Changes in mean forest age in Canada's forests could limit future increases in area burned but compromise potential harvestable conifer volumes

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2 **area burned but compromise potential harvestable conifer volumes**

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Abstract (199 words)

Forest fire activity is projected to increase with climate change in Canada, but vegetation feedbacks are usually not considered. Using new information on the selectivity or avoidance of fire as a function of stand age and composition, we ran simple simulation models that consider the changes in the regional age matrices induced by fire and harvesting to project future burn rates. We also projected estimated future regional vulnerability of timber supply to fire by considering these new burn rates. The inclusion of age-related feedbacks would have a large impact on projected increases in burn rates, mostly in very fire active zone under aggressive climate forcing. Projected burn rates would still increase, but would be 50% less in 2100 than if projected without this biotic feedback in some zones. Negative feedbacks would be virtually non-existent when potential burning rates are below 1%, whereas realized burning rates would be lowered by more than 0.5 percentage points when potential burning rates exceed 2.5%. Including fire-vegetation feedbacks had virtually no impact on total volume harvested. As fire burn more old-growth coniferous stands, slightly negative impacts were projected on conifer harvested almost everywhere. These results underline the need to incorporate vegetation – fire feedbacks when projecting future burn rates.

Keywords: fire – vegetation feedbacks, boreal forest, forest fires, climate change, timber supply, Canada

Introduction

Recent shifts observed in burn rates and fire seasonality in boreal North America appear to be directly related to recent changes in climate patterns, mostly in temperature regimes (Gillett et al. 2004; Kasischke and Turetsky 2006). Further increases in area burned and the number of fires are projected as a result of warmer temperatures with increased anthropogenic climate forcing (e.g., Balshi et al. 2009a; Wotton et al. 2010; Boulanger et al. 2014; Girardin and Terrier 2015). These modifications of the fire regime could alter future forest ecosystems, notably by shifting species composition (Boulanger et al. 2016), reduce carbon stocks (Kurz et al. 2008; Balshi et al. 2009b), increase fire suppression costs (Podur and Wotton 2010; Hope et al. 2016) and reduce our ability to perform sustainable forest management (Gauthier et al. 2015).

Beside climatic (“top-down”) controls, it is increasingly acknowledged that burn rates are strongly affected by fuel characteristics as driven by dynamic forest properties such as cover, density, structure and age, known as “bottom-up” controls (Hély et al. 2001, 2010; Krawchuk and Cumming 2011; Héon et al. 2014, Terrier et al. 2013, Girardin et al. 2013a). Fire is itself a strong driver of forest properties (Keane et al. 2013), any fire-induced variations in these properties may generate substantial feedbacks on subsequent fire activity (Héon et al. 2014). Fire-prone landscapes could thus be self-regulating and resilient to fire (Héon et al. 2014; Parks et al. 2015). In some Canadian forest regions, it has been suggested that fire generates a negative feedback over a post-fire period of 40-50 years (Héon et al. 2014, Erni et al. in press) through the reduction of fuel availability and flammability. As a consequence, one may hypothesize that the temporary expansion of fire-resistant landscapes may attenuate a potential climate change-induced increase in

fire activity. However, because of a general lack of quantitative evidence (Héon et al. 2014), fire-vegetation feedbacks are seldom taken into account when projecting future fire activity within the Canadian forests (e.g., Flannigan et al. 2005; Balshi et al. 2009a; Boulanger et al. 2014, but see Krawchuk and Cumming 2011; Terrier et al. 2013; Girardin and Terrier 2015). Such exclusion of biotic feedbacks could have resulted in an overestimation of projected burn rates (Héon et al. 2014).

In addition to improving burn rate projections, the inclusion of biotic feedbacks could also provide information as to the future vulnerability of timber supply. Fire competes with harvesting by burning stands that would have been harvested in the future. Negative effects on harvest levels have been estimated to take place only when burn rates are roughly above $0.4\%.\text{yr}^{-1}$ (Boyчук and Martell 1996; Savage et al. 2010; Leduc et al. 2015). Recently, using historical harvest levels, Gauthier et al. (2015) identified specific forest management areas across Canada in which projected increases in fire activity might trigger periodic shortfalls in timber supply. However, their study did not take into account biotic feedbacks to fire. Such feedbacks could potentially lower increases in projected burn rate, but also decrease the proportion of mature conifers available for harvest. There is therefore a need to explore whether or not the inclusion of biotic feedbacks affects projected estimates of timber availability.

Using medium-resolution maps of fire (Guindon et al. 2014) and spatial estimates of forest properties (Beaudoin et al. 2014), Bernier et al. (2016) recently estimated fire selection ratios for classes of forest composition and age across Canada's managed forests. These quantitative fire selection ratios now offer the possibility to incorporate fire-vegetation feedbacks in order to provide more realistic estimations of future fire

activity. In this study, we project future burn rates across Canada by accounting for dynamic changes in the forest age matrix. Only the effect of age as a biotic feedback is considered in this study as potential changes in forest composition are too uncertain for their inclusion. Forest growth is also assumed to be unchanged. In situations where age feedbacks cannot directly be accounted for in projections (e.g., Bergeron et al. 2004, 2006; Flannigan et al. 2005; Balshi et al. 2009a; Girardin et al. 2013a), we propose that it be indirectly accounted for by applying the correction factor presented in caption of Fig. 3 to potential burn rate estimates. We further built a model to predict future realized burning rates as a function of potential burning rates (i.e., without fire selectivity) in order to provide a simple solution to correct burn rates in situations where age feedbacks cannot directly be accounted for in projections (e.g., Bergeron et al. 2004, 2006; Flannigan et al. 2005). We hypothesized that the decrease in mean stand age induced by increased burn rates will result in projected burn rates lower than those predicted without the inclusion of such a feedback (e.g., Boulanger et al. 2014). We further hypothesized that the inclusion of biotic feedback to the increased burn rate will result in lower available area and volume for harvest as well as lower coniferous proportions in harvested volumes than when not considering biotic feedback to fire activity.

Material and methods

Our starting point for projection of future burn rates was the Boulanger et al. (2014) study in which future monthly burn rates were projected within Homogeneous Fire Regime (HFR) zones delineated within forested regions of Canada (Figure 1). The projections, which do not take vegetation feedbacks into account, are based on Multivariate Adaptive Spline Regressions (MARS) models using HFR-specific monthly weather metrics.

Models were built using climate and fire data covering the 1959 – 1995 period. Monthly climate variables were computed from daily data obtained from Environment Canada weather stations that were interpolated to the centroid of each HFR zone using BioSIM v10.0.6.20 (Régnière et al. 2014). BioSIM projected daily maximum and minimum temperatures (°C), precipitation (mm), mean daily relative humidity and wind speed by matching georeferenced sources of weather data (weather station with daily weather data) to spatially georeferenced points, adjusting the weather data for differences in latitude, longitude, and elevation between the source of weather data and each cell location by spatial regressions. More details about these fire models can be found in Boulanger et al. (2014). Fire data comes from the Canadian National Fire Data Base. Fire data in models were restricted to large fires, i.e., above 200 ha, as data from smaller fires are known to be incomplete, especially those that occurred before 1980 and in remote areas. These fires were responsible for 97% of the area burned across Canada during 1959-1997 (Stocks et al. 2003). Simulations for the current exercise were done for all HFR zones, except the Eastern Subarctic and the Western subarctic zones on account of poor MARS model fit for burning rates estimations, or the low reliability of initial forest age data, as well as the Pacific zone because of unrealistic behaviour of the conifer yield curves given the area's extreme precipitation values.

Climate data and regional climate-driven burn rates

In Boulanger et al. (2014), future burn rates were projected according to the SRES A2 climate scenario used in the previous IPCC fourth assessment report. In this study, we used updated climate projections based on three Representative Concentration Pathways (RCP, e.g., van Vuuren et al. 2011) scenarios, namely RCP 2.6, RCP 4.5 and RCP 8.5.

123 The RCP 2.6 scenario represents a situation where radiative forcing peaks at $\sim 3 \text{ W.m}^{-2}$
124 before 2100 and then declines to reach 2.6 W.m^{-2} by 2100. In the RCP 4.5 scenario,
125 radiative forcing is assumed to stabilize at 4.5 W.m^{-2} after 2100 without an “overshoot”
126 pathway. Conversely, in the RCP 8.5 scenario, the forcing reaches 8.5 W.m^{-2} in 2100 and
127 continues to increase for some time afterwards. The appropriate outputs from the
128 Canadian Earth System Model version 2 (CanESM2) were downloaded from the World
129 Climate Research Program (WCRP) Climate Model Intercomparison Project Phase 5
130 (CMIP5) archive.

131 Using the stochasticity functionality of BioSIM (Régnière et al. 2014), we simulated
132 three thousand daily time series lasting one year for all combination of HFR zone x
133 climate scenario x time period. From these daily time series, we then derived the standard
134 components of the Canadian Forest Fire Weather Index System (CFFWS) (Van Wagner
135 1987) as well as other temperature- and precipitation-related variables on a monthly-
136 basis. With these variables, regional climate-driven burn rates projections ($BR_{regclim}$) were
137 computed for all three thousand yearly weather data sets for each HFR zone x climate
138 scenario x time period using HFR zone-specific MARS models developed by Boulanger
139 et al. (2014).

140 *Fire selection ratios*

141 Estimates of fire selection ratios for classes of forest age and composition were produced
142 by Bernier et al (2016). Briefly, maps of 2001 forest properties of Canada’s managed
143 forests (Beaudoin et al., 2014) and yearly 2001-2011 Canada-wide maps of fire and
144 harvest (Guindon et al., 2014), both on the 250m MODIS grid, were used to obtain 2001

estimates of age and composition of all pixels with a forest cover ($\geq 80\%$ ($n = 76\,678\,906$) identify pixels, and to identify from this set all pixels that had burned between 2002 and 2011 ($n = 2\,739\,728$). All burned and unburned pixels were binned by HFR zone (Table 1) in 12 forest cover classes composed of three age classes, i.e., young (0 to 29 years), mature (30 to 89) and old (90+) and for four composition classes i.e., coniferous ($> 75\%$ in conifer species), mixed coniferous (50-75% in conifer species), mixed hardwood (25-50% in conifer species) and hardwood ($< 25\%$ in conifer species). Stand properties such as forest composition and age were proved to be powerful and independent classifiers of fire selectivity (Bernier et al. 2016). Indeed, composition (conifers or deciduous species) represents a valuable proxy for flammability, i.e., the propensity to burn, while fuel load, i.e. the amount of flammable biomass, is well-represented by stand age. Lower fire risks have already been associated with young or deciduous stands in regional analyses of fire statistics and forest composition (Héon et al. 2014; Krawchuk and Cumming 2011). Within each HFR zone, the fire selection ratio for a given forest cover class was calculated as the fraction of burned pixels in that cover class divided by fraction of total pixels in that cover class. For a given class, a selectivity ratio of 1 indicated a regional burn rate proportional to its regional availability. A value above 1 indicated fire preference, while a ratio below 1 indicated fire avoidance. These ratios proved to be not affected by large burn rate differences among-HFR and were therefore averaged across HFR zones into Canada-wide mean selection ratios. These mean selection ratios were then attributed to individual pixels using composition and age information, and were combined with the HFR zone's current or projected regional burn rates to estimate the burn probability of each pixel at time t .

Harvesting data

Regional harvesting levels were retrieved from MODIS-based annual (2001-2011) forest disturbance maps (Guindon et al. 2014) combined with Beaudoin et al. (2014) forest properties maps at a 250m resolution. The cumulative aboveground biomass harvested during 2002-2011 was estimated at the 250-m grid cell level and was summarized by “management areas” (i.e., either by forest management units for public lands or by ecodistricts for private lands). These harvesting levels were then directly translated in regional proportions of land harvested at each time step.

Calculation of pixel-level volumes

Pixel-level yield curves adjusted for conifer species group and for hardwood species group, developed by Gauthier et al. (2015) for the same forest cover dataset than in our study, were used to estimate changes in total volume and in volume harvested in each HFR zone. These curves were formulated as:

(eq. 1)
$$\log(V) = \beta_0 + \beta_1 + \beta_2 + \frac{\beta_3 + \beta_4 T + \beta_5 P}{A}$$

where V is the pixel-level volume (in $\text{m}^3 \cdot \text{ha}^{-1}$) based on all tree’s woody and foliar components, A is stand age, T is historical (1970–2000) mean annual air temperature of the pixel ($^{\circ}\text{C}$), P is historical mean annual precipitation of the pixel (mm), and β_0 to β_5 are adjusted parameters (see Ung et al. 2009). The conversion from logarithmic units to arithmetic units entailed the use of a correction factor, as suggested by Duan (1983) (in Ung et al. 2009):

(eq. 2)
$$V = \exp[\log(V)] * C_D$$

where C_D , the Duan correction factor, is equal to the mean of exponentiated residuals. For each pixel, we merged the results of the two curves according to the proportion of coniferous and hardwood species at time $t = 0$ based on the Beaudoin et al. (2014) forest composition maps. The resulting composite yield curve was then rescaled according to a ratio of measured to modeled V at age at time $t = 0$ to force the curves to locally adjust the observations (Gauthier et al. 2015). We did not attempt to model changes in growth rate as a result of predicted changes in precipitation or temperature using eq. 1 as its the formulation was designed to be only descriptive of the current interaction between climate and species distribution (Ung et al, 2009).

Simulations

For each HFR zone and climate scenario combinations, we ran non spatially-explicit simulations including both fire and harvesting as stand replacing disturbances. Two sets of simulation were conducted, i.e., one including the impact of fire selectivity relative to stand age and composition classes and another where fire selectivity was not considered. For each set, sixty simulations were run for 100 years using a 5-yr time step, starting in 2000. Initial pixel-level age, volume and coniferous/hardwood fraction were derived from the 2001 forest cover maps of Beaudoin et al. (2014). Pixels in which the forest was identified as recently disturbed prior to 2001 (Guindon et al. 2014) or comprising < 80% of vegetation cover (Beaudoin et al. 2014) at time $t = 0$ were not considered in simulations.

Stochasticity at each simulation time step stemmed from the random selection i) of pixels to be burned, ii) of pixels to be harvested, and iii) of the regional climate-driven burn

probability ($BR_{regclimt}$). At each time step, a value of $BR_{regclimt}$ was randomly drawn from the 3000 values available for that time period under the given climate scenario for that HFR zone. As in Bernier et al. (2016), burn probabilities (eq. 3) were normalized within each HFR zone by dividing by the mean probability of the twelve forest cover classes (Table 1) at time $t = 0$ ($BaselineMeanP$), thereby transforming the probabilities into a set of normalised selection ratios (Manly 1993).

The burning of a given pixel at time t was then drawn from a binomial distribution with probabilities $Prob_{burn}$ estimated from:

$$(eq. 3) \quad Prob_{burn} = BR_{regclimt} * cover_m * linear_{mod} * t_{step} / BaselineMeanP$$

where t_{step} is time step length in years (i.e., 5), $cover_m$ (Table 1) is the fire selectivity ratio of the forest cover class of this pixel, and $linear_{mod}$ is a correction factor included to consider a linear trend in $BR_{regclimt}$ within each 30-yr periods. The $cover_m$ was set to 1 for all forest cover classes in simulation where fire selectivity was not considered. Burn rate was then assessed as the annual proportion of burned pixels at each time step. The *potential* burning rate (BR_{pot}) was defined as the burning rate simulated when no fire selectivity was included in simulations whereas the *realized* burning rate (BR_{rea}) corresponded to the burning rate calculated in simulations with fire selectivity.

Harvesting was applied at the management area level after burning in the simulations. For a given time period and management area, the harvesting rate was area-based and was initialized at the mean yearly 2001-2011 level observed in the Guindon et al (2014) dataset in each management area. Only pixels that reached commercial maturity, here

arbitrarily defined as $V = 100 \text{ m}^3 \cdot \text{ha}^{-1}$, could be randomly selected for harvesting (as in Gauthier et al. 2015, see also Raulier et al. 2013). At each time step, the harvest rate was allowed to adjust downward with the decreasing availability of harvestable stands as a result of both fire and harvest. Indeed, simulated harvest rates were capped to the number of harvestable pixels when this number was lower than the historical harvest rates. A random number of pixels were then selected for harvesting according to the adjusted regional harvesting levels. Coniferous and hardwood volumes within harvested pixels were considered to have been entirely removed in their respective proportion. The age of pixels that were selected for burning or harvesting was then reset to 0, but their composition remained unchanged from the 2001 estimates provided by Beaudoin et al (2014). By doing so, we thus assume (as in Bernier et al. 2016) that burn rates in harvested and post-fire stands are similar. Other pixels were aged accordingly to time step length, while their new volume was estimated following equations 1 and 2.

The burning rate was computed for each HFR zone at each time step for each simulation. In order to assess if we could develop a correction factor for burned rates that were estimated without considering the vegetation selectivity, we modeled realized burning rates as a function of potential burning rates as averaged over the 100-yr period using the following nonlinear least square model:

$$(eq.4) \quad BR_{rea} = v * BR_{pot} / (k + BR_{pot})$$

where v (maximum achievable BR_{rea}) and k (potential burning rate at which rate of increase in BR_{rea} is half of v) are the two constants to parameterize. The sixty burning rates coming from the same simulation dataset of a given HFR zone were averaged in

order to avoid pseudoreplication, leaving four observations per HFR zone (one for each forcing scenario, total $N = 52$). In parallel, we also estimated the following variables at the management area level in all simulations: i) the realized harvesting rates, expressed as the yearly proportion of vegetated area harvested and ii) the total, conifer and hardwood volume harvested. Simulations were performed using R 3.2.4 (R Core Team 2016). The nonlinear regression model was fitted using the *nls* function in R.

Results

Impacts on burn rates

Effects of age-related biotic feedback on projected increases in burn rate was important in the great majority of HFR zones, but was more pronounced in zones with higher projected fire activity under the most important climate forcing RCP 8.5, i.e. Great Slave Lake, Lake Athabasca, Lake Winnipeg, Eastern James Bay, Western Ontario and Southwestern Yukon. Therein the inclusion of age-related biotic feedback generated large drops in projected burn rates increases as compared to projections that did not incorporate such feedbacks (Fig. 2). These feedbacks lowered projected burn rates increases by as much as half the potential burn rate, representing a difference of 2 to 8 percentage points in annual area burned, under the RCP 8.5 climate scenario. Furthermore, high fire activity as simulated under baseline condition was sufficient in Eastern James Bay and Lake Athabasca zones to generate negative feedback and to decrease annual area burned by 0.5 – 1 percentage points. Rather strong negative feedbacks (25-30%) were also simulated for The Great Bear Lake and Interior Cordillera zones under RCP 8.5 by 2100. Important but lower negative feedbacks were also generated for all these zones under RCP 2.6 and

RCP 4.5 (Fig. 2). Negative feedbacks were minimal regardless of climate forcing in Eastern Temperate, North Atlantic, and Western James Bay zones. As a rule of thumb, potential and realized annual burning rates when averaged over a 100 year period were generally similar below 1% (Fig. 3). Burning rates clearly diverged at higher values: realized annual burning rates was lowered by more than 0.5 percentage point when the potential burning rate exceeded 2.5%, and by more than 2 percentage points when the potential burning rate exceeded 5.5% (Fig. 3).

Even with these negative feedbacks, burn rates were projected to increase in virtually all HFR zones as a result of increase anthropogenic climate forcing when compared to actual (2000) burning rates (Fig. 2). Increases remained substantially higher under the most aggressive (RCP 8.5) climate forcing scenario. Burn rates were projected to remain the highest in HFR zones located in central Canada (Lake Athabasca and Lake Winnipeg HFR zones), reaching *ca* 6 - 7% by 2100 under RCP 8.5. Divergences in burn rates among the three climate forcing scenarios were perceptible mostly after *ca* 2050 in the majority of HFR zones.

Impacts on harvesting

Accounting for biotic feedbacks on projected future burn rates had virtually no impact on harvested area and total volume harvested (Figures 4 and 5). Exceptions were for Eastern James Bay where volume harvested would be *ca* 60% higher by 2100 under RCP 8.5 when considering biotic feedbacks. However, the inclusion of biotic feedback in projections of burn rates slightly reduced the proportion of conifer potentially harvested in Eastern James Bay, Interior Cordillera and Western Ontario as compared to

simulations that did not include such feedbacks (Figure 6). The proportion of potential harvested conifer volume was lower by 5% to 10% under RCP 8.5 as well as, to a lesser degree, under milder forcing scenarios. Virtually no impacts were simulated elsewhere (Figure 6).

Compared to current (2000) levels, higher burn rates very slightly decreased potential harvestable areas to levels lower than those necessary to maintain historical harvesting rates (in terms of area harvested), but only in Eastern James Bay only under RCP 8.5 beyond 2075 (Fig. 4). Negative impacts when compared to current volume harvested remained minimal elsewhere with the exception of Interior Cordillera for which harvestable volume would drop by 20-30% by 2100 depending of climate forcing. The potential harvestable volume was projected to increase regardless of forcing scenario and most markedly in Eastern Temperate and Southern Cordillera (Figure 5). However, decreases in the proportion of potential harvestable conifer volume were projected especially for most fire active zones under the most aggressive climate forcing. Decreases were important in Eastern James Bay, Western Ontario and Interior Cordillera (10 - 20% under RCP 8.5 by 2100).

Discussion

This study provides nationwide projections of future burn rates that account for climate change effects on fire risks and dynamic feedbacks between fire and mean landscape age. Our approach based on Canada-wide maps of disturbances and vegetation features contrasts with approaches taken by typical process-based models in which biotic feedbacks to burn rates are modelled using complex and interacting climate, fuel and ignitions submodels (e.g., Arora and Boer 2005, Scheller et al. 2007, Keane et al. 2011). Given their intrinsic complexity, these models have to sacrifice either spatial extent (e.g., forest landscape models) or resolution (e.g. Terrestrial Ecosystem Models), thereby compromising their ability to obtain Canada-wide assessments of biotic feedbacks to future fire activity and their inherent impact on timber supply. The issue of sustainable forest management within a climate change context urgently needs to be addressed, and advances as proposed here based on the characterisation of disturbances and vegetation features by remote-sensing may offer valuable insights.

Future burn rates could be substantially constrained by biotic feedbacks, especially in zones where burn rates were projected to be high (Boulanger et al. 2014, Balshi et al. 2009a). Indeed, a steady decrease in mean forest age could temper future increases in burn rates by more than 50% in some HFR zones in 2100, notably in Lake Winnipeg, Lake Athabasca, Eastern James Bay and Great Slave Lake. Comparable attenuations under warmer and drier climates were projected for the occurrence of large fires when considering negative vegetation feedback resulting from boreal needleleaf transition to boreal mixedwood landscapes (Girardin et al. 2013a). Likewise, Héon et al. (2014) showed that increasing the proportion of young stands significantly reduced the potential

burn rates of a highly fire-prone region within the boreal forest. Nevertheless, biotic feedbacks would not be sufficient to completely offset the climate-induced increase in fire activity relative to baseline conditions in most HFR zones as projected burn rates are likely to attain or exceed the range of natural variability in burn rates observed within the last millennia (Bergeron et al. 2004, Girardin et al. 2013b). In other zones and under mild climate forcing, negative feedbacks were rather small to non-existent: original projections of burning rates were low and current forest landscape cover proportions were not sufficient to significantly lower burn rates. Indeed, we found that negative biotic feedbacks would be virtually non-existent when potential burning rates are below 1%. Burn rates that were previously projected to values below 1% without consideration of biotic feedbacks should not be corrected for negative feedback. Most regional projections of future burn rates for most of Canada correspond to these conditions (e.g., Flannigan et al. 2005; Bergeron et al. 2006; Balshi et al. 2009; Boulanger et al. 2014; Suppl. Mat S1). However, realized burning rates would be lowered by more than 0.5 percentage points when potential burning rates exceed 2.5% over a 100-yr period. There is hence clearly a need for accounting for such feedbacks when projecting future burning rates. In situations where age feedbacks cannot directly be accounted for in projections (e.g., Bergeron et al. 2004, 2006; Flannigan et al. 2005; Balshi et al. 2009a; Girardin et al. 2013a), we propose that it be indirectly accounted for by applying the correction factor presented in caption of Fig. 3 to potential burn rate estimates. By doing so, it can be shown that studies projecting large increases in area burned under climate change be biased by the non-accounting for age feedbacks (Suppl. Mat. S1).

We found that the inclusion of age-related feedbacks could lower the proportion of conifer potentially harvested while, contrary to our expectations, having no impact on total harvested volume. The same applies for harvestable stands as for mean stand age (Suppl. Mat. S2). Such apparently contradictory results stem from our premise of no harvesting preference between coniferous and hardwood stands. The selection ratios of Bernier et al. (2016) give conifers a higher-than-average risk of burning and hardwoods a lower-than-average one. As time progresses, in the most fire-prone HFR zones, the proportion of conifer stands that reach commercial maturity (above $100 \text{ m}^3 \cdot \text{ha}^{-1}$) thus tends to decrease while that of deciduous stands tends to increase, giving as a result a either rather stable or slightly decreasing (when burn rates are high) area in mature coniferous stands available for harvest. Harvesting substitution of coniferous for hardwood stands is highly theoretical at this point and would imply a change in forest management paradigms for which the short-term achievability remains to be explored. Currently, softwood comprises the great majority of volume harvested in Canada's forest regions (Natural Resources Canada 2015), and management strategies (including plantations) are largely implemented to favor softwood species over hardwood species. Consequently, current harvesting strategies directly "compete" with fire as they both tend to "select" mature coniferous stands. Including harvesting preferences in simulations could thus reveal a much worst portrait in future potential harvestable volume.

Our results concur with those of Gauthier et al. 2015a in which forest regions in interior British Columbia, northwestern Ontario and north central Quebec (see Suppl. Mat. S3) may be at higher risk of timber shortfall especially within management areas where tree growth is slow, timber harvest is important and projected increases in burn rate are

383 significant. When compared to the situation prevailing in 2000, several FMUs located in
384 these zones would experience a decrease in the proportion of harvestable stands (i.e.
385 stands with merchantable volume greater than $100\text{m}^3\text{ha}^{-1}$) and hence total volume
386 harvested. Our analyses also suggest that, even with the current baseline fire and
387 harvesting conditions, timber availability in some of these FMUs may decrease within the
388 next decades (Suppl. Mat. S3), with potential impacts on the supply value chain even
389 without considering increased climate forcing (Irland et al. 2001; Williamson et al. 2009;
390 Gauthier et al. 2015a). Prior work has suggested that salvage logging can mitigate but
391 never eliminate fire impacts on timber availability (Leduc et al, 2015).

392 There are numerous limitations to our analyses. Simplistic assumptions about fire
393 selectivity that do not incorporate complex fire initiation and fire spread functions as
394 computed by process-based models (e.g., Burn-P3, FIREBGCv2, Parisien et al. 2005;
395 Keane et al. 2011) were used in our model. As such, fine-scale topographic and weather
396 conditions are not taken into account in burned area projections. As mentioned above, the
397 extent at which our analyses were performed impedes the use of these complex process-
398 based fire models. Also, we have not incorporated the extent to which successional
399 changes in forest composition e.g., from conifers to hardwood species, could act as an
400 additional feedback mechanism in response to increased fire activity. The reason for this
401 exclusion is simply uncertainties regarding post-fire succession rules as affected by
402 climate change at the scale of Canada. Possible conversion of late-succession coniferous
403 stands to mixed- or hardwood stands could further strengthen negative feedbacks
404 considering higher foliar moisture loading and lower flammability for broadleaf species
405 (Päätaalo 1998; Hély et al. 2001) and could further decrease harvested conifer volume.

Furthermore, anticipated northward migration of hardwood mesophytic species at the expense of boreal coniferous species, notably along the boreal-temperate transition zone (Boulanger et al. 2016, McKenney et al. 2011) could also further hinder fire activity (Carcaillet et al. 2010; Girardin et al. 2013a; Terrier et al. 2013). Other partial or stand-replacing disturbances (e.g., windthrow, insect outbreaks) that would have further lowered mean age were not considered. Also, we used Bernier et al. (2016) fire selectivity ratios for which similar values are attributed to young stands of all ages between 0 and 29 years. Recent studies (Erni et al. 2016, Héon et al. 2014) found that negative age-related biotic feedbacks exponentially decrease during this successional stage. Consequently, feedbacks might be much more important than simulated in zones where projections suggested the potential for very short fire return intervals (e.g., < 20 years). Furthermore, fire selectivity ratios computed by Bernier et al. (2016) did not consider different potential flammability between e.g., post-fire and harvested stands. Both of these stand types are likely to evolved different fuel load and type, notably fine fuel abundance, making recently harvested stands being more prone to fire initiation than post-fire stands (Krawchuk and Cumming 2009). One might then expect higher burn rates than simulated in areas where harvest is a significant component (e.g., Eastern James Bay and Interior Cordillera). In addition, our projections imply stable fire suppression efficiency in the future. However, more frequent period of high fire load might significantly decrease the efficiency of fire protection agencies if management resources are kept unchanged (Podur and Wotton 2010), thus leading to higher overall burn rates. Continuous monitoring will be needed to document potential changes in biotic feedbacks.

429 By integrating biotic feedbacks on fire activity, our analyses likely reduced the
430 uncertainties related to the projection of future burn rates, and could therefore improve
431 projections of nationwide e.g., carbon budget (Bond-Lamberty et al. 2007), smoke
432 (Anderson 2013), delivered wood costs (McKenney et al. in prep), biodiversity (Stralberg
433 et al. 2015) and community vulnerability to fire (Beverly and Bothwell 2011). Our
434 analyses suggested that some forest regions might be at higher risk of timber shortfall
435 especially within management areas where tree growth is slow and projected increases in
436 burn rate are significant (Gauthier et al. 2015). As projected climate-induced decrease in
437 forest productivity were not included in this study, some regions might face a double-
438 whammy, i.e., a decrease of harvestable volume through higher fire activity and
439 decreased climate-induced productivity along with potential changes in softwood
440 proportions available for harvest. Serious impacts on the supply value chain are thus to be
441 expected (Irland et al. 2001; Williamson et al. 2009; Gauthier et al. 2015) with potential
442 effects on market prices of forest products and consumer preferences (McCarl et al. 2000;
443 Hanewinkel et al. 2012). Given the extent to which fire selectivity and biotic feedbacks to
444 fire might impact wood type and wood volume available, we therefore advocate for their
445 thorough consideration when projecting future timber resources. In this context, rapid
446 adaptation of the forest sector is paramount (Lemprière et al. 2008; Williamson et al.
447 2009; Ochuodho et al. 2012; Edwards and Hirsch 2012).

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Table 1. Fire selectivity ratios ($cover_m$ in eq. 3) as a function of forest age and composition classes used in simulations considering biotic feedback to fire activity. A ratio of 1 indicated that regional burn rates for a given each class was proportional to its regional availability while values above 1 indicated fire selection while ratio below 1 indicated fire avoidance.

Composition classes (% of conifer)	Age classes		
	Young (0 – 30 yrs)	Mature (30 – 90 yrs)	Old (> 90 yrs)
Conifer (>75%)	0.80	2.00	2.90
Mixed-Conifer (50-75%)	0.43	1.16	1.79
Mixed-Hardwood (25 – 50%)	0.22	0.57	0.96
Hardwood (< 25%)	0.15	0.40	0.63

641 Figure captions

642 Figure 1. Homogeneous fire regime (HFR) zones as delineated in Boulanger et al. (2014)
643 along with their associated burn rates ($\%.\text{yr}^{-1}$) for the 1959-1999 period. EJB: Eastern
644 James Bay; ET: Eastern Temperate; GBL: Great Bear Lake; GSL: Great Slave Lake; IC:
645 Interior Cordillera; LA: Lake Athabasca; LW: Lake Winnipeg; NAT: North Atlantic; SC:
646 Southern Cordillera; SP: Southern Prairies; SY: Southwestern Yukon; WJB: Western
647 James Bay; WO: Western Ontario. Eastern Subarctic (ES), Pacific (P) and Western
648 Subarctic (WS) HFR zones were not included in the analyses.

649 Figure 2. Mean burn rates ($\%.\text{yr}^{-1}$) according to different anthropogenic forcing scenarios
650 when considering (solid lines) or not (dashed lines) fire – vegetation feedbacks within
651 each HFR zone. Black: Baseline climate; Blue: RCP 2.6; Yellow: RCP 4.5; Red: RCP
652 8.5. Note that the scale of y axis is variable.

653 Figure 3. a) Realized burning rates as a function of potential burning rates. Recall that
654 these rates were averaged over the 100-year period. See figure 2 for color meaning. The
655 1:1 line is shown dashed. The full black line illustrates the nonlinear fitted model where
656 $BR_{rea} = 9.466 * BR_{pot} / (9.254 + BR_{pot})$. $N = 52$; Pseudo-Rsquare = 0.992. b) Importance
657 of the negative feedback (% of burning rate) as a function of potential burning rates. This
658 basically represents the 1:1 line minus the full black line in figure 3a.

659 Figure 4. Proportion of the HFR zone total area harvested (%) per year according to
660 different anthropogenic forcing scenarios when considering (solid lines) or not (dashed
661 lines) fire – vegetation feedbacks. See figure 2 for color meaning.

662 Figure 5. Same as figure 4 but for total harvested volume ($\text{m}^3 \cdot \text{ha}^{-1} \cdot \text{yr}^{-1}$). Only HFR zones
663 where initial yearly proportion of harvested area was above 0.05% are shown.

664 Figure 6. Proportion of conifer volume harvested (%) according to different
665 anthropogenic forcing scenarios when considering (solid lines) or not (dashed lines) fire –
666 vegetation feedbacks. See figure 2 for color meaning. Only HFR zones where initial
667 yearly proportion of harvested area was above 0.05% are shown.

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672 Figure 1

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Figure 2

Figure 3

Figure 4

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697 **Supplementary material S1** – Proposed corrections (in red) for negative feedbacks of future burning rates previously projected by
 698 Bergeron et al. (2004, 2006) and Flannigan et al. (2005). Previously published burning rates above 1% per year are in **bold**.

699 **a) Bergeron et al. (2004)**

Site No	Study area	Current burn rate (%)	2xCO2 burn rate	3xCO2 burn rate	2xCO2 burn rate Adjusted	3xCO2 burn rate Adjusted	2 x CO2 / current ratio	3 x CO2 / current ratio	2 x CO2 / current ratio Adjusted	3 x CO2 / current ratio Adjusted
1	Jasper Park W. Central	0.027	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
2	Alberta	0.019	0.024	0.028	0.025	0.028	1.253	1.418	1.278	1.446
3	Glacier Park	0.063	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
4	Yoho Park	0.049	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
5	Kootenay	0.065	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6	Kananaskis Wood Buffalo	0.050	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7	Park Northern	0.660	0.812	0.773	0.764	0.729	1.230	1.170	1.157	1.105
8	Alberta	0.222	0.351	0.427	0.346	0.417	1.580	1.920	1.557	1.877
9	Rutledge Park	0.903	1.210	1.734	1.095	1.494	1.340	1.920	1.212	1.654
10	Prince Albert Northern	0.470	0.451	0.484	0.440	0.470	0.960	1.030	0.936	1.001
11	Ontario	0.462	1.998	2.658	1.681	2.112	4.330	5.760	3.643	4.577
12	LAMF	0.046	0.060	0.064	0.061	0.065	1.320	1.410	1.342	1.432
13	Western	0.032	0.033	0.048	0.033	0.049	1.019	1.500	1.038	1.526

	Quebec									
	Algonquin									
14	Park	0.007	0.007	0.006	0.007	0.006	0.985	0.836	1.007	0.854
	Abitibi east									
15	Quebec	0.032	0.026	0.020	0.027	0.021	0.811	0.632	0.827	0.645
	Northern									
16	boreal Quebec	0.261	0.415	0.869	0.406	0.812	1.590	3.330	1.557	3.114
17	Central Quebec	0.111	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	Southeastern									
18	Labrador	0.038	0.057	0.068	0.057	0.069	1.491	1.799	1.516	1.827

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702 **b) Bergeron et al. (2006)**

Site No.	Study Area	Current burn rate (%)	2 x CO ₂ burn rate (%)	3 x CO ₂ burn rate (%)	2 x CO ₂ burn rate (%) Adjusted	3 x CO ₂ burn rate (%) Adjusted
1	Abitibi northwest	0.239	0.394	1.262	0.387	1.136
2	Abitibi southwest	0.258	0.282	0.302	0.280	0.299
3	Abitibi east	0.239	0.394	1.262	0.387	1.136
4	Abitibi southeast	0.048	0.064	0.072	0.065	0.073
5	Temiscamingue north	0.048	0.064	0.072	0.065	0.073
6	Temiscamingue south	0.036	0.03	0.02	0.031	0.020
7	Waswanipi	0.239	0.394	1.262	0.387	1.136
8	Central Quebec	0.258	0.282	0.302	0.280	0.299
9	North Shore	0.155	0.179	0.384	0.180	0.377
10	Gaspesia	0.205	0.306	0.503	0.303	0.488

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705 c) Flannigan et al. (2005)

Modified ecozones (No.)	Area with fuel (ha)*	AAB** 3 x CO2 Canadian§ (ha)	AAB 3 x CO2 Hadley§ (ha)	AAB 3 x CO2 Canadian (%)	AAB 3 x CO2 Hadley (%)	AAB 3 x CO2 Canadian (%) Adjusted	AAB 3 x CO2 Hadley (%) Adjusted
Boreal Cordillera (12)	33 924 875	36 000	353 000	0.11	1.04	0.11	0.96
Boreal Plains (9)	63 233 813	237 000	751 000	0.37	1.19	0.37	1.08
Eastern Boreal Shield (62)	90 664 044	252 000	266 000	0.28	0.29	0.28	0.29
Taiga Shield + Hudson Plains (5)	116 821 963	821 000	817 000	0.70	0.70	0.67	0.67
Montane Cordillera (14)	38 385 800	0	51 000	0.00	0.13	0.00	0.13
Taiga Cordillera (11)	16 804 213	88 000	120 000	0.52	0.71	0.51	0.68
Taiga Plains (4)	48 578 650	508 000	574 000	1.05	1.18	0.96	1.07
Western Boreal Shield (61)	64 206 406	824 000	947 000	1.28	1.47	1.15	1.30

706 *As Flannigan et al. (2005) reported the total annual area burned in thousands of hectares, we had to first estimate the total area with fuel within each
707 ecozone in order to estimate the proportion of annual area burned (%) which comply with our model. Area of fuel was estimated using EOSD imagery
708 (Beaubien et al. 1999).

709 **AAB: Annual area burned

710 § The Canadian model used was the First Generation Coupled GCM (CGCM1) for the time period 2080–2100 which roughly corresponds to an equivalent 3
711 × CO2 scenario when including both greenhouse gas and sulphate aerosol forcing. The Hadley model, HadCM3GGa1, contained only greenhouse gas
712 forcing and used 2080–2099 as its equivalent 3 × CO2 scenario.

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Supplementary material S2 – a) Proportion (%) of the HFR zone that is “harvestable”, i.e., where total volume is at least 100 m³.ha⁻¹ and b) mean stand age according to different anthropogenic forcing scenarios when considering (solid lines) or not (dashed lines) fire – vegetation feedbacks. See figure 2 for color meaning.

a)

735

b)

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738 **Supplementary Material S3** – Animated figures showing the ratio between year
739 t and year = 2000 for the fraction of harvestable stands (above 100 m³.ha⁻¹) or for
740 the total harvested volume in each FMU as simulated under baseline, RCP 2.6,
741 RCP 4.5 and RCP 8.5. Results are shown for simulations where biotic feedbacks
742 were included.

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