Climatic Change
https://doi.org/10.1007/s10584-020-02711-8

Forest harvesting and the carbon debt in boreal east-central Canada

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Received: 8 February 2019 / Accepted: 20 March 2020/Published online: 11 April 2020

Abstract

Conversion of carbon-rich, primary boreal landscapes to managed ones through clearcut-based silviculture has the potential to decrease landscape-level carbon storage and thereby incur a significant carbon debt. We calculated carbon debts and payback periods associated with production of wood pellets to replace coal, oil and natural gas in electricity generation for such landscape conversion in boreal east-central Canada. Local forest inventory information in combination with the Carbon Budget Model (CBM-CFS3) was used to estimate biomass and dead wood carbon stocks after fire or clearcutting, and resulting age- and disturbance-specific carbon stock estimates were used to populate simulated landscapes. Based on empirical information, we investigated a range of fire-return intervals in the primary landscapes (114–262 years), harvest rotation ages (80–100 years) and conversion efficiency factors (0.17–0.71 tonnes fossil fuel carbon eliminated per tonne harvested wood carbon). After a first rotation of harvesting, carbon stocks declined 33–50% relative to stocks in the natural, fire-dominated landscapes and payback periods ranged from 92 to 757 years. The type of fossil fuel had the strongest effect on payback periods: under average efficiencies, ranges were 122–207, 156–268 and 278–481 years for coal, oil and natural gas respectively. These calculations suggest that under a wide range of assumptions, clearcut-based management of boreal primary landscapes to produce wood pellets to replace fossil fuels in electricity generation will result in net emissions of greenhouse gases to the atmosphere for many decades.

Keywords Boreal forest · Forest harvesting · Carbon debt · Wood pellets · Electricity generation

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10584-020-02711-8) contains supplementary material, which is available to authorized users.

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1 Introduction

Forests are valuable from a global climate change perspective not only because they remove carbon from the atmosphere as they grow but also because they store large amounts of carbon (Harmon et al. 1990; Mitchell et al. 2012). These two characteristics are fundamental to understanding the landscape-level concepts of "carbon neutrality" and the "carbon debt". Disturbed stands can be expected to be net emitters of atmosphere greenhouse gases either because of combustion or because of decay of organic material or both. However, these emissions are simultaneously being countered by older stands in the landscape that are growing and sequestering carbon. Under a constant disturbance regime, this give and take among stands can lead to a steady state with respect to emissions, that is carbon neutrality (Weisser 2007). Carbon neutrality has important implications for forest management: any net savings of emissions resulting from disturbances, for example through the use of harvested wood as biofuel to replace fossil fuels (Richter et al. 2009), can play a positive role in mitigating global climate change.

There is an important caveat; however, the net biosphere flux in the presence of bioenergy production needs to be compared with the net biosphere flux in the absence of such production (IPCC 2014, p. 877; Searchinger et al. 2009). If management over time results in a decrease in amount of carbon stored in the landscape, then any emissions savings must be judged against this carbon debt (Fargione et al. 2008; Holtsmark 2012). Management in such cases will result in a net atmospheric benefit only once the debt has been paid off. Fargione et al. (2008) considered several land management strategies to replace fossil fuels with renewable biofuel and found that the pay-off period was sometimes long; for example, conversion of natural prairie to a corn-ethanol system resulted in a debt pay-off period of 93 years and conversion of Brazilian tropical rainforest to a soybean-biodiesel system resulted in a debt pay-off period of 319 years. Thus, rather than contributing to the important goal of reducing greenhouse emissions in the near future (IPCC 2014; Sanderson et al. 2016), they found that these management practices were instead resulting in increased concentrations of greenhouse gases in the atmosphere for long periods of time.

The possibility of a carbon debt is of particular concern in boreal forests of Canada, where forest management often relies upon clearcutting of carbon-rich, primary forests. Canada is home to the largest areas of intact, primary boreal forest in the world; defined for example as forested areas of at least 500 km^2 that exhibit no remotely-detected signs of human activity or associated habitat fragmentation (Potapov et al. 2017; see also Kormos et al. 2018). Its boreal forest is estimated to hold 50.6 billion tonnes of carbon or 6% of the global stock (Pan et al. 2011). Most of the ongoing loss of these intact forests is due to forest management. For example, 60% of the 129,487 km^2 lost between 2000 and 2013 was within forestry tenures (Smith and Cheng 2016). In Ontario, which represents one of the largest concentrations of intact forests in Canada (Watson et al. 2018), 86% of the loss was within forestry tenures (Smith and Cheng 2016). Although poorly known, decreases in carbon storage as a result of forest management are potentially large. Kurz et al. (1998) estimated that conversion from natural to managed landscapes via clearcutting resulted in a biomass reduction of 32% for a boreal forest and 76% for a British Columbia coastal forest. At the global level, Erb et al. (2018) estimated that forest management resulted in 24–36% decreases in forest biomass carbon stocks. Clearly, an understanding of the effects of forest management on greenhouse gas balance requires consideration of such decreases in carbon storage (e.g. Zhang et al. 2010; Chen et al. 2014), and the possibility that management will entail a considerable period of carbon debt.

The pay-off period for any such debt of course is dependent not only on the nature of forest management practices but also on the specific use of the harvested wood products. Of particular interest is the use of wood in electricity generation to supplant fossil fuels, including coal, oil and natural gas. Electricity and heat generation are the largest contributors to global CO_2 emissions, comprising 41% of total CO_2 emissions in 2017 (IEA 2019). Although biofuels as yet represent a relatively minor part of the Canadian wood product mix, their production is increasing. For example, wood pellet exports increased from 2.8 to 3.9 million tonnes between 2012 and 2017, and Canada is the world's second largest exporter of wood pellets (FAOSTATS 2018). The use of biofuel to supplant fossil fuels in power generation is often discussed as a strategy to mitigate global warming (see Norton et al. 2019) and in light of the increasing importance of the biofuel sector in Canada, it is important to consider the climate change mitigation potential of bioenergy harvests from Canadian boreal forests.

Unfortunately, few efforts as yet have been devoted to understanding the carbon debt in temperate or boreal contexts. Holtsmark (2012) calculated the carbon debt for harvesting of wood pellets in boreal Norway, but instead of considering the conversion of primary to secondary (managed) forests, he examined the debt incurred during intensification of harvests in secondary landscapes. Mitchell et al. (2012) examined conversion of old-growth U.S. Pacific northwest forests to managed forests and found that clearcutting for bioenergy production resulted in carbon debt payback times of > 100 years, irrespective of the forest type. Several studies calculated the length of time for harvested forests to regain their carbon stocks following harvesting, but did not explicitly consider the disturbance regimes of natural forests, which Kurz et al. (1998) showed to importantly influence carbon stores. For example, Harmon et al. (1990) considered current-day wood product production and calculated that temperate rainforest stands took 250 years to regain carbon stocks following harvesting. They estimated a payback period of > 350 years in converting 450-year-old forests to 60-year-old plantations. For boreal upland coniferous forests, Bernier and Paré (2013) used CO_2 fluxes measured from flux towers to calculate payback periods of 90 years or more in converting 120-year-old forests to managed forests (rotations of 60–120 years) in which wood pellets were used to replace heating oil. Laganière et al. (2017) used a stand-level approach (i.e. time to carbon parity in stand regrowth) to assess the relative merits of various feedstocks in replacing coal, oil or natural gas, and found that payback periods for stemwood harvested from Canada's managed forests generally exceeded 100 years. As noted by Holtsmark (2012), however, calculation of the payback period for stand regrowth ignores long-term landscape dynamics; for example, he found that payback times more than doubled when considering a series of harvest events compared with one. Kurz et al. (1998) incorporated landscape-scale disturbance regimes in estimating carbon storage in natural and managed landscapes for several Canadian forest types, but did not undertake carbon debt calculations. Ter-Mikaelian et al. (2015) calculated a carbon debt period of 112 years for stemwood used to replace coal in electricity generation at a specific location in boreal northwestern Ontario. Given changes in fossil fuel use over time (for example, coal has now been phased out of use in Ontario (Environment Canada 2017)), calculations for a broader range of fossil fuels are of interest, as is consideration of the influence of key disturbance regime characteristics (such as fire-return intervals and rotation lengths) on the debt period.

In this paper, we calculate carbon debt payback periods for the conversion of primary, fire-dominated boreal landscapes of northeastern Ontario to landscapes managed by clearcut harvesting for wood fibre; specifically, the use of wood pellets to supplant fossil fuels (coal, oil and natural gas) in electricity generation. We consider a range of likely historic fire regimes, which can vary considerably within the boreal forest (e.g. Cyr et al. 2009), and consider a range of likely rotation periods. We also considered best- and worst-case emission replacement scenarios by using the range of greenhouse gas emissions reported from life cycle analyses.

2 Methods

2.1 Study site

Our focus was on the Boreal Shield East Ecoregion of Canada (Kurz et al. 1992), with forest inventory information coming specifically from the Lake Abitibi Ecoregion of Ontario (Crins et al. 2009; Fig. 1). Main tree species in the region include black spruce (*Picea mariana*), white spruce (*P. glauca*), balsam fir (*Abies balsamea*), jack pine (*Pinus banksiana*), tamarack (*Larix laricina*), white birch (*Betula papyrifera*), trembling aspen (*Populus tremuloides*) and balsam poplar (*P. balsamifera*) (Rowe 1972). Winters are long and cold, and summers are short and warm. In the ecoregion, mean annual temperature ranges from − 0.5 to 2.5 °C and mean annual precipitation ranges from 652 to 1029 mm (Mackey et al. 1996a, b).

2.2 Estimation of carbon stocks

Similar to Sharma et al. (2013), our overall approach was to use the Carbon Budget Model (CBM-CFS3, v. 1.2) to estimate carbon stocks after fire or clearcutting for typical forest landscapes of the region, and then, use the resulting age- and disturbance-specific carbon stock

Fig. 1 Map of the province of Ontario, Canada, and the Great Lakes (light grey), showing the location of three forest management units (dark grey) used to estimate carbon stocks. Also shown are the three Ecodistricts of Ontario and, within the Ontario Shield Ecodistrict, the location of Ecoregion 3E (Lake Abitibi Ecoregion; diagonal lines)

estimates as "generalized stands" that could then be used to populate simulated landscapes. CBM-CFS3 is an operational-scale, aspatial model that has been used extensively to estimate carbon stocks in Canada and elsewhere (Kurz et al. 2009). Because of uncertainty concerning the effects of forest harvesting on soil carbon pools, following Holtsmark (2012) and Laganière et al. (2017), we considered only carbon stocks in biomass and dead wood pools. Possible consequences of this simplification are discussed in Section 4.

2.2.1 Gross merchantable volume

Gross merchantable volume (GMV) is a key stand characteristic used in CBM-CFS3 to estimate carbon stocks. In addition to climate and forest age, GMV is importantly influenced by three stand attributes: tree species composition, productivity of the site (i.e. site class) and tree density relative to what is deemed typical (i.e. stocking). To calculate GMV, we used stand-level estimates of these attributes from digital Forest Resource Inventories (FRI; OMNR 2001) for forest management units in the study region (Fig. 1).

Specifically, following Penner et al. (2008), tree species composition was characterized based on combinations of "forest unit" (major forest type) and dominant (leading) species. We focused on the major groups in the study site by picking the combinations that comprised at least 1% of the total forest area, excluding bogs (which are not a focus of forest management in the region). The result was 15 combinations of forest unit and leading species that together comprised > 96% of the forested area (Online Resource, Table S1). For each of these, we determined site index, average stocking and total area for each of the five site classes in the FRIs (Online Resource, Tables S2 and S3) and then used Penner et al. (2008) to calculate GMV as a function of forest age (see Online Resource, Gross merchantable volume calculations). In these growth curves, GMV was assumed to remain approximately constant after 100–200 years of age, which was similar to other studies in which GMV was assumed to remain constant or to slightly increase with time (e.g. Holtsmark 2012; Bernier and Paré 2013; Laganière et al. 2017). By contrast, several other boreal studies assumed that stand volumes (and carbon stocks) started to decline when stands went through a "breakup" period (e.g. Kurz et al. 1998; Ter-Mikaelian et al. 2015). We were able to test this assumption for one of our management units from a time-since-fire map created by Gauthier et al. (2002). For forests > 100 years of age, we found that average GMV increased slightly with time since fire, although not significantly so; hence, our assumption of constant GMV seemed a reasonable one (see Online Resource, GMV in old-growth forests).

2.2.2 Use of CBM-CFS3 to estimate carbon stocks

In addition to GMV curves for the various forest types, CBM-CFS3 requires (1) the ecoregion of the forest; (2) an inventory of areas and ages of forest types; (3) the timing of disturbances; and (4) rules for determining forest succession after disturbances. In our runs, we used the default climate and decay parameters for the "Ontario Boreal Shield East". As detailed above, GMV curves and stand areas in the inventory were based on information for the various combinations of forest unit, leading species and site class in the FRIs. To facilitate comparison with Holtsmark (2012), we estimated carbon stocks in an area of 75,000 km^2 (52% of the area of Ontario's Ecoregion 3E shown in Fig. 1). We used the model to calculate age-dependent carbon stocks following the two dominant types of disturbances characteristic of natural and managed landscapes in the region, respectively: "wildfire" and "clearcutting without salvage" (i.e. harvesting of 85% of merchantable trees from a stand without harvesting of snags). Following a disturbance, we assumed that species composition, site class and stocking remained unchanged and that there was no regeneration delay. The age- and disturbance-dependent carbon stocks obtained from the model were then used to populate our simulated landscapes (see below).

A complication is evident, however. At least in the short term, the age of the forest at the time of a disturbance will influence subsequent stocks of dead organic matter. For example, snag populations and dynamics following a fire can be expected to differ considerably between a forest burned at a young age compared with one burned at an older age (Online Resource, Fig. S2). Accordingly, we used CBM-CFS3 to model not only carbon stocks following the two disturbance types but also for all possible forest ages at the time of the disturbance (in 5-year increments). For example, in one run, we set inventory ages to zero at the start of a simulation, burned the forest at 5 years of age and then tabulated carbon stocks at various ages following the burn. In another run, we instead burned the forest at 10 years of age, etc. In theory, such legacy effects could extent to even further back in time in natural landscapes (for example, the age of the forest at the time of the previous two disturbances), but such effects are bound to be small and were ignored. Because we were interested in the first rotation of clearcutting (see below), the disturbance prior to clearcutting was wildfire. Examples of age- and disturbance-dependent carbon stocks in the generalized stand are shown in Online Resource, Fig. S2.

2.3 Simulated landscapes

Our primary interest was the carbon debt incurred during the transition from primary forest landscapes dominated by fire to ones managed under clearcut silviculture. We used the geometric distribution to model the age distribution in primary (fire dominated) landscapes (Van Wagner 1978). These landscapes were then transitioned via harvesting to a uniform age distribution characteristic of regulated, managed forests (see Van Wagner 1978). Mitchell et al. (2012) suggested that a more appropriate measure than the carbon debt payback period was the time to “carbon parity”; namely, the time period required to attain the carbon stores that would have been attained in the landscape in the absence of management (rather than carbon stores at the start of management). Note that the two are equivalent in our case: we considered natural landscapes that were in steady state with the respect to disturbance regimes; hence, their carbon stores were constant over time.

To estimate the fire-return interval for the original, fire-dominated (primary) landscapes, we used empirical information in the literature. Cyr et al. (2009) estimated fire intervals during a 7600-year period from charcoal in the bottom of lake sediments near the Ontario-Quebec border and Bergeron et al. (2001) estimated times since fire in four landscapes of boreal eastern Ontario and western Quebec from fire scars, ages of oldest trees and historical fire records. For their data, we establish the average fire-return interval based on the best fit of the geometric distribution to the observed age distributions (Online Resource, Figs. S5, S6). The estimated fire interval was the same for the two studies (166 years); hence, we used it as our central estimate.

To investigate natural bounds of variability in fire-return intervals, we used the “conservative range of variability” estimated by Cyr et al. (2009); namely, the range of fire intervals that they observed during periods of relatively constant fire regimes. Lower and upper limits in this case corresponded to fire-return intervals of 114 and 262 years (Online Resource, Fig. S5). As noted earlier, our implementation of CBM-CFS3 included forest age at the time of the disturbance. These were not known in either Cyr et al. (2009) or Bergeron et al. (2001); hence, in our natural landscapes, we assumed that ages-at-burn followed expectations from the geometric model.

To transform fire-dominated landscapes to harvested landscapes, it was assumed that a fixed proportion of the landscape area was harvested each year, with the oldest forests harvested first. The rotation age of the regulated (or normalized) managed forest is the reciprocal of this proportion. For example, harvesting 1% of the area per year means that by year 100, all of the primary forest has been harvested once, and results in a uniform stand age distribution up to a maximum age of 100 years. Subsequent harvesting with a rotation age of 100 years maintains this age distribution. Rotation ages in the region are usually given at 80–100 years (e.g. Cyr et al. 2009); hence, we investigated carbon debts within this range. We used the default assumption in CBM-CF3 that 85% of the merchantable volume goes into wood products (wood pellets in this case).

2.4 Calculating the payback period of the carbon debt

The time period to pay back any carbon debt depends importantly on the specific use of the harvested fibre (Holtsmark 2012). We were interested in the use of wood pellets to eliminate fossil fuels (coal, oil or natural gas) during electricity generation (other wood product mixes are considered in Section 4). A key parameter here is the "conversion efficiency factor", often also termed the "displacement factor" (Schlamadinger and Marland 1996, Brandão et al. 2019, Kalt et al. 2019). It is defined as the number of tonnes of carbon in fossil fuel eliminated per tonne of carbon in the harvested wood. The higher this conversion efficiency factor, the faster the debt will be paid off. Two life cycle studies were used to estimate this number (Weisser 2007; Laganière et al. 2017 (upstream only)). Their average values tended to be similar (see calculations in Online Resource, Table S4); hence, we used the means across the two as our central estimates (0.55, 0.44 and 0.26 for coal, oil and natural gas, respectively). In these calculations, we used the IPCC (2006) default emission factor for wood of 114 kg CO_2e GJ^{-1}. We also assumed that 10% of the harvest was used to dry the pellets that there were 1694 kWh m^{-3} of dried wood, and, following Laganière et al. (2017), that percent energy efficiency was 26, 33, 35 and 45% for wood, coal, oil and natural gas, respectively. To investigate a range of values, we used the life cycle greenhouse gas emissions reported by Weisser (2007) (his Fig. 5). We modelled a "worst-case" scenario by using the lowest reported life cycle emissions for the fossil fuel and the highest for wood. Conversely, in a "best-case" scenario, we used the highest for the fossil fuel and the lowest for wood. Resulting lower and upper values were 0.38 and 0.71 for coal, 0.24 and 0.66 for oil and 0.17 and 0.43 for natural gas (see calculations in Online Resource, Table S5). The resulting overall range of values (0.17 to 0.71) was similar to the range of values used by Mitchell et al. (2012) to investigate wood bioenergy as a replacement for fossil fuels (0.2 to 0.8).

3 Results

With increasing long fire-return intervals in natural landscapes, steady-state carbon stocks increased (Fig. 2). The increase was relatively modest, from 514 Mt C in the entire landscape (68.5 t ha^{-1}) under a fire-return interval of 110 years to 608 Mt C (81.1 t ha^{-1}) under a fire-return interval of 265 years. The percentage of biomass C that was below ground was

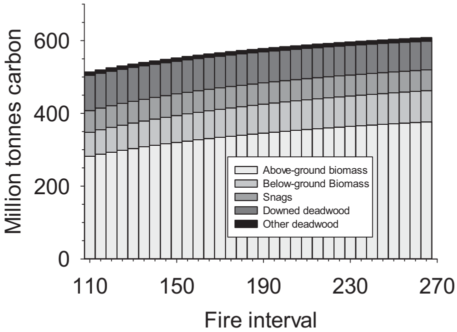

Fig. 2 Modelled stocks of carbon in various pools as a function of the fire-return interval in a 75,000 km² natural landscape of boreal northeastern Ontario. Carbon stocks following fire for forested areas typical of the Boreal Northeast Shield region of Ontario were estimated by use of CBM-CFS3, and the estimates where then used to populate natural landscapes modelled under the various fire-return intervals

relatively constant at 19%, but as the fire interval increased, the percentage of C in the deadwood pool decreased from 32% at a return interval of 110 years to 24% at a return interval of 265 years.

The steady-state store of carbon in the landscape after a first rotation of harvesting was strongly influenced by the length of the rotation period, from 143 Mt C at a rotation length of 5 years to 487 Mt C at a rotation length of 200 years (Fig. 3). Within the range 80–100 years, steady-state C stocks (excluding soils) varied from 304 to 346 Mt C, i.e. a decline of 33–50% relative to stocks in the natural, fire-dominated landscapes. Within that same range, below-ground biomass C comprised 19–20% of the total carbon, and deadwood C declined from 29% of the total at a rotation age of 80 years to 24% at a rotation age at 100 years. The proportion of the total C pool in snags increased steadily with rotation age from < 1 at 80 years to 5% at 100 years (vs. 9–12% in natural landscapes). Interestingly, the maximum attainable harvest under steady-state conditions (i.e. once the regulated age distribution was attained) was 11.8 Mm³ year^{-1} (2.49 Mt C year^{-1}) at the relatively young rotation age of 65 years (Fig. 3). This is a reflection of the relatively early attainment of maximum sustained yield observed in the GMV curves (i.e. where a line through the origin is tangential to the GMV curve; Online Resource, Fig. S1). Over the range of 80 to 100 years, steady-state harvests ranged from 11.5 to 10.6 Mm³ year^{-1}, respectively.

As a specific example of the carbon debt, we modelled the transition from a natural forest with a fire-return interval of 166 years (the central estimate for the region) to a regulated landscape with a rotation age of 100 years. During the rotation, landscape carbon stocks declined by 38% from 564 to 347 Mt C (Fig. 4). Below-ground biomass C percentage remained relatively constant over time at 19% of the total; however, the deadwood C pool declined from 28 to 24% and the snag C pool declined from 10 to 4%. Variation in wood

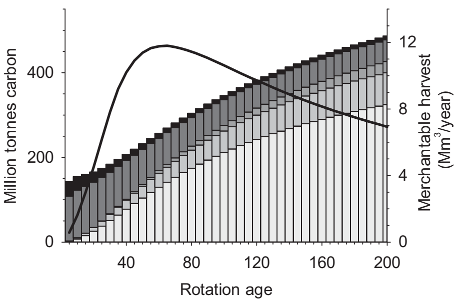

Fig. 3 As Fig. 2 except that carbon stocks after one rotation are shown in normal (or regulated) landscapes of 75,000 km² managed via clearcut silviculture under different rotation ages. The average annual merchantable harvest at equilibrium (i.e. at the end of the first rotation) is also shown (black line)

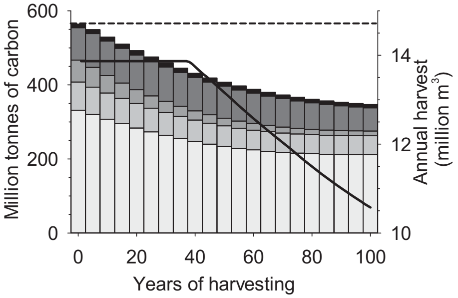

Fig. 4 Modelled carbon stocks in a fire-dominated natural forest in boreal northeastern Ontario of 75,000 km^2 during the first rotation of harvesting (length of rotation = 100 years). The annual harvest (black line) is also shown. Each year, 1% of the forest was harvested, starting with the oldest stands first. By the end of the rotation, all of the original forest had been cut once. The dashed line represents the carbon stock of the original forest (in this case with a fire-return interval of 166 years). The difference in height between the dashed line and the stacked bars represents a carbon debt. Shades of the various carbon stocks are as shown in Fig. 2

harvests over the 100 years illustrated the fall-down effect. During the first 37 years, stands of maximum volume were harvested, resulting in a constant annual harvest of 13.9 Mm^3. Thereafter, as younger stands comprised an increasing fraction of the harvest, the harvest dropped nearly linearly to 10.6 Mm^3 year^{-1}, which was the expected harvest thereafter (i.e. the equilibrium harvest in the regulated landscape, ignoring any forest area lost due to roads and landings).

In this example, the use of wood pellets in electricity generation to eliminate coal, oil or natural gas resulted in carbon debt payback periods of 155, 200 and 356 years, respectively (i.e. where solid lines intersected the dash-dot-line in Fig. 5). Best- and worst-case estimates yielded a wide range of values: 116–237 years for coal, 126–384 years for oil and 207–558 years for natural gas (dotted lines in Fig. 5).

Considering the full range of simulated fire-return intervals and rotation ages, the type of fossil fuel being replaced had the strongest effect on payback periods, with central estimates of the payback period varying within the range of 122–207 for coal, 156–268 for oil and 278–481 for natural gas (Fig. 6). Debt periods increased as rotation ages decreased relative to the fire-

Fig. 5 The carbon debt repayment period for forest harvesting in boreal northeastern Ontario showing the cumulative reduction in carbon stocks (dash-dot line) when a 75,000 km^2 natural landscape with a fire interval of 166 years is converted to a clearcut-managed landscape with a 100-year rotation age. The debt is calculated for electricity generation in which harvested wood pellets replace coal (black lines), oil (dark grey lines) or natural gas (light grey lines). For each fossil fuel, a central estimate is shown (solid line) and upper and lower estimates (dotted lines; see text for details). The repayment period is the number of years until a fossil fuel line intersects the dash-dot line; that is, until the carbon debt is paid off by the eliminated fossil fuel emissions

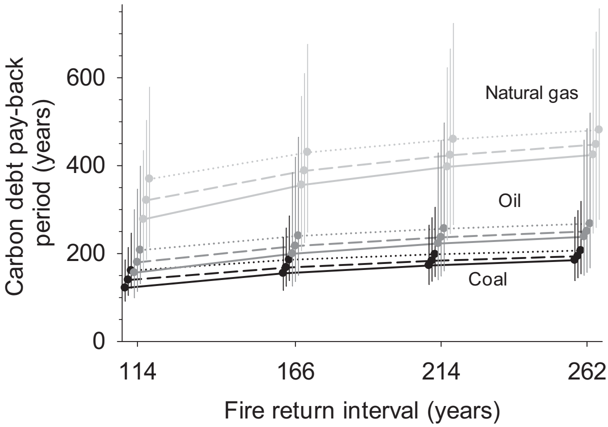

Fig. 6 The carbon debt payback period incurred during harvesting of natural post-fire landscapes of boreal northeastern Ontario as a function of the original fire-return interval and the rotation age of harvesting. For each fossil fuel type, three rotation ages are shown: 80 years (dotted line), 90 years (dashed line) and 100 years (solid line). Symbol positions on the x axis have been offset slightly for presentation purposes

return interval. Lowest debts were when the rotation age most closely approximated the fire-return interval (100-year rotation and 114-year fire interval), although they still exceeded 100 years even for coal (central estimates were 122, 156 and 278 for coal, oil and natural gas respectively). At the other end of the spectrum, they were longest when the rotation was short (80 years) and the fire interval was long (262 years); in this case, they exceeded 200 years (corresponding central estimates for the three fossil fuels were 207, 268 and 481).

4 Discussion

These results suggest that clearcut harvesting of primary boreal forests to produce wood pellets to replace fossil fuels in electricity generation is unlikely to be useful in mitigating climate change in the near term; indeed, it is likely to exacerbate CO_2 emissions for many decades, a conclusion that corroborates Norton et al. (2019). Even under the best-case scenario of a relatively short fire-return interval (114 years), a long management rotation (100 years) and an upper-limit conversion efficiency of 0.71 in eliminating coal, the carbon debt payback period was > 90 years. At the other extreme, a relatively long fire interval (262 years), short rotation (80 years) and lower limit efficiency factor (0.17 for natural gas) yielded a payback period of > 500 years. In addition, our estimates of the payback period are probably conservative in several respects. We excluded forest loss and regeneration delays due to roads and landings, which will reduce harvest amounts in the second and subsequent harvest rotations and hence can lengthen payback periods. Also, we excluded soil carbon from our calculations because of uncertainties concerning its response to harvesting. Even small changes in soil carbon are potentially important because of the size of soil carbon pools. Our results showed reductions in deadwood carbon pools due to harvesting which may translate into lower soil pools; indeed, reduced deadwood amounts due to harvesting are well known (e.g. Hansen et al. 1991).

Our findings of long carbon debt payback periods are in general agreement with other studies. Holtsmark (2012) found a payback period of 190 years for a relatively modest decrease in steady-state carbon stocks of 10% (95 Mt); however, the debt in this case had to be paid off by a relatively modest increase in the harvest (0.76 Mt year^{-1}). In the present paper, for the same-sized landscape under a 166-year fire interval and 100-year rotation, the debt was much larger (217 Mt), but the annual harvest also was much higher (an average of 2.71 Mt), resulting in a shorter debt period (138 years using the same conversion efficiency factor as Holtsmark (2012) of 0.65). In one of the most comprehensive examinations of carbon debt payback periods in the use of wood bioenergy to replace fossil fuels, Mitchell et al. (2012) studied forests of the U.S. Pacific northwest under various bioenergy conversion efficiencies, land use histories, forest growth rates and biomass longevities, and harvest frequencies and intensities, and found that clearcut harvesting of primary (old growth) landscapes always resulted in payback periods of > 100 years irrespective of the forest type. Similar to us, they found that payback periods increased as rotation intervals decreased and were especially long for the shortest rotation periods (25 years in their case; see also Kurz et al. 1998; Smyth et al. 2017). In general, their payback periods for conversion of primary ecosystems to managed systems averaged longer than ours: for an average biofuel conversion efficiency of 0.51, they were always > 500 years, and usually > 1000 years. Laganière et al. (2017) also found that use of stemwood from medium- or slow-growing forests to eliminate coal, oil or natural gas showed no atmospheric benefits for > 100 years, although they examined the debt incurred in achieving stand regrowth rather than the carbon debt per se (see also Bernier and Paré 2013).

In northwestern Ontario, Ter-Mikaelian et al. (2015) reported a relatively low payback time to carbon parity under intensified stemwood harvests to replace coal (91 years), despite using the same conversion efficiency parameter as us (our central estimate of 0.55). Two methodological differences in combination likely contributed to their relatively low value. First, they relied on provincial estimates of ages of primary forests, which often underestimate actual ages because they are based on tree heights rather than time since the most recent fire (Etheridge and Kayahara 2013). Secondly, they assumed that stand volumes decreased in the old-growth phase (from a maximum at 130 years of age to 65% lower by 250 years of age), whereas we assumed that they remained constant. Concerning this last difference, it appears that more research is needed in order to better characterize volume dynamics of old-growth stands. At present, declines in stand volumes assumed by the province of Ontario are dependent in part upon successional rules that are based on expert judgement (Etheridge and Kayahara 2013). Unfortunately, existing empirical information is from even-aged stands, despite the fact that old boreal stands exhibit an uneven-aged structure with multiple cohorts of trees (Harvey et al. 2002; Etheridge and Kayahara 2013). We found little empirical evidence to suggest declines in volume with age (see Online Resource, GMV in old-growth forests); however, we had information for only our most northerly management unit.

These payback periods of many decades or more beg the question: are there situations in which clearcut harvesting of boreal primary forest landscapes will act to mitigate CO_2 emissions within significantly shorter time periods? Clearly, concerning electricity generation, possibilities are more favourable for substitution of the most emission-intensive fossil fuels. In Canada at least, however, such possibilities are relatively rare. The average emission intensity of electricity generation in Canada in 2015 was 140 g CO_2e kWh^{-1} (Environment Canada 2017), which is much lower than natural gas (approximately 550 g CO_2e kWh^{-1} (Weisser 2007)). Ontario had zero coal emissions by 2015 (Environment Canada 2017) and the average emission intensity in 2015 was 40 g CO_2e kWh^{-1}, which is lower than the biomass average of 70 g CO_2e kWh^{-1} reported by Weisser (2007) (assuming carbon neutrality). Use of wood energy to replace fossil fuel heating has more favourable conversion efficiencies than for electricity generation. However, Laganière et al. (2017) again found no emissions savings for medium- and fast-growing trees within 100-year time frames, except for medium-growing trees in a coal-heating scenario. Estimates suggest than emissions from present-day wood product harvesting in North America similarly would not pay off our estimated carbon debts in a timely fashion. In their life cycle analysis of the U.S. forest sector in 2005, Heath et al. (2010) reported a net transfer of emissions to the atmosphere of 103.5 Tg CO_2e (assuming carbon neutrality). For 1990–2015, Environment Canada (2017) estimated a net carbon balance for Canadian harvested wood products of approximately − 42 Tg CO_2e per year (their Fig. 5-6) in a managed forest of 190 million ha, including emissions from harvests and processing and transfers to stored wood products, although it was assumed that all wood transferred to landfills was instantly oxidized. If we apply that same rate to the carbon debt calculated here; that is, a debt of 217 Tg C incurred during the conversion of a 7.5 million ha natural landscape with a 166-year fire-return interval to a managed one with a 100-year rotation, then the pay-off period would be approximately 480 years. Even at the lowest carbon balance that they observed (− 65 Tg CO_2e per year), the pay-off period would still be approximately 310 years.

Several studies have suggested that the most favourable mitigation scenario is when wood products are long lived and are judged against other high-emission building materials such as cement and steel (e.g. Sathre and O’Connor 2010; Lemprière et al. 2013). Indeed, the primary use of wood from boreal harvests is often in the form of relatively long-lived wood products such as timber, rather than as biofuel. In one of the most comprehensive examinations of various wood product portfolios, Matthews et al. (2014) found that emissions from wood product mixes (including sawn timber, particle board, fibreboard and fuel) had slightly better mitigation potential than pure use of wood as biofuel; however, in both cases, better mitigation was provided by reference scenarios in which harvesting of managed forests instead was stopped and forests were allowed to accumulate carbon. For example, under a 100-year time horizon for conifer production forests, savings were approximately 5, 4 and 8 t CO_2e ha^{-1} year^{-1} respectively for wood product mixes, fuel only and reference scenarios (see their Figs. 5.12 and 5.15). However, the situation changed when wood product mixes were judged relative to counterfactual scenarios in which wood was replaced by other materials such as cement, plasterboard and plastic. Under that assumption, wood product mixes generally performed better than wood fuel or reference scenarios and for conifer forests were as high as 24 t CO_2e ha^{-1} year^{-1} (100 year time horizon). However, this finding was importantly affected by two assumptions. All of their highest ranking mixes included particle board with 70% recycled fibre. When this percentage dropped to zero, emission benefits for the highest ranked mix decreased by 48%. Similarly, in the “standard” mixes considered by Matthews et al. (2014), end-of-life disposal of wood products was assumed to be incineration for power. If disposal instead was to “wet” landfills, over a 100-year time horizon, none of the mixes performed better than the reference scenario, and even with energy capture from such landfills, only the best mixes (all of which included recycled particle board) performed better than the reference scenario (Matthews et al. 2014). As a final note, Matthews et al. (2014) considered one forest type in which carbon stocks were currently high, and thus where management could potentially incur a carbon debt (their “neglected broadleaf forests”). For these forests under 100-year time frames, none of the wood product uses, irrespective of counterfactual or end-of-life assumptions, provided emission’s savings relative to the no-harvest reference (their Fig. A1.33).

Compared with stemwood, many authors have identified more favourable mitigation potential for harvest-associated salvaged and residual woody debris, which incur a smaller or non-existent debt compared with biomass (e.g. Lamers and Junginger 2013; Ter-Mikaelian et al. 2015; Laganiere et al. 2017). Although use of such feedstocks might help to reduce the length of the carbon debt payback period, it is important to note that the carbon debt applies to the entire suite of management activities in a landscape, including the management activities that are generating the residuals. We did not consider such residual use in the present study for two reasons. First, although the long-term effects of forest harvesting on soil carbon remain uncertain, soil carbon reductions are more likely as harvest intensity increases (Kishchuk et al. 2016; Hume et al. 2018; Wan et al. 2018). Second, and more firmly established, is the importance of deadwood for biodiversity. Arguably, the greatest overall impact of forest management on biodiversity is loss of woody debris habitats, which include snags and downed dead wood (Siitonen 2001; Tikkanen, et al. 2006). Literally, 100s of species are threatened and endangered in the EU as a result of this aspect of forest management (Rassi et al. 1992 cited by Siitonen 2001; Tikkanen et al. 2006; Nieto and Alexander 2010). Part of the problem is the nature of logging itself: compared with natural forests, harvesting-associated exports of wood coupled with improved tree health results in reduced accumulations of dead wood from disturbances and stand development (e.g. Hansen et al. 1991). Harvesting of residuals has the potential to magnify this effect, with even more serious implications for biodiversity (Berch et al. 2011). Certainly, it makes little sense to consider climate change mitigation strategies outside of an overall framework of ecological sustainability.

Certain modifications to harvesting practices might help to reduce payback periods, including reduced forest degradation, increased stand growth rates, alteration of existing harvesting practices to reduce impacts on soil C and improved regeneration (Lemprière et al. 2013; Laganière et al. 2017). However, probably the simplest and most effective approach is to reduce the magnitude of the debt in the first place by managing forests to maintain or even maximize carbon stocks (Mitchell et al. 2012; Law et al. 2018), for example by reducing harvesting frequency and/or intensity. An interesting experiment in this regard was undertaken by Hennigar et al. (2008), who used wood supply modelling to investigate scenarios designed to maximize carbon storage either in wood products, forest stocks or wood products plus forest stocks. Compared with wood product maximization, wood product plus forest carbon maximization had lower harvest levels, longer rotations and an increased reliance on partial harvesting compared with clearcutting. Triviño et al. (2017) used multi-objective optimization to investigate trade-offs between timber, carbon storage and biodiversity in a production landscape in central Finland. Within a 50-year time frame, by modifying current practices to increase the area of extended rotations and by applying less thinning, they could maintain timber revenues at 80% of the maximum, carbon storage at > 80% of the maximum and biodiversity habitat at > 90% of the maximum. In this regard, multi-cohort forest management of boreal forests, which envisions inclusion of partial harvesting to better emulate the structural diversity of natural forests (Harvey et al. 2002), could prove useful in providing both carbon and biodiversity benefits relative to clearcut harvesting.

However, as indicated by Triviño et al. (2017), carbon and biodiversity considerations both call for increased conservation and protection of primary forests (Watson et al. 2018), which are an increasingly rare and valuable resource worldwide. Any such protection reduces the magnitude of the carbon debt. In our analyses, we did not include any effects of future climate change, whose overall impacts on forest ecosystems are very uncertain. Intact forests may provide key advantages in this regard; for example, natural interconnectivity allows for adaptive responses of species, and high intra-specific genetic diversity provides the raw materials for local adaptations and phenotypic plasticity (Watson et al. 2018). Some authors have suggested that because of increased future risks of boreal fires and associated emissions

(Flannigan et al. 2005; Balshi et al. 2009), older forests are of lesser value for climate change mitigation (e.g. Sharma et al. 2013). Fire suppression, which is already practiced extensively in the forest tenures of Canada, is useful as a mitigation strategy in this regard (Lemprière et al., 2013). However, it is also worth pointing out that the length of the fire-return interval had a relatively minor effect on carbon debt payback periods (Fig. 6), indicating that future increases in fire frequency may not negate carbon debts. In their investigation of future burn rates using 19 global climate model experiments for a study area geographically close to ours, Bergeron et al. (2010) found that even the most extreme burn rates projected for 2100 were within the range of fire-return intervals that we modelled. To our mind, the possibility of increased fires due to global warming is one of many threats that argue for reducing carbon debts in the first place and a shift in focus towards activities that maintain and rebuild forest carbon stores.

In conclusion, our calculations indicate that clearcut harvesting of boreal primary forests to replace fossil fuel use in electricity generation entails considerable carbon debt payback periods, which are in the range of 122–481 years for mid-range conversion efficiencies. Debts increased as the mismatch between natural fire intervals and rotation periods increased. More generally, we argue that the magnitude of such debts makes it unlikely that clearcut logging of primary forests can serve as a climate change mitigation strategy and that mitigation would be better served by increased conservation of primary forests and use of harvesting methods that entail lower offtakes over longer periods of time.

Acknowledgements We are indebted to M. Penner and S. Kull, who provided assistance in modelling stand volumes and implementing CBM-CF3, respectively. We would also like to thank anonymous reviewers for relevant literature and comments and J. Boan, F. Daviet, M. von Mirbach and J. Ray for comments.

Author's contributions JRM undertook the main analyses and wrote the first draft; BH provided key parameterization for calculation of conversion efficiency factors and contributed to a second draft; PWP identified key literature and contributed to a second draft.

Funding information The research was funded by Greenpeace Canada and the Natural Sciences and Engineering Research Council of Canada (Discovery Grant to JRM).

Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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