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ARTICLE

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

Yan Boulanger, Martin Girardin, Pierre Y. Bernier, Sylvie Gauthier, André Beaudoin, and Luc Guindon

Abstract: 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 a 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 nonexistent when potential burning rates are below 1%, whereas realized burning rates would be lowered by more than a 0.5 percentage point when potential burning rates exceed 2.5%. Including fire–vegetation feedbacks had virtually no impact on total volume harvested. As fire burns more old-growth coniferous stands, slightly negative impacts were projected on conifer harvested almost everywhere. These results underline the need to incorporate fire–vegetation feedbacks when projecting future burn rates.

Key words: fire–vegetation feedbacks, boreal forest, forest fires, climate change, timber supply, Canada.

Résumé : On anticipe une augmentation de l'activité des feux de forêt au Canada à cause du changement climatique mais on ne tient généralement pas compte des rétroactions de la végétation. À l'aide de nouvelles informations concernant la sélectivité et l'évitement du feu en fonction de l'âge et de la composition des peuplements, nous avons utilisé des modèles de simulation simples qui tiennent compte des changements dans les matrices d'âge régional engendrés par le feu et la coupe pour prévoir les futurs taux de brûlage. Nous avons également prévu la vulnérabilité régionale estimée de l'approvisionnement en bois face aux feux de forêt en tenant compte de ces nouveaux taux de brûlage. L'inclusion de rétroactions reliées à l'âge devrait avoir un impact important sur l'augmentation prévue des taux de brûlage, surtout dans les zones soumises à un forçage climatique agressif où le feu est très actif. Les taux de brûlage devraient augmenter encore mais devraient être 50 % moins élevés en 2100 que s'ils étaient anticipés sans rétroaction biologique dans certaines zones. Les rétroactions négatives devraient être pratiquement inexistantes lorsque les taux de brûlage potentiels sont inférieurs à 1 %, tandis que les taux de brûlage effectifs devraient diminuer de plus de 0,5 point de pourcentage lorsque les taux potentiels de brûlage dépassent 2,5 %. L'inclusion des rétroactions entre le feu et la végétation n'a eu pratiquement aucun impact sur le volume total récolté. À mesure que le feu brûle davantage de vieux peuplements de conifères, des impacts légèrement négatifs sur les conifères récoltés sont prévus presque partout. Ces résultats font ressortir la nécessité d'incorporer les rétroactions entre le feu et la végétation lorsqu'on prévoit les taux futurs de brûlage. [Traduit par la Rédaction]

Mots-clés : rétroactions entre le feu et la végétation, forêt boréale, feux de forêt, changement climatique, approvisionnement en bois, 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. 2009*a*; 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 (Balshi et al. 2009*b*), 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).

Besides 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; Girardin et al. 2013*a*; Terrier et al. 2013; Héon et al. 2014). 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

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Y. Boulanger, M. Girardin, P.Y. Bernier, S. Gauthier, A. Beaudoin, and L. Guindon. Natural Resources Canada, Canadian Forest Service, Laurentian Forestry Centre, 1055 du P.E.P.S., P.O. Box 10380, Stn. Sainte-Foy, Québec, QC G1V 4C7, Canada.

Corresponding author: Yan Boulanger (email: yan.boulanger@canada.ca).

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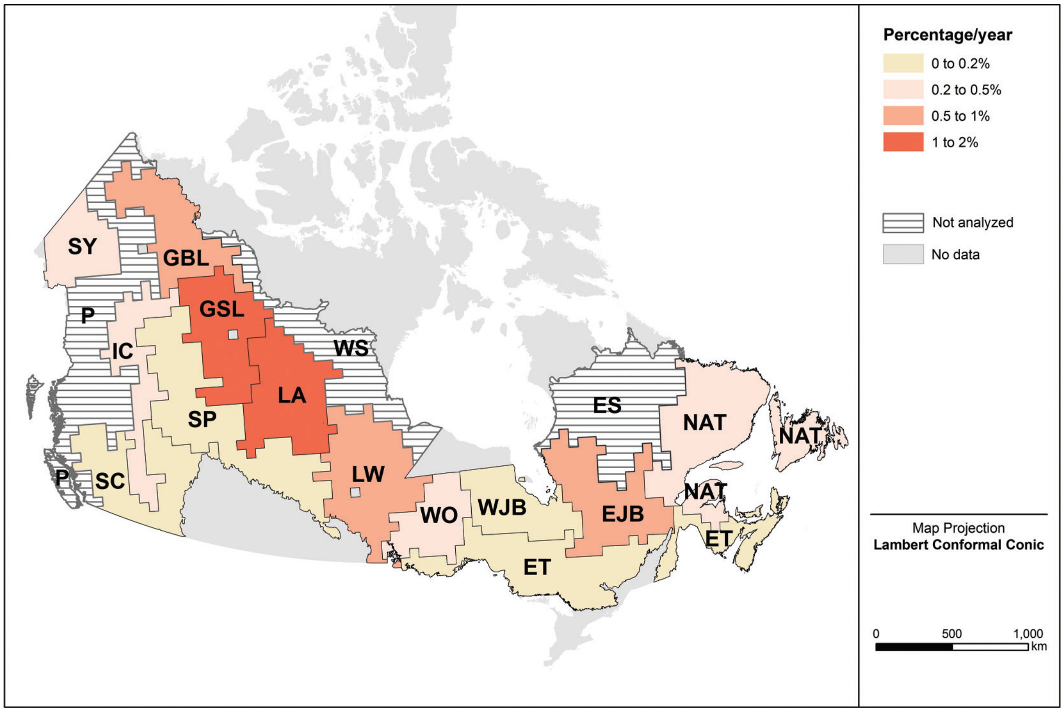
Fig. 1. Homogeneous fire regime (HFR) zones as delineated in Boulanger et al. (2014) along with their associated burn rates for the 1959–1999 period. EJB, Eastern James Bay; ET, Eastern Temperate; GBL, Great Bear Lake; GSL, Great Slave Lake; IC, Interior Cordillera; LA, Lake Athabasca; LW, Lake Winnipeg; NAT, North Atlantic; SC, Southern Cordillera; SP, Southern Prairies; SY, Southwestern Yukon; WJB, Western James Bay; WO, Western Ontario. Eastern Subarctic (ES), Pacific (P), and Western Subarctic (WS) HFR zones were not included in the analyses. [Color online.]



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. 2017) 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. 2009*a*; 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%·year^{-1} (Boychuk 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 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. 2009*a*; Girardin et al. 2013*a*), we propose that it be indirectly accounted for by applying the correction factor presented in the caption to 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) 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 (Fig. 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 come 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 rate 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. The RCP 2.6 scenario represents a situation where radiative forcing peaks at $\sim$3 W·m^{-2} before 2100 and then declines to reach 2.6 W·m^{-2} by 2100. In the RCP 4.5 scenario, radiative forcing is assumed to stabilize at 4.5 W·m^{-2} after 2100 without an "overshoot" pathway. Conversely, in the RCP 8.5 scenario, the forcing reaches 8.5 W·m^{-2} in 2100 and continues to increase for some time afterwards. The appropriate outputs from the Canadian Earth System Model version 2 (CanESM2) were downloaded from the World Climate Research Program (WCRP) Climate Model Intercomparison Project Phase 5 (CMIP5) archive.

Using the stochasticity functionality of BioSIM (Régnière et al. 2014), we simulated 3000 daily time series lasting 1 year for all combinations of HFR zone × climate scenario × time period. From these daily time series, we then derived the standard components of the Canadian Forest Fire Weather Index System (CFWI) (Van Wagner 1987) as well as other temperature- and precipitation-related variables on a monthly basis. With these variables, regional climate-driven burn rates projections ($BR_{regclimt}$) were computed for all 3000 yearly weather data sets for each HFR zone × climate scenario × time period using HFR zone-specific MARS models developed by Boulanger et al. (2014).

Fire selection ratios

Estimates of fire selection ratios for classes of forest age and composition were produced by Bernier et al. (2016). Briefly, maps of 2001 forest properties of Canada's managed forests (Beaudoin et al. 2014) and yearly 2001–2011 Canada-wide maps of fire and harvest (Guindon et al. 2014), both on the 250 m MODIS grid, were used to obtain 2001 estimates of age and composition of all pixels with a forest cover of ≥80% (n = 76 678 906) to 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 (0–29 years), mature (30–89 years), and old (90+ years) 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 (Krawchuk and Cumming 2011; Héon et al. 2014). 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 zones 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.

Table 1. Fire selectivity ratios ($cover_m$ in eq. 3) as a function of forest age and composition classes used in simulations considering biotic feedbacks to fire activity.

Composition class (% of conifer)	Age class: Young (0–30 years)	Mature (30–90 years)	Old (>90 years)
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

Note: A ratio of 1 indicated that regional burn rate for a given each class was proportional to its regional availability, while values above 1 indicated fire selection and values below 1 indicated fire avoidance.

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 250 m 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.

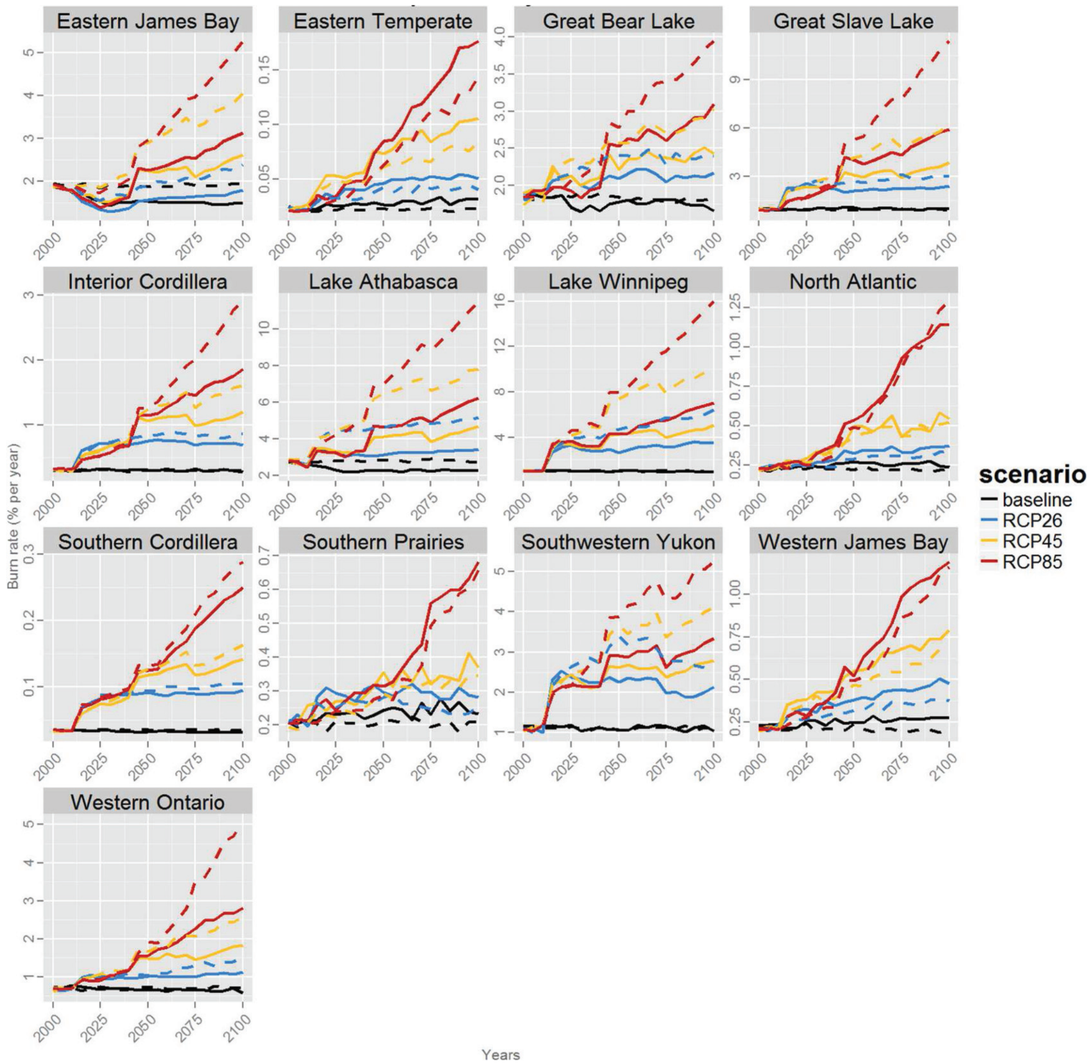
Calculation of pixel-level volumes

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 data set as in our study, were used to estimate changes in total volume and in volume harvested in each HFR zone. These curves were formulated as

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

where V is the pixel-level volume (m^3·ha^{-1}) based on all tree's woody and foliar components, A is stand age (years), T is historical (1970–2000) mean annual air temperature of the pixel (°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):

Fig. 2. Mean burn rates according to different anthropogenic forcing scenarios when considering (solid lines) or not (broken lines) fire–vegetation feedbacks within each HFR zone. Black, baseline climate; blue, RCP 2.6; yellow, RCP 4.5; red, RCP 8.5. Note that the scale of the *y*-axis is variable. [Color online.]



$$V = \exp[\log(V)] \times C_D \tag{2}$$

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 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 combination, we ran nonspatially explicit simulations including both fire and harvesting as stand-replacing disturbances. Two sets of simulations 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, 60 simulations were run for 100 years using a 5 year time step starting in 2000. Initial pixel-level age, volume, and coniferous/hardwood fractions 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 the 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 12 forest cover classes (Table 1) at time $t = 0$ (BaselineMean*P*), thereby transforming the probabilities into a set of normalised selection ratios (Manly et al. 1993).

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

$$Prob_{burn} = BR_{regclimt} \times cover_m \times linear_{mod} \times t_{step}/BaselineMeanP \tag{3}$$

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 year period. The $cover_m$ was set to 1 for all forest cover classes in simulations 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 the 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) data set in each management area. Only pixels that reached commercial maturity, here arbitrarily defined as $V = 100\ m^3{\cdot}ha^{-1}$, 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 eqs. 1 and 2.

The burning rate was computed for each HFR zone at each time step for each simulation. 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 year period using the following nonlinear least square model:

$$BR_{rea} = v \times BR_{pot}/(k + BR_{pot}) \tag{4}$$

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 60 burning rates coming from the same simulation data set of a given HFR zone were averaged 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 were important in the great majority of HFR zones but were 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 rate increases as compared to projections that did not incorporate such feedbacks (Fig. 2). These feedbacks lowered projected burn rate increases by as much as half the potential burn rate, representing a difference of 2–8 percentage points in annual area burned under the RCP 8.5 climate scenario. Furthermore, high fire activity as simulated under baseline conditions was sufficient in the 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 of these zones under RCP 2.6 and RCP 4.5 (Fig. 2). Negative feedbacks were minimal regardless of climate forcing in the 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 a 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 anthro-

Fig. 3. (*a*) Realized burning rates as a function of potential burning rates. Recall that these rates were averaged over the 100 year period. See Fig. 2 for color meaning. The 1:1 line is shown broken. The solid black line illustrates the nonlinear fitted model where $BR_{rea} = 9.466 \times BR_{pot}/(9.254 + BR_{pot})$; $n = 52$, pseudo-$R^2 = 0.992$. (*b*) Importance of the negative feedback (% of burning rate) as a function of potential burning rates. This basically represents the 1:1 line minus the solid black line in Fig. 3*a*. [Color online.]

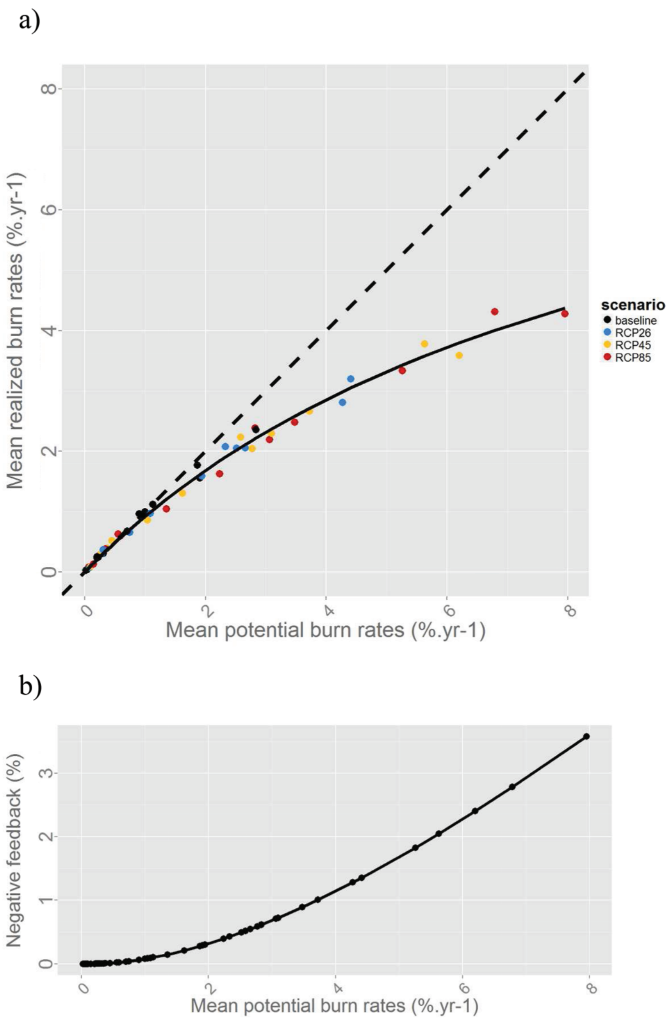
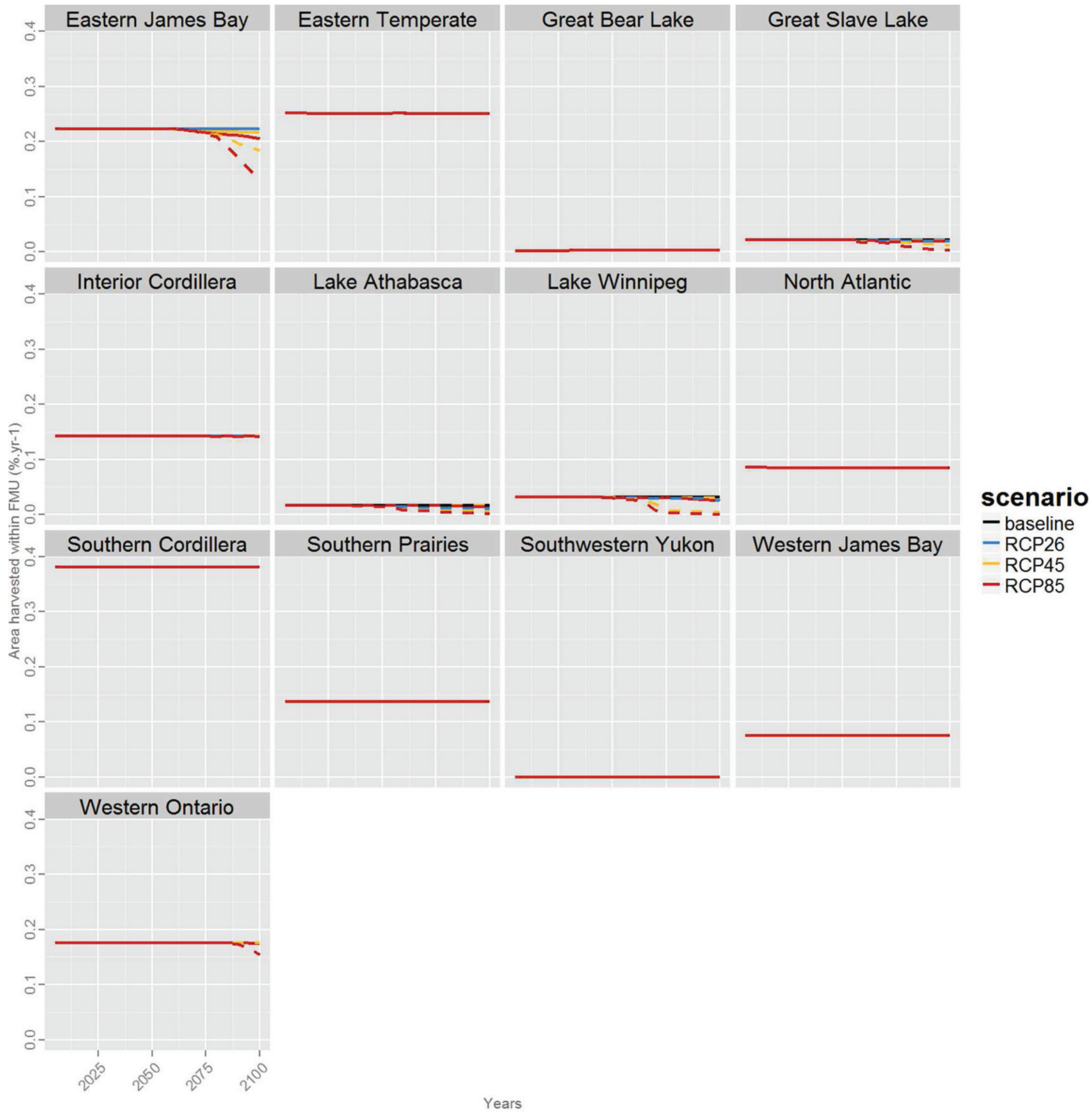


Fig. 4. Proportion of the HFR zone total area harvested per year according to different anthropogenic forcing scenarios when considering (solid lines) or not (broken lines) fire–vegetation feedbacks. See Fig. 2 for color meaning. [Color online.]



pogenic 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 approximately 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

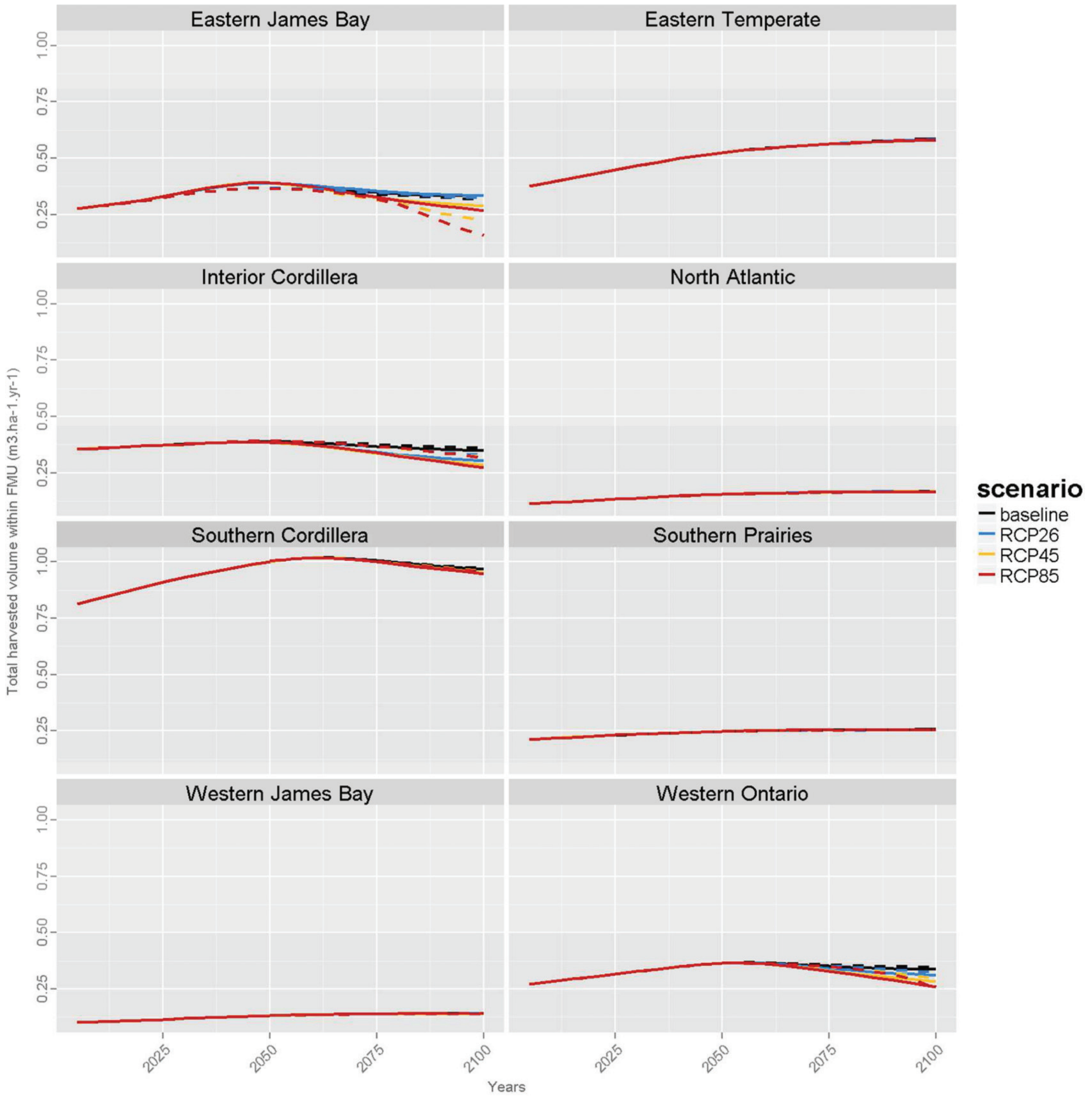
Accounting for biotic feedbacks on projected future burn rates had virtually no impact on harvested area and total volume harvested (Figs. 4 and 5). Exceptions were for Eastern James Bay where volume harvested would be approximately 60% higher by 2100 under RCP 8.5 when considering biotic feedbacks. However, the inclusion of biotic feedbacks 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 (Fig. 6). The proportion of potential harvested conifer volume was lower by 5%–10% under RCP 8.5 as well as, to a lesser degree, under milder forcing scenarios. Virtually no impacts were simulated elsewhere (Fig. 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 on climate forcing. The potential harvestable volume was projected to increase regardless of forcing scenario and most markedly in Eastern Temperate and Southern Cordillera (Fig. 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

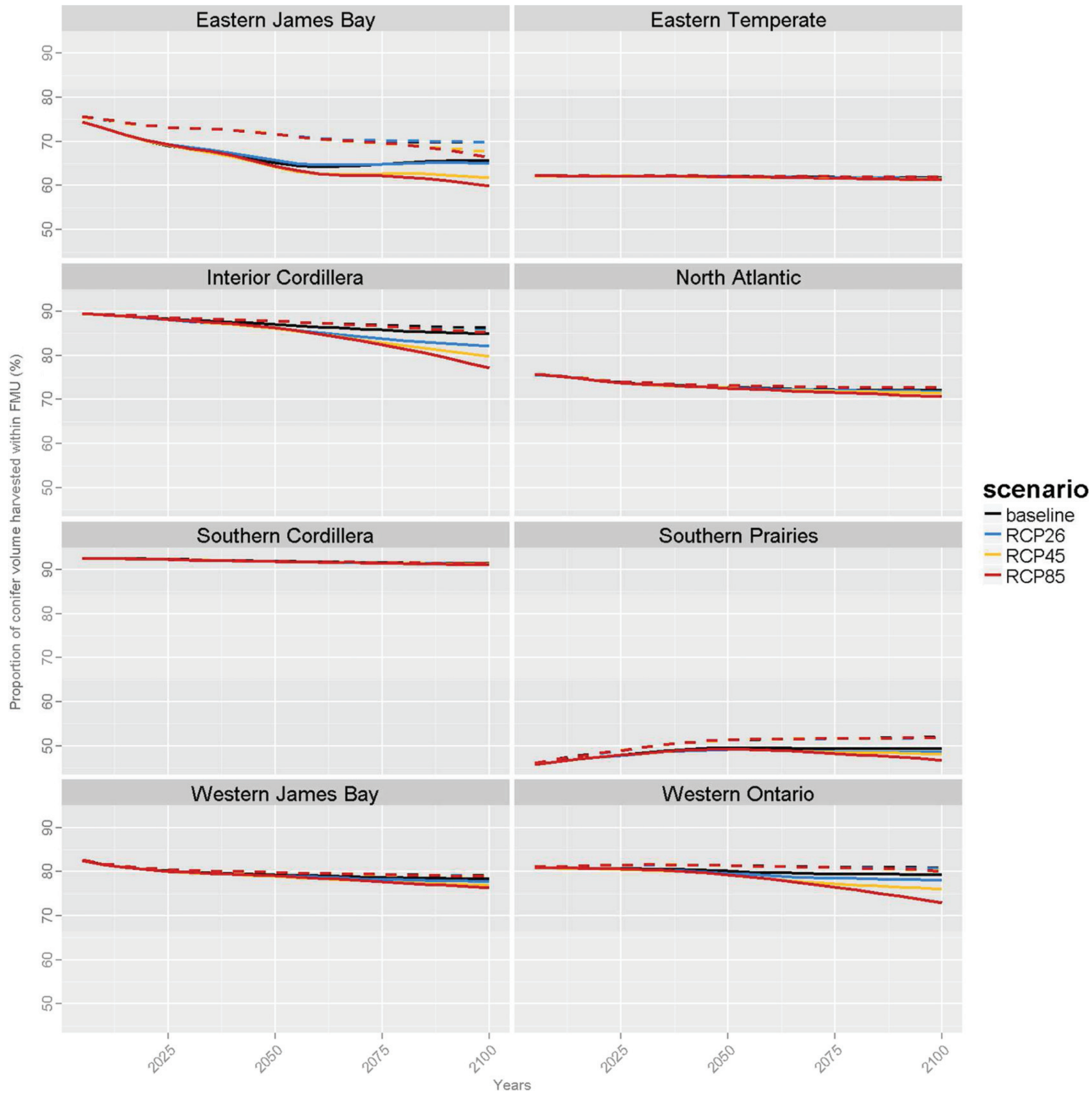
Fig. 5. Same as Fig. 4 but for total harvested volume. Only HFR zones where initial yearly proportion of harvested area was above 0.05% are shown. [Color online.]



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 modeled 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 (Balshi et al. 2009*a*; Boulanger et al. 2014). 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. 2013*a*). 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. 2013*b*). In other zones and under mild climate forcing, negative feedbacks were rather small to nonexistent: 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 nonexistent 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.

Fig. 6. Proportion of conifer volume harvested according to different anthropogenic forcing scenarios when considering (solid lines) or not (broken lines) fire–vegetation feedbacks. See Fig. 2 for color meaning. Only HFR zones where initial yearly proportion of harvested area was above 0.05% are shown. [Color online.]



2014; suppl material S1[1]). However, realized burning rates would be lowered by more than a 0.5 percentage point when potential burning rates exceed 2.5% over a 100 year 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. 2009*a*; Girardin et al. 2013*a*), we propose that it be indirectly accounted for by applying the correction factor presented in the caption to 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 nonaccounting for age feedbacks (supplementary material S1[1]).

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 (supplementary material S2[1]). 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 $m^3 \cdot ha^{-1}$) thus tends to decrease, while that of deciduous stands tends to increase, giving as a result an 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

[1]Supplementary data are available with the article through the journal Web site at http://nrcresearchpress.com/doi/suppl/10.1139/cjfr-2016-0445.

preferences in simulations could thus reveal a much worse portrait in future potential harvestable volume.

Our results concur with those of Gauthier et al. (2015) in which forest regions in interior British Columbia, northwestern Ontario, and northcentral Quebec (see supplementary material S3[1]) 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 significant. When compared to the situation prevailing in 2000, several forest management units located in these zones would experience a decrease in the proportion of harvestable stands (i.e., stands with merchantable volume greater than 100 $m^3 \cdot ha^{-1}$) and hence total volume harvested. Our analyses also suggest that, even with the current baseline fire and harvesting conditions, timber availability in some of these forest management units may decrease within the next decades (supplementary material S3[1]), with potential impacts on the supply value chain even without considering increased climate forcing (Irland et al. 2001; Williamson et al. 2009; Gauthier et al. 2015). Prior work has suggested that salvage logging can mitigate but never eliminate fire impacts on timber availability (Leduc et al. 2015).

There are numerous limitations to our analyses. Simplistic assumptions about fire selectivity that do not incorporate complex fire initiation and fire spread functions as computed by process-based models (e.g., Burn-P3, FIREBGCv2: Parisien et al. 2005; Keane et al. 2011) were used in our model. As such, fine-scale topographic and weather conditions are not taken into account in burned area projections. As mentioned above, the extent to which our analyses were performed impedes the use of these complex process-based fire models. Also, we have not incorporated the extent to which successional changes in forest composition, e.g., from conifers to hardwood species, could act as an additional feedback mechanism in response to increased fire activity. The reason for this exclusion is simply uncertainties regarding post-fire succession rules as affected by climate change at the scale of Canada. Possible conversion of late-succession coniferous stands to mixed- or hardwood stands could further strengthen negative feedbacks considering higher foliar moisture loading and lower flammability for broadleaf species (Päätalo 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 (McKenney et al. 2011; Boulanger et al. 2016), could also further hinder fire activity (Carcaillet et al. 2010; Girardin et al. 2013*a*; 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 (Héon et al. 2014; Erni et al. 2017) 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 evolve different fuel load and type, notably fine fuel abundance, making recently harvested stands 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.

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

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