophisticated, ecozone specific algorithms (Boudewyn *et al.*, 2007) to convert woody biomass volumes to carbon. Its simulations cover tree growth, litterfall, and ecozone specific turnover and decay rates allowing annual descriptions of carbon stocks (Kurz *et al.*, 2008a).

Site definitions. Four different sites were defined to represent the forest conditions and insect-damage of the study area (Table 1). The Pine-dominated (Pd) site represents the vast majority of current, salvage focused, harvests in the study region (>70% pine) (BC-MoF, 2007, 2010b). Given official timber supply analysis, sites with Pd characteristics will dominate the harvest portfolio until 2020 (BC-MoF, 2010b,c). The Spruce-dominated (Sd) site reflects the composition (<30% pine) across the greater part of the Timber Harvesting Land Base (THLB) in the Prince George TSA (BC-MoF, 2011b) (Fig. 2); and thus the expected harvest portfolio post 2020. Tree species composition on both sites were matched with values reported by foresters interviewed for this study (Bysouth, 2012; Perdue, 2012). As part of a sensitivity analysis, we also simulate stand mixes for a mere Spruce-Fir (SF) (0% pine) and Pine-only (Po) site (100% pine). While these represent the theoretical lower and upper pine and thus deadwood volumes, sites with SF characteristics are common across the Prince George and Fort St. James Forest District (FD), and sites with Po-composition occur in the Vanderhoof FD and western Quesnel TSA (FPB, 2007b; BC-MoF, 2010c, 2011b,c). Due to the limited shelf-life of MPB infested wood, Po-sites would be targeted as part of harvesting operations before 2020, while non-affected SF-sites will rather be part of harvest compositions post 2020. The spruce-fir-ratio across

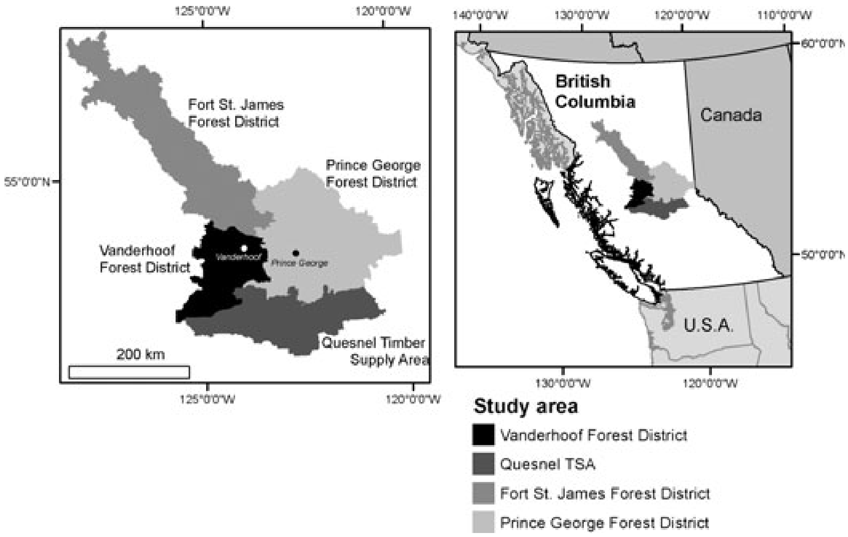

Fig. 1 The study area, encompassing the timber supply areas of Quesnel and Prince George. The latter area is split into the forest districts of Vanderhoof (17%), Fort St. James (40%), and Prince George (43%).

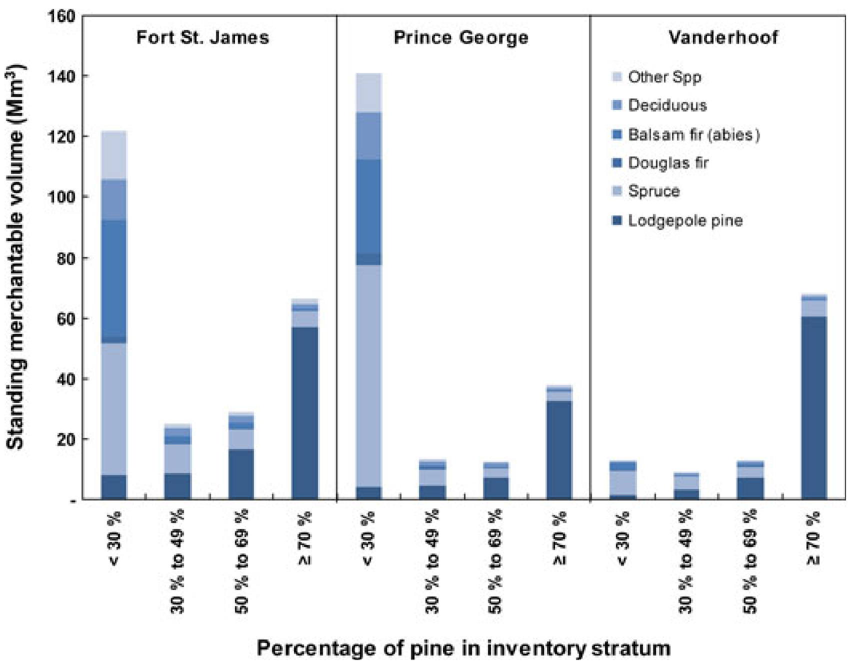

Fig. 2 The standing merchantable volume by tree species across the timber harvesting land base (THLB) of the Prince George timber supply area (TSA), stratified by the percentage of pine in the forest inventory, shows a 37% overall frequency of stands with >70% pine share. Within the individual forest districts (FD), the frequency is 71% for Vanderhoof and 26% for both Prince George and Fort St. James. Across Quesnel TSA, 73% of the THLB is made up of stands >70% pine.

all sites (except Po) matches average conditions as reported by interviewed foresters.

Timber harvest in interior BC started only around 1958 (Price *et al*., 1996) implying that currently harvested forest areas are in 'natural condition' and not replanted, i.e., not previously harvested stands. Their age structure varies with the historical wildfire return rate, which differs significantly between ecozones (Wong *et al*., 2003). Empirical data by interviewees showed that stands currently harvested typically fall within the 100–140 age range, of which the average was used for our simulations. The wildfire return frequency was set to 150 years, which matches the CBM-CFS3 default for the Montane Cordillera ecozone and average data as reported by foresters in the study region. Ranges given for stand-initiating events

Table 1 Characteristics and calibration details per simulated forest site

	Pine-dominated (Pd)	Pine only (Po)	Spruce-dominated (Sd)	Spruce-Fir (SF)
Characteristics	Current salvage logging activities	Upper pine limit to Pd	Majority of THLB	Lower pine limit to Sd
Regional scope	Vanderhoof Forest District Quesnel Forest District		Prince George Forest District Fort St. James Forest District	
Temporal scope	Current harvest; until 2020		Future harvest; post 2020	
Stand mix (dominant species in bold)				
Lodgepole pine (*Pinus Contorta var. latifolia*)	**73%**	**100%**	26%	–
Hybrid spruce (*Picea glauca x engelmannii)*	22%	–	**59%**	**80%**
Subalpine Fir (*Abies lasiocarpa*)*	5%	–	15%	20%
Site Index				
Natural stand	18.25	18.25	17	17
Managed stand	20.5	20.5	21.5	21.5
Stand condition				
Deadwood share (of merchantable volume)	62%	85%	22%	–
Pine kill rate (of total pine volume)	85%	85%	85%	–

The Pine-dominated (Pd) site represents the vast majority of current, salvage focused, harvests in the study region (>70% pine); valid until 2020 (BC-MoF, 2007, 2010b). The Spruce-dominated (Sd) site reflects the composition (<30% pine) across the greater part of the Timber Harvesting Land Base (THLB) in the Prince George TSA and thus the expected harvest build-up post 2020. Tree specie distributions on both sites were matched with values reported by foresters interviewed for this study. Site index data is based on empirical values provided by Farnden, 2004, 2006; Coates *et al.*, 2009.
*Subalpine Fir is locally referred to as 'balsam fir'.

by Wong *et al.* (2003) for SBS and SBPS are in the order to 100 and 125 years. Hence, our assumption can be regarded as conservative, since it leads to a higher average carbon level in undisturbed (e.g., protected) forests.

Target harvest volumes in the Prince George TSA range between 300–350 m^3 ha^{-1} (Bysouth, 2012; Vinnedge, 2012). Forests undergoing natural regeneration take longer to reach this level compared to planted sites (Fig. 3) (Sutherland *et al.*, 2012). This implies that future harvests on replanted sites will be in the time window between 50–70 years (BC-MoF, 2011b), when growth curves have reached their largest annual increments, depending on site productivity and species mix (Fig. 3) (Sutherland *et al.*, 2012). We use a harvest interval of 60 years.

Seventy percent of all pine in the BC interior is projected to be killed by 2017 under the worst case scenario (BC-MoF, 2010b). Spatial data for the study area shows that the vast majority of stands is severely affected (>71% killrate) (BC-MoF, 2011a, 2012b). The variations seem to be connected to the age structure of the stands. MPB prefer mature pine trees (>60 years) (BC-MoF, 2008), whereas even mature pine can survive MPB attack (Safranyik & Wilson, 2006; FPB, 2007a) suggesting that in affected stands a portion of trees survive. Current salvage logging, as harvest in general, targets accessible older stands first. These will generally be more heavily infested (>71%). Also, the lack of remaining mature pine in the study area and the sheer amount of MPB has lead to the infestation of immature pine in recent years (BC-MoF, 2010b). In our scenarios, we apply a pine kill rate (amount of total merchantable and unmerchantable pine volume killed by the MPB in a stand) of 85%, reflecting the average of the spatial information range (71–100%).

Tree growth curves. CBM-CFS3 requires external data on net merchantable tree volumes for the simulated sites. These were generated via the Variable Density Yield Prediction (VDYP) model for unmanaged, natural stands, and via the Table Interpretation Program for Stand Yield (TIPSY) (Mitchell, 1975), a timber planning software, for managed, i.e., replanted stands (Fig. 3). The models, provided by the BC Ministry of Forests, Lands, and Natural Resource Operations (BC-MoF), are empirical yield prediction systems building on regionally specific tree growth curves. The calibration was based on empirical data (gathered via questionnaires sent to BC forest companies), BC government default values (e.g., merchantability requirements), and growth functions (Goudie, 1984; Thrower, 1994; Chen & Klinka, 2000). Site-indexes were taken from recent empirical analyses (Farnden, 2004, 2006; Coates *et al.*, 2009).

Forest management assumptions. CBM-CFS3 enables the user to transition sites to different yield curves depending on the disturbance regime. Post-harvest, we transition the site from an initial natural tree growth curve to a managed yield curve. Post-beetle attack, non-affected trees continue growing along their respective natural growth curve. The area under beetle attack however is transitioned back to age zero of the natural growth curve, assuming natural regeneration. Regeneration delay in MPB affected stands was not explicitly modeled as it is assumed to be sufficiently covered by the low initial increments of natural growth curves (Fig. 3). Also, empirical analyses of stands infected several decades ago by MPB did not reveal a disproportionate regeneration delay (FPB, 2007a; Coates *et al.*, 2009; Brown *et al.*, 2012; Hawkins *et al.*, 2012).

Assumptions on standing dead tree shelf-life (15 years) and snag fall times (majority post 20 years) in the official Prince George and Quesnel timber supply prognosis (BC-MoF, 2010b,c) appear optimistic. Empirical data for the biogeoclimatic zone shows that while most trees remained standing by year 8 post

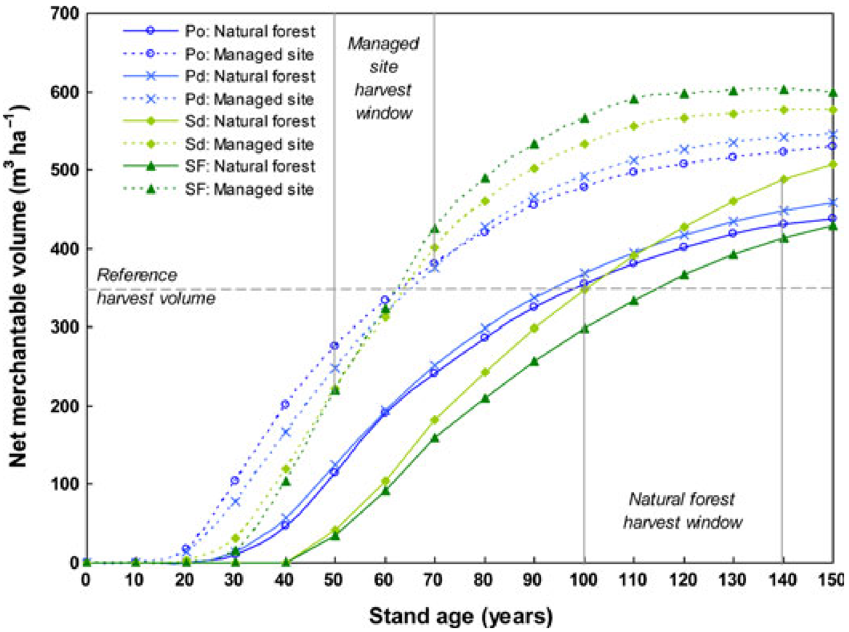

Fig. 3 Tree growth curves for the modeled sites. Empirical observations by Farnden (2004, 2006) and Coates *et al.* (2009) indicate a faster initial tree growth for replanted as compared to natural forest sites. This implies that reference harvest volumes in the region (350 m^3 ha^{-1} of net merchantable tree volume with >12.5 cm diameter at breast height) are reached earlier on managed sites. The vertical lines indicate the current harvest window of natural forest (100–140 years of age) and future managed/replanted sites (50–70 years).

attack, snag fall increases in the following years (Lewis & Thompson, 2011). Current harvest data from the study region shows a reduction of 20–30% of merchantable tree volume in stands 8–10 years post beetle attack (Bysouth, 2012; Perdue, 2012). We reduced the CBM-CFS3 default snag fall rate of 3.2% for the Montane Cordillera accordingly to 2.8%. The CBM-CFS3 carbon pool transfers (disturbance matrixes) for MPB infestation and wildfire were not changed. However, harvesting matrixes (Table S1–S3) were updated with empirical harvest levels (for green and dead tree volumes) provided by local foresters (Bysouth, 2012; McCormack, 2012; Perdue, 2012; Vinnedge, 2012).

End-use modeling. Post-harvest calculations, including carbon break-even/-parity, were done outside CBM-CFS3. Harvest volumes are allocated to three carbon pools via estimates given by BC timber mills (which supply feedstock to BC pellet factories), taking woody biomass and thus carbon losses across the value chain into account: long-lived timber products, short-lived products (pulp and paper), and wood pellets (Table S4). Over their whole life-span, paper and pulp products are sometimes regarded to be net carbon emitters (Ingerson, 2011). Most BC paper products although are used and disposed of in North America, where carbon storage in landfills is higher than in other world regions (Earles *et al.*, 2012). To limit the uncertainty linked to short-lived products, we did not account for carbon storage in pulp and paper products, or in landfills and consider carbon in these products as fully emitted to the atmosphere. The temporal carbon storage in wood pellets is also ignored since they are combusted almost immediately. To measure carbon storage in timber products over time, we follow the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) and apply the half-life factor approach (Skog, 2008; Earles *et al.*, 2012). We believe this reflects the temporal aspect of the carbon discussion more appropriately than the key alternative, i.e., the 100 year approach (Ingerson, 2011). Modeling by Dymond (2012) showed that IPCC default timber half-life values of 30 years are lower than is the case for BC wood product consumption in North America, but nevertheless provides a reasonable estimate of biogenic GHG emissions over the whole life-cycle when it includes disposal.

In addition to carbon storage, we account for fossil fuel substitution effects which are cumulative and not discounted over time (see also Schlamadinger & Marland, 1996b). Wood pellets directly replace hard coal in North-Western Europe. Our displacement factor of 0.923 for coal is based on Damen & Faaij (2006) and Jonker *et al.* (2013) who analyzed pellet supply streams from Canada and the USA to the same power plants in Europe (Table S5). BC timber substitutes more fossil fuel intensive materials, almost exclusively in construction in North America (Dymond, 2012). A review by Sathre & O'Connor (2008) calculated a global average displacement factor of 2.0 for building products. We took a more conservative approach and applied 1.7, which was the median of studies from North America within the Sathre & O'Connor (2008) review (Table S6). Fossil fuel value chain emissions are based on Majer & Liemen (2012) who investigated specific wood pellet supply streams from the study region to Europe (Table S7). Their values are generally in-line with previously published literature such as Sikkema *et al.* (2010) and Magelli *et al.* (2009), although the latter calculated higher emissions for ocean transport (Table S8).

Scenarios

No harvest: Protection (P). In contrast with harvest scenarios, protecting a site implies no human intervention (at any given point). Protected areas though are still exposed to natural disturbances, primarily insects and wildfire (Mitchell *et al*., 2012). Lodgepole pine, the key MPB host, is a fire-maintained species; although it may also regenerate without fire disturbance (Shore *et al*., 2006). The age distribution of lodgepole pine, and therefore the spatial and temporal susceptibility of forests to the MPB, is determined by the frequency of stand replacing disturbances. Without harvesting, this frequency is primarily defined the mean fire return interval and fire size (Shore *et al*., 2006). Therefore, we thought it was appropriate to include wildfire in the simulation of the *Protection* scenario. However, we determined that by modeling a single stand that was burned, the simulation would create varying carbon break-even or parity times depending on the year the wildfire takes place in the simulation. To avoid this effect, we modeled a theoretical landscape, on which each consecutive year one hectare (ha) is burnt, while the remaining keep (re-) growing. This approach is consistent with other theoretical harvest simulations on landscape level (Schlamadinger & Marland, 1996b; Marland *et al*., 1997; Mitchell *et al*., 2012; Eliasson *et al*., 2013). It implies that, per site, a 150 ha area was defined with each hectare having a different age from 1 to 150 years, with no hectare reburning sooner than 150 years. The *Protection* reference scenario is the average per hectare over the entire area. For each site an individual *Protection* scenario was modeled.

Pellets from sawdust: Business-As-Usual (BAU). The *Business-As-Usual (BAU)* scenario describes the typical harvesting practices in the study region, as gathered via questionnaires. It encompasses clear-cutting including salvage logging of merchantable dead trees. The fraction of the latter currently averages 70% in the study region, 8–12 years post beetle attack. Harvesting residues (slash) are piled at the roadside and burnt to reduce fire hazard. The amount of slash burn is ecozone specific (Kurz *et al*., 2012) and volumes identified for the coast (Hall, 2011) are assumed to be lower than in the wildfire dominated interior. The CBM-CFS3 default for the Montane Cordillera, however, tends to burn significant amounts of slash. We hence applied an average between these and the slightly lower values in Hall (2011). In the *BAU* scenario, wood pellets are only derived from secondary i.e., mill/processing residues, which otherwise would be combusted (in beehive burners with no energy recovery) or landfilled. The *BAU* is compared to the no harvest (*Protection*) reference scenario.

Pellets from slash: Slash Use (SU). In the *Slash Use* scenario, non-merchantable (green and dead) trees plus tops and branches of merchantable trees are chipped and directly transported to the pellet factory as feedstock (in addition to sawdust). The recoverable fraction of the otherwise burnt slash volume is reduced by handling losses during chipping. We base this fraction on empirical data for the Quesnel district (Friesen & Goodison, 2011), matching assumptions made by Titus *et al.* (2009) and Dymond *et al.* (2010). The primary reference scenario to *Slash Use* was slash burn, i.e., *BAU*, but we also compare it to a *Protection* scenario.

Pellets from salvaged dead trees: 1st harvest for pellets (1st hfp). Roundwood harvest (green tree) for pellet production has become common practice at new, large-scale plants, e.g., in the USA and Russia (Cocchi *et al*., 2011; Lamers *et al*., 2012). However, green trees are not harvested for pellets in BC and are highly unlikely to be used this way in the future, primarily due to the higher economic value of using the trees for lumber (Bysouth, 2012; Cornwell, 2012; Perdue, 2012; Sutherland *et al*., 2012). Current pellet exports from Western Canada are largely residue based. The expected future decline in lumber harvest, however, will also entail a decline in residue volumes, thus making an expansion of merely residues based pellet production improbable (Fig. S12). A mid-term timber supply gap is expected as previously salvaged stands will be in their re-growing stage and more deadwood has turned non-merchantable, rendering unsalvaged stands uneconomic to harvest (BC-MoF, 2010b,c, 2011b,c, 2012a; Prasad, 2012). The latter will especially be the case for sites with (very) high percentage of dead pine and (very) small trees (Cornwell, 2012). Given that these sites may have insufficient stocking (and natural regeneration) to grow a future crop for timber, widening the supply gap, the BC Government developed the Innovative Timber Sale License for cruise-based (lump sum) timber sales targeting areas in interior BC impacted by MPB and wildfires that will not be regenerated as a result of harvesting activities (Cornwell, 2012). Bidding for such plots is additionally incentivized as the BC Government (under the Forests-For-Tomorrow program) carries reforestation costs. Thus, fiber otherwise uneconomic to harvest and likely destined to decay can be recovered and prepared planting ground is provided for little cost to the BC Government (Cornwell, 2012).

Under these circumstances, dedicated salvage logging of uneconomic timber sites to derive pellet feedstock from roundwood could take place, i.e., as the 'first harvest' in our scenario (*1st harvest for pellets*). The reference scenario in this case is no harvest (*Protection*). Replanting increases the future timber supply in the model and the eventual (second and following) harvests of these replanted sites were simulated as generating sawlog timber and pellet feedstock as under the *Slash Use* scenario (primary and secondary residues). Sites with less than 70% pine, i.e., Sd and SF, have an insufficient amount of deadwood, and so will not fall under this category (of dedicated salvage logging) due to having a high amount of future live tree volume for harvest. We have included the *1st harvest for pellets* scenario on these sites only to provide a sensitivity analysis.

Results

Detailed carbon (C) dynamics are illustrated exemplarily for a pine-dominated (Pd) site undergoing regular timber harvest (*BAU*) in Fig. 4a and c and, as a reference, no harvest (*Protection*) in Fig. 4b and d (the remaining sites and scenarios are provided in Supporting Information Figs S4–S11). Simulations start in 1992 (i.e., year −20) when live biomass C increased proportionally to the

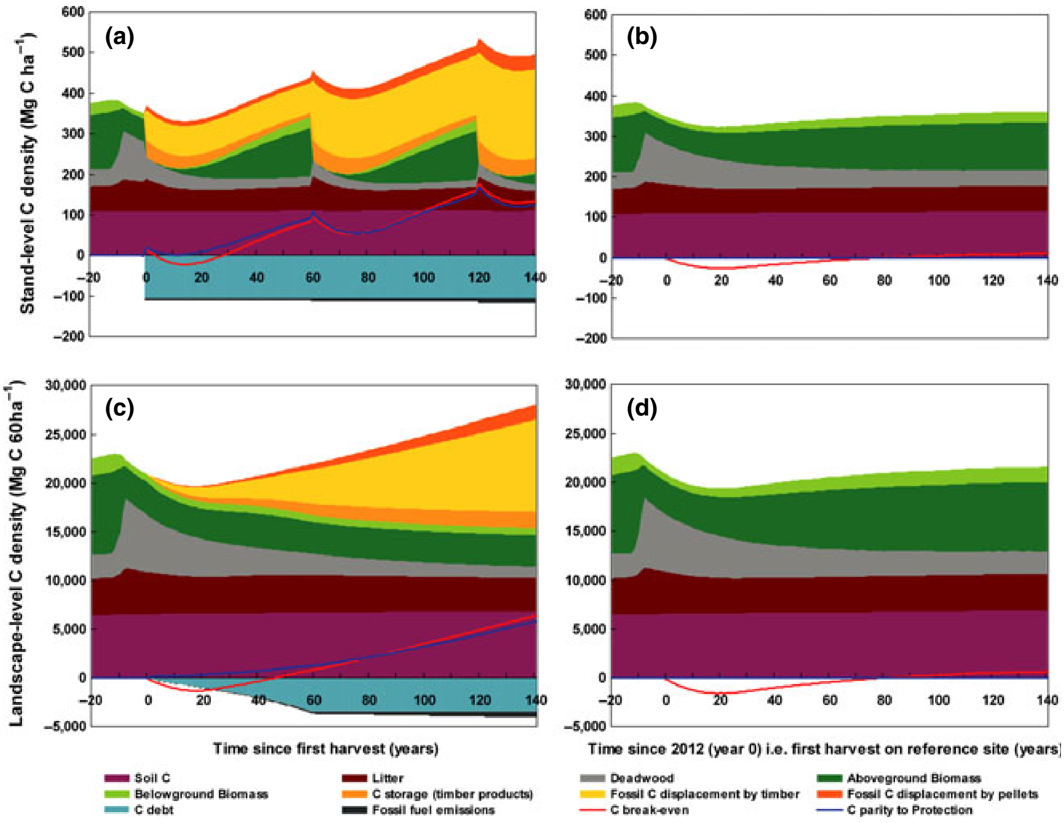

Fig. 4 Temporal carbon dynamics among forest and wood product pools, including displacement factors for solid wood products and bioenergy for a pine-dominated (Pd) site undergoing regular timber harvest (*BAU*) at the stand- (a) and landscape (c) level and a forest *Protection* scenario at the stand- (b) and landscape (d) level. The first harvest takes place in 2012, equaling simulation year 0. Simulations start 20 years prior to account for MPB infestation between 2000 and 2004 (simulation years −12 to −8).

natural yield curve (Fig. 3) up to the time of the MPB outbreak. The MPB epidemic across the study region started in 2000 and peaked in 2004 (BC-MoF, 2010b,c, 2011b,c) (simulation years −12 to −8), after which the pine mortality has reached its maximum prescribed level (Table 1). The first possible harvest takes place in 2012, i.e., simulation year 0. Additional harvests occur every 60 years for each managed stand.

At the stand level (Fig. 4a and b), mortality from the MPB reduced living biomass stocks and increased dead wood and litter stocks from year −12 to 0. In 2012 (year 0) the site was harvested for timber and slash was burned, reducing living biomass and dead wood stocks, creating a C debt in the forest (relative to pre-harvest C stocks), which was further increased by transportation and manufacturing (Fig. 4a). Most of the harvested C (timber) was converted into long-lived wood products which store C and displace more energy intensive building products. Mill residues (sawdust) were simulated as if pressed into pellets and replacing coal in European power plants. After replanting, the model regrows the site using the respective yield curve for managed forests until year 59 (Fig. 3). Total C stocks decline with time due to decomposition of dead wood and litter and the end of the life-span of timber products. The system has a positive C balance ($C_{\text{break-even}}$ line, calculated via formula 1) immediately following harvest, which lasts for 3 years until some of the deadwood, litter, and wood products decompose. It remains permanently positive 28 years after harvest, thus having a net $C_{\text{break-even}}$ point of 25 years. C_{parity} (calculated via formula 2) relative to the *Protection* scenario (Fig. 4b) exists throughout the simulation because a large amount of deadwood on the site in year 0 was used to offset coal emissions under *BAU*, rather than decomposing over time and, at least partially, returning to the atmosphere (under the *Protection* scenario). With the second harvest in year 60, the C debt does not change as only regrown forest is harvested, fossil fuel emissions increase however with value chain operations. During the simulated period, the stand never reaches the pre-beetle forest C stock of year −20.

On a theoretical landscape of 60 ha with a 60-year harvest rotation (Fig. 4c), 1 ha is harvested and replanted per year, and the first replanted hectare/site is harvested again after 60 years. As the entire 60 ha of modeled forest was assumed to be established after wildfire, the area is even aged. The MPB attacks the entire 60 ha landscape at once and reduces the live pine tree biomass on all non-harvested hectares over time. The deadwood pool created by the MPB outbreak decays slowly over time and is also reduced by salvage logging operations each year. Over time, dead trees are less usable for timber production and are increasingly burnt as part of the post-harvest operations. The combination of the landscape, products, and substitution benefits reaches the forests' pre-harvest C level, i.e., $C_{break\text{-}even}$, after 47 years. As with the stand-level results, C_{parity} determined relative to the *Protection* scenario (Fig. 4d) is reached immediately (year 0). The C debt within the forest itself increases as different stands are harvested by the model. The C debt stabilizes in year 60 when the first re-planted hectare is harvested again and ongoing harvest/replanting means the forest reaches a uniform age structure with 1 ha of forest of each age from 1–60 years.

Stand-level scenarios

Current timber harvesting practices across the study region (*BAU*), i.e., when slash is burnt and only secondary (sawmill) residues are used for pellet production, require between 20–25 years for beetle-impacted pine and 37–39 years for spruce-dominated sites to reach pre-harvest C levels ($C_{break\text{-}even}$) (Fig. 5a, Table S9). Using pellets made from sawdust for energy shortened the break-even period by 5–7 years. Using slash for pellet feedstock instead of burning slash in the forest without energy production creates direct C benefits, shortening break-even points by 9–20 years, and resulting in net zero-year break-even times for pine-only (Po) sites (Fig. 5a). In general, $C_{break\text{-}even}$ points are shorter for sites with a higher proportion of pine, as less C is extracted during harvest, i.e., more deadwood remains on-site, and planted pine grows faster over the first 50 years than the respective replanted spruce-leading sites.

The displacement and C storage effects of long-lived timber products outweigh coal substitution in our analysis. This can be seen in the longer $C_{break\text{-}even}$ periods for pellet harvest as under the *1st harvest for pellets* scenario compared to the combined harvest for timber and pellet feedstock (as in the *BAU* or *Slash Use* scenarios) (Fig. 5a). This is due to the higher C storage benefit and the higher displacement factor of long-lived timber products (1.7) compared to coal (0.932).

C_{parity} when scenarios are compared to a *Protection* reference is immediately achieved under *BAU* for both pine-dominated sites, but takes 37–54 years for spruce-dominated sites (Fig. 5b, Table S9). Salvage logging for pellet feedstock (*1st harvest for pellets*) compared to forest *Protection* entails net C loss periods of 16–30 years

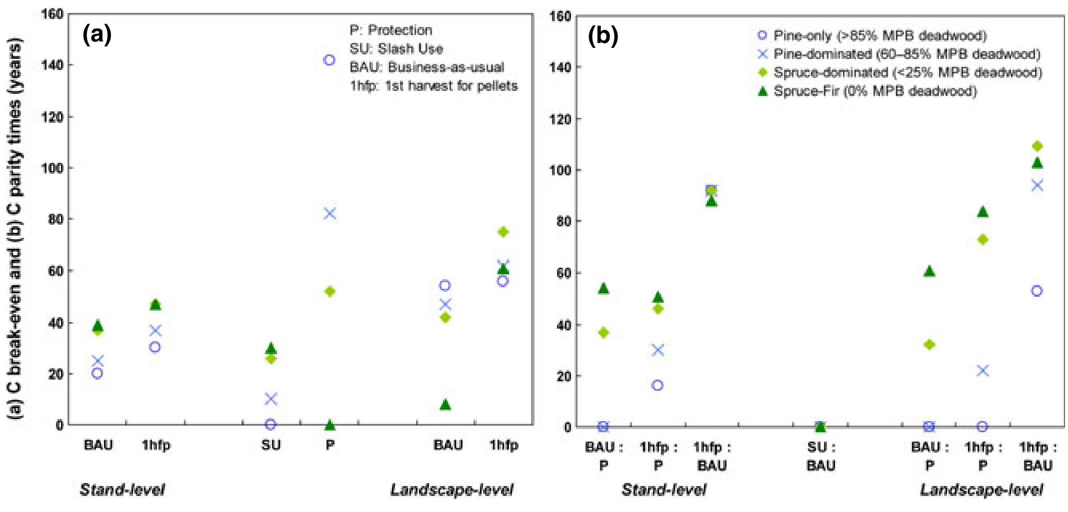

Fig. 5 Overview of $C_{break\text{-}even}$ (a) and C_{parity} (b) times for all sites and scenarios. $C_{break\text{-}even}$ (a) describes the net years required for systems to reach preharvest carbon stocks including storage in the ecosystem, products, and displacement factors. C_{parity} (b) shows the net years required for systems to reach a greater carbon stock (including storage in the ecosystem, products, and displacement factors) than a particular reference scenario. For C_{parity} (b), the first scenario is the one of interest. The second one is the reference. $C_{break\text{-}even}$ times on stand-level are shortest for pine-dominated sites due to faster tree growth (as compared to spruce); the higher share of decaying deadwood though on landscape level reduces future timber harvest on pine-dominated sites, increasing their break-even times compared to spruce-leading sites. The long time it takes protected pine-leading sites to recover post beetle-attack though shortens the periods until C_{parity} is reached for any alternative scenario, e.g., *BAU* or *1st harvest for pellets*. $C_{break\text{-}even}$ values for *Slash Use* and *Protection* are the same on stand- and landscape level.

for pine and 46–51 years for spruce-dominated stands. The major influencing factors for C_{parity} toward *Protection* are the amount of deadwood decaying or burning due to wildfire in the reference case (higher for pine-leading sites), and the stable greenwood pool on spruce-dominated sites.

Comparing *Slash Use* to a *BAU* reference, an immediate C benefit is achieved, i.e., C_{parity} is reached in year zero. This benefit slowly drops over the rotation period as litter in the *BAU* case gradually decays, reducing its offsetting emission benefits relative to slash use for bioenergy (Fig. S2). The average benefit of 17–21 tonnes C ha^{-1} (Table 2) varies between sites and is highest for spruce-dominated plots due to the higher total green tree harvest and thus greater slash volume from tops and branches. Initial *BAU* harvest on the pure pine site Po is already largely made up of stem snags (85%) with lesser slash volume. Also, pine-leading sites have a lower total harvest volume due to fallen stem snags (8 years post beetle attack).

As the C_{parity} of *1st harvest for pellets* against *BAU* shows, harvesting trees for wood pellets rather than for timber is not beneficial from a C perspective, resulting in net C loss for a single harvest of around 90 years (Table S9). To further evaluate the C benefits of one harvest scenario over another, we calculate the average C stock over the 60-year rotation (Table 2). For $C_{break\text{-}even}$ this represents the average net C balance of a site and scenario. For C_{parity} it shows the average net C difference of a scenario vs. the reference case. A net positive C average ($C_{break\text{-}even}$) over a 60-year rotation can thus be achieved for *Slash Use* for pellet (instead of burning slash in the forest) on all sites, for the *BAU* scenario on pine-dominated sites (stand-level), and for the *1st harvest for pellets* scenario only in almost pure pine stands with high mortality from MPB. Comparing net C differences (C_{parity} average) indicates that *BAU* and *1st harvest for pellets* scenarios store more C over a 60-year rotation on pine-dominated sites than a respective *Protection* case at both stand- and landscape levels, with the *BAU* scenario being the preferable one from a C viewpoint.

Landscape level scenarios

In drastic comparison to the stand-level *BAU* scenario results, the landscape $C_{break\text{-}even}$ times are much shorter

Table 2 Average carbon mass (forest, products, and displacement) across the 60-year rotation for $C_{break\text{-}even}$ and C_{parity} comparisons in tonnes C ha^{-1}

$\frac{1}{60} \times \sum_{t=0}^{59} C_{\mathrm{break-even(t)}}$	Po: Pine-only	Pd: Pine-dominated	Sd: Spruce-dominated	SF: Spruce-Fir
Stand-level				
BAU	33	18	−4	−10
1st harvest for pellets	12	−8	−33	−36
Stand-/LS level				
Slash Use	49	37	17	12
Protection	−31	−18	−2	14
Landscape (LS) level				
BAU	−19	−9	−2	4
1st harvest for pellets	−20	−17	−18	−13
$\frac{1}{60} \times \sum_{t=0}^{59} C_{\mathrm{parity(t)}}$	Po	Pd	Sd	SF
Stand-level				
BAU vs. Protection	64	36	−3	−24
1st harvest for pellets vs. Protection	43	10	−31	−50
Slash Use vs. BAU	17	19	21	21
1st harvest for pellets vs. BAU	−21	−26	−29	−26
Landscape level				
BAU vs. Protection	12	9	0	−10
1st harvest for pellets vs. Protection	11	1	−16	−27
1st harvest for pellets vs. BAU	−1	−8	−16	−17

The average carbon stock over the rotation period represents the average net carbon balance of a site and scenario for $C_{break\text{-}even}$. For C_{parity} it shows the average net carbon difference of a scenario compared to the reference case. Landscape values are also expressed per hectare for comparability. *Protection* values do not change per hectare between stand- and landscape level as the stand-level is a 60th portion of the landscape level. *Slash Use* values are equal between stand- and landscape level.

for spruce-dominated sites (Fig. 5a, Table S9). The only net C benefit over the 60-year rotation period is reached for the non-MPB-infested site SF (Table 2). Formerly positive at the stand-scale, Po and Pd average $C_{break\text{-}even}$ values turn negative over a 60-year landscape rotation (Table 2). The reason is that pine-dominated sites carry a much larger deadwood volume, which provides less and less merchantable timber over time. The shrinking harvest volumes reduce wood product displacement and storage effects. Also, on pine-dominated sites, MPB affected areas regrow slowly, i.e., return to their initial natural growth curve (at age 0). On spruce-leading sites, pine mortality leads to increased spruce and fir growth (presumed enhanced light and nutrient availability). Thus, in addition to reduced current harvest, future volumes remain lower on pine than on spruce-dominated sites, impacting the C balance of the respective forest and future timber supply; a clear argument for salvage logging pine-dominated sites.

As on stand-level, $C_{break\text{-}even}$ times are longer for *1st harvest for pellets* than for *BAU* (Fig. 5a); again due to the higher displacement and storage benefits of long-lived timber products. The difference between the two scenarios is smallest for the pure pine site, Po, whose substitution and storage effects in *BAU*, because of a relatively low sawlog harvest, are of a similar magnitude to the coal displacement effects generated by a higher salvage harvest volume used as pellet feedstock. *Slash Use* scenarios were not calculated on the landscape level, as the stand-level findings of a C benefit compared to *BAU* (slash burning) do not change.

C_{parity} of either lumber (*BAU*) or pellet feedstock extraction (*1st harvest for pellets*) against a *Protection* reference shows positive average C sequestration rates over the rotation period for sites Pd and Po (Table 2). This suggests that salvage harvest scenarios are a preferential option from a climate mitigation perspective for beetle-impacted, pine-dominated sites. This finding is clearly influenced by the long $C_{break\text{-}even}$ times of the pine-dominated sites under a *Protection* scenario (Fig. 5a). C_{parity} between the *BAU* and *1st harvest for pellets* scenario shows that *BAU* is more C efficient due to the higher displacement and storage effects of long-lived timber products over the first rotation (Fig. 5b, Table 2).

Sensitivities

The influencing factors for both stand- and landscape level break-even and parity times are the underlying tree growth curves, the amount of pine present (i.e., the amount of deadwood), post beetle-site dynamics, the chosen reference scenario, C storage assumptions, and C displacement factors. We designed our initial analysis to show the influence of varying growth curves, deadwood volumes, and counterfactuals. Therefore, we did additional analyses to test the sensitivity of the results to different C displacement factors. Changing the displacement factors for timber (Fig. 6) or pellets (Fig. 7) reveals a direct link with $C_{break\text{-}even}$ times, depending on the volume of wood used for either product (timber or pellet). A higher timber displacement factor mainly increases the C benefits of the *BAU* (Fig. 6a and c) and *Slash Use* scenarios (Fig. 6e) on spruce-dominated sites. A higher pellet displacement factor mainly reduces $C_{break\text{-}even}$ times for the dedicated bioenergy harvest *1st harvest for pellets* scenario (Fig. 7b and d) for all sites, and *Slash Use* on pine-leading sites due to the higher deadwood and thus slash mass (Fig. 7e).

Discussion

Comparison to other studies

Most of our results are in-line with previous findings of temporal C analyses (e.g., McKechnie *et al.*, 2011; Mitchell *et al.*, 2012; Zanchi *et al.*, 2012), which show that timber harvest (*BAU*) creates net C emissions to the atmosphere needing up to several decades to be compensated by tree (re-)growth on the same land (see Lamers & Junginger, 2013; for a detailed comparison of recent temporal forest C studies). However, even when live trees are harvested for bioenergy, we did not observe stand-level C break-even periods of several 100 years, as reported by Zanchi *et al.* (2010) and Greenpeace (2011). We did find that the C break-even time was nearly immediate for sites severely impacted by MPB (85% mortality) when the pellets derived from slash were used to displace coal, instead of being burnt in the forest. We believe this is the first study showing that even with forest and fossil fuel emissions taken into account that a year-zero C parity time for bioenergy and a year-zero break-even point can be obtained in a typical forest management context (i.e., not afforestation).

Our analytical framework is similar to that of McKechnie *et al.* (2011), although they used a different C model (FORCARB-ON) and applied it to a realistic landscape of millions of hectares of forest in Ontario, Canada. In that region, insect disturbance is not a significant factor and forest ecozones are predominantly boreal, i.e., spruce-dominated. The C parity times in McKechnie *et al.* (2011) for residue pellets replacing coal (16 years) are higher compared to ours (0 years) because slash decays in their reference case and is not burnt in the forest. Their C parity times for roundwood pellets against a *Protection* reference (38 years) are similar to ours (i.e., *1st harvest for pellets*) for spruce-dominated sites on stand-level (46 years), but differ significantly on landscape level (73 years). The latter

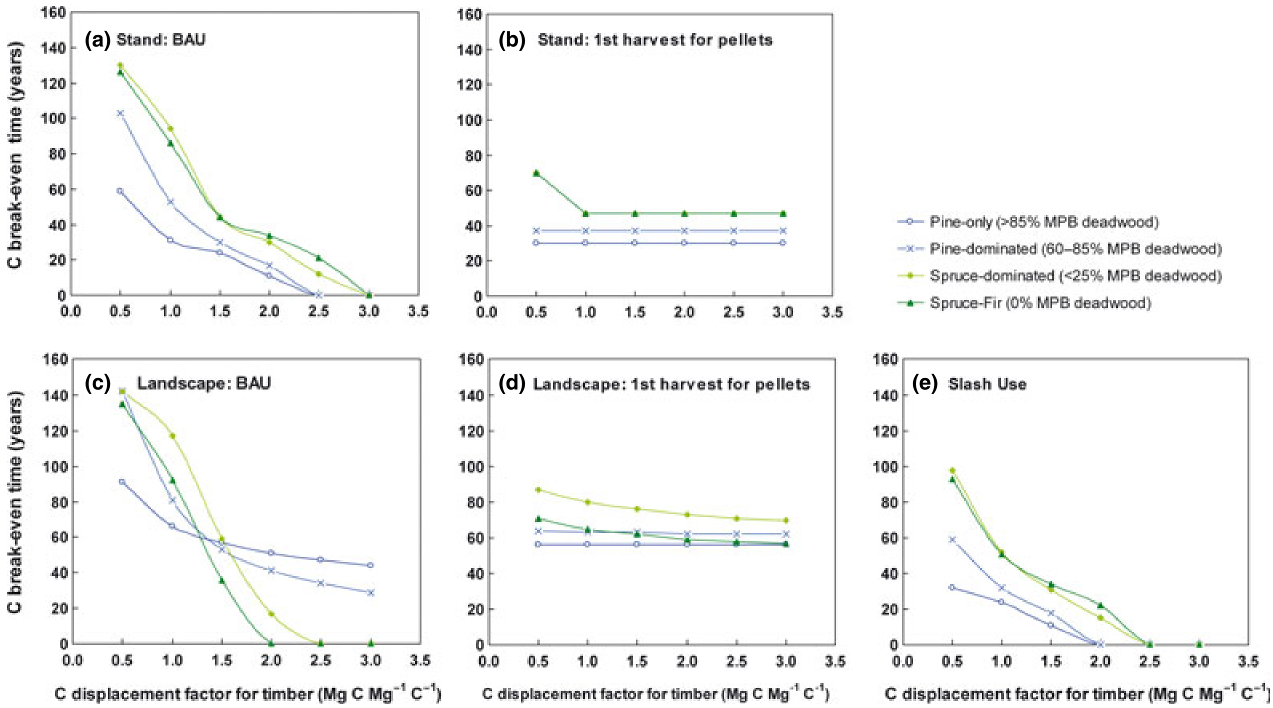

Fig. 6 Sensitivity analysis shows that varying the C displacement factor for timber products (from the initial 1.7) has stronger effects on $C_{\text{break-even}}$ times, i.e., time spans needed to reach preharvest C levels, of scenarios with a higher harvest share for timber (*BAU*: a, c). Also, it affects spruce-dominated more than pine-leading sites, especially on landscape level (c), due to the share of decaying deadwood. The *1st harvest for pellets* scenarios are also influenced by a 0.5 timber displacement factor on spruce leading sites since the break-even times initially reach into the second harvest cycle (post 60 years). The pellet displacement factor was kept constant at 0.923. Changing either does not affect the *Protection* reference as no wood is harvested. In (b), Spruce-dominated values equal those for Spruce-Fir.

may be linked to the novelty of modeling uneven aged stands in CBM-CFS3 (see below) and the higher initial deadwood C pool in our scenarios. Despite these discrepancies, our modeling appears in-line with other findings, but emphasizes the importance of taking regionally specific circumstances into account when conducting studies such as these.

Our findings of varying parity times depending on the reference case selection stress the significance of defining the correct counterfactual. Mitchell *et al.* (2012) compared all harvest cases to a protection scenario (taking wildfire into account). This however neglects leakage effects, i.e., increased timber harvesting in forests will occur should any particular forest be protected. Our study region is and remains a timber supply area, albeit reductions in harvest volumes are expected to occur until at least 2070 due to declining timber flows caused by the MPB outbreak and increased harvesting in response. This suggests that the *BAU* scenario is the most appropriate counterfactual in our case. What's more, we found that salvage logging and reforesting sites most severely affected by MPB actually provides higher C benefits than the protection of such sites.

We modeled theoretical sites which are representative of actual stands across the forest area earmarked for (future) timber harvest in the study region. Note that in reality only 44% of BC forests are available to harvest. The remaining 56% cover 7.6 Mha of permanently protected areas and 23 Mha which are not economical or not desirable to harvest (BC-MoF, 2010d). These forests contribute to the sustainability of forestry in the province by maintaining C storage and sequestration, providing habitat for wildlife and fish, maintaining water quality, providing cultural, recreational, and other values.

Robustness of our approach

This study used regionally specific tree growth curves and C pool dynamics to represent the study area as accurately as possible. In addition, to correctly account for C benefits and losses of harvesting activities (Schlamadinger *et al.*, 1997; Markewitz, 2006; Hennigar *et al.*, 2008), we added end-use C cycles and fossil fuel emissions to the forest C analysis. All factors (half-life, conversion, substitution) and emissions reflect current local

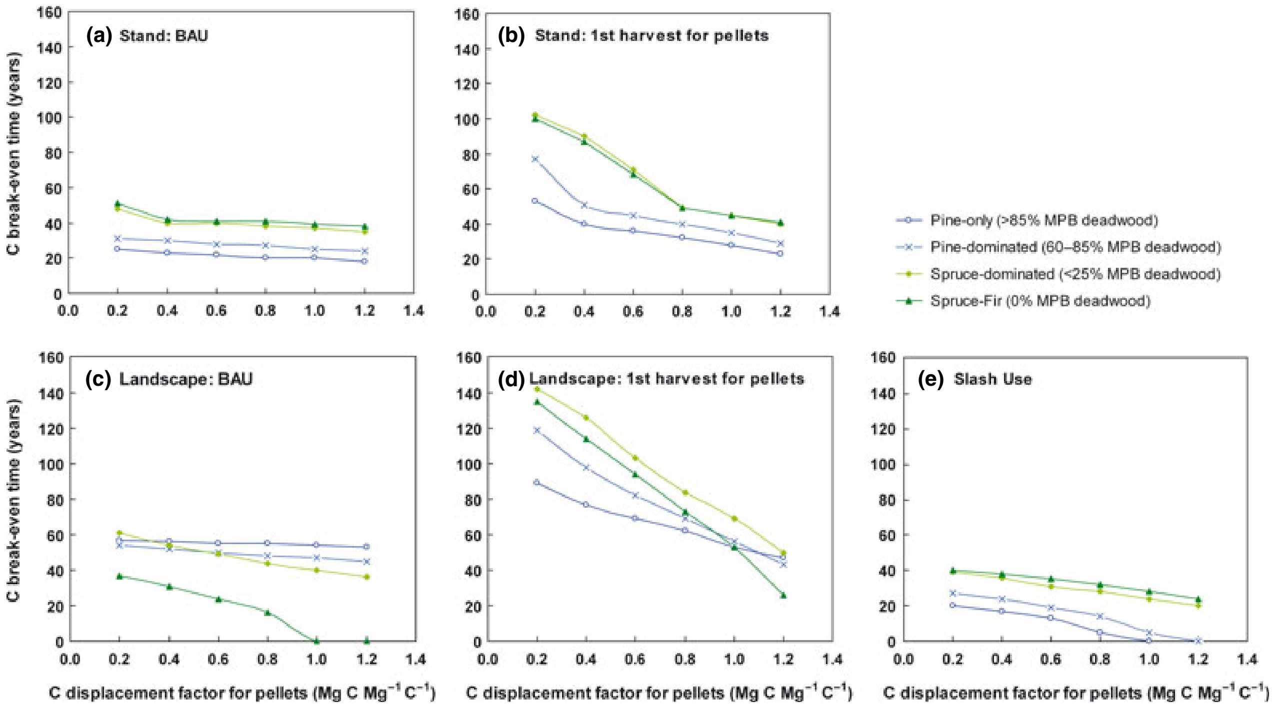

Fig. 7 Sensitivity analysis indicates that varying the pellet displacement factor (from the initial 0.923) has larger impacts on the $C_{\text{break-even}}$ time of scenarios with a higher share of harvest volume for pellet feedstock (b, d). The timber replacement factor is kept constant at 1.7. Changing either factor does not affect the *Protection* reference as no wood is harvested.

conditions in the respective value chains. While these may change over the long time horizons involved in forestry, future values are hard to predict with accuracy. This study modeled the current situation where pellets directly displace coal as they are co-combusted in coal-fired power plants in Europe. An alternative approach would have been to model the European electricity mix, and to discount energy displacement factors over time, given that the fuel mix in Europe is expected to become less C intensive in the future.

The assumption of constant displacement factors has been criticized by economists (e.g., Rajagopal *et al.*, 2011; Thompson *et al.*, 2011 for liquid biofuels). Our factors for pellets vs. coal and for wood products vs. other building materials only apply to the simulation period. Discounting displacement factors would need to take into account complex leakage and rebound effects, which go beyond the scope of this article. Also, it can be considered highly uncertain whether or to what extent wood pellets from BC (which replace coal in Europe) affect global coal markets.

In our view, an elaborate projection of future displacement factors is less pressing than a more sophisticated modeling of biomass C stored in landfills. Our sensitivity analysis revealed that even with a very low pellet displacement factor, break-even times still remain under 30 years for *Slash Use* on pine-dominated sites (Fig. 7e). In addition, previous studies suggest that C stored in wood products is dwarfed by C remaining in landfills over time (Price *et al.*, 1996; Earles *et al.*, 2012). Although we exclude C stored in landfills in this analysis, we do account for biogenic C emissions of long-lived wood products. The short half-life of 30 years used in this study is an underestimate of storage and therefore our emissions from wood products is likely too high. Temporal C storage in pulp and paper was excluded as lifecycle emissions and respective assumptions are disputable (Ingerson, 2011), regionally specific (Earles *et al.*, 2012), and a proper investigation goes beyond the scope of this paper.

This analysis is the first to simulate post-MPB stand dynamics via splitting sites onto different yield curves, thus creating uneven aged stands/landscapes. While our results at the stand-level seem robust compared to alternative approaches (e.g., MANOMET, 2010), results on landscape level may be subject to larger error, though the size of such errors are hard to quantify. Interpretations of our landscape results should take this uncertainty into account. An alternative approach would have been to generate yield curves under varying MPB infestation rates and age structures, resulting in a very large number of modeling scenarios (roughly

750 variations and 900 different transition rules per site). Such an analysis would go well beyond the scope of this article. In addition, research on post-beetle stand dynamics has not yet fully quantified actual changes in yield curves for various conditions (undergrowth structure, stand age, etc.), leaving further sources of error.

Given current conditions in the study area where extensive areas of pine forest have been killed by MPB attack, and where future timber supply volumes are expected to decline due to the infestation, it appears preferable to harvest pine-dominated over spruce-dominated stands. This conclusion is in-line with current BC government efforts on harvesting preferences (BC-MoF, 2007). A generalization of this finding however should be avoided, as many MPB-infested pine stands in the interior of BC can show significant re-growth from understory trees and trees in mixedwood stands (FPB, 2007a; Coates *et al.*, 2009; Vyse *et al.*, 2009; Hawkins *et al.*, 2012). Salvage logging could remove this secondary structure, further reducing future harvest volumes. From a C perspective, official BC forest guidance to focus salvage logging on stands with pine shares over 70% (BC-MoF, 2007), i.e., ranges between our Sites Pd and Po, is supported by our study. Additional research though is needed to investigate whether large-scale salvage logging may cause unintended adverse environmental impacts, e.g., on biodiversity or hydrology.

Given local timber supply estimations, harvesting in the study area will continue to predominantly utilize pine-dominated stands until 2020. Post 2020, however, spruce-dominated stands are expected to make up the majority of harvests. This implies that pellet production based on salvage-focused roundwood, i.e., pine-dominated sites otherwise not harvested, remains a short-term feedstock supply and C mitigation option. The use of (primary and secondary) residues is a long-term pellet feedstock and C mitigation option, when pellets replace coal. The use of lumber for wood products however entails higher C benefits than its use as pellet feedstock, and should therefore be the main focus in times when live trees will again make up the majority of harvest volumes.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. C_{parity} for *BAU* and *1st harvest for pellets* against their *Protection* reference scenario.
Figure S2. C_{parity} for *Slash Use* and *1st harvest for pellets* against their *BAU* reference scenario.
Figure S3. C_{parity} for *BAU* and *1st harvest for pellets* against their *BAU* reference scenario.
Figure S4. Temporal carbon details for the pine-dominated site Pd (73% pine), stand-level.
Figure S5. Temporal carbon details for the spruce-dominated site Sd (26% pine), stand-level.
Figure S6. Temporal carbon details for the spruce-fir site SF (0% pine), stand-level.
Figure S7. Temporal carbon details for the pine-only site Po (100% pine), stand-level.
Figure S8. Temporal carbon details for the pine-dominated site Pd (73% pine), landscape-level.
Figure S9. Temporal carbon details for the spruce-dominated site Sd (26% pine), landscape-level.
Figure S10. Temporal carbon details for the spruce-fir site SF (0% pine), landscape-level.
Figure S11. Temporal carbon details for the pine-only site Po (100% pine), landscape-level.
Figure S12. Calculated potential annual wood pellet production in Quesnel and Prince George TSA in relation to potential exports from Western Canada (based on BC-MoF, 2010b,c, 2011b,c; Cocchi *et al*., 2011; Friesen & Goodison, 2011; BC-MoF, 2012a; CAN-BIO, 2012; Prasad, 2012).
Table S1. Harvest matrix for *BAU*.
Table S2. Harvest matrix for *Slash Use*.
Table S3. Harvest matrix for *1st harvest for pellets*.
Table S4. Distribution of sawlog volume at BC sawmills in the study region.
Table S5. Displacement factor for pellets (Damen & Faaij, 2006; Jonker *et al*., 2013).
Table S6. Displacement factors for long-lived wood products in studies with North American geographic reference as calculated and cited in Sathre & O'Connor (2008).
Table S7. Fossil fuel emissions per value chain in kg CO_{2e} / tonne output.
Table S8. Literature values of fossil fuel emissions for BC sawdust wood pellets for comparison**.**
Table S9. Exact $C_{break\text{-}even}$ and C_{parity} points in years *t* on stand- and landscape level.
Table S10. Legend explanation for the detailed temporal carbon balances.