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Mitigating risks of future wildfires by management of the forest composition: an analysis of the offsetting potential through boreal Canada

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Abstract

Wildfire activity is projected to increase through upcoming decades in boreal Canada due to climatic changes. Amongst the proposed strategies to offset the climate-driven fire risk is the introduction of broadleaf species into dense-coniferous landscapes so as to decrease the intensity and rate of spread of future wildfires. Here we examine this offsetting potential through boreal Canada by searching for optimal conifer to broadleaf conversion rates that would stabilize the burn rate metric, and an upper bound for the maximum potential effect. We developed an empirical model relating regional burn rates to mean annual fire weather conditions and tree genus proportions, and applied it to regional climate and forest composition change scenarios covering the interval from 1971 to 2100. Results suggested that many areas in the southern and northern boreal regions will record either a constant or a decreasing burn rate and, therefore, will not require a change of forest composition. Besides, a conversion rate of 0.1% to 0.2% year^{-1} starting in year 2020 was sufficient to maintain burn rates constant across much of the southern boreal forest. In northern forests, however, higher conversion rates were required to meet the fire objectives (0.3 to 0.4% year^{-1}). This mitigation option will be difficult to implement over northern forests given the size of areas involved. Nonetheless the estimated conversion rate for much of the southern boreal forest is attainable, considering that harvesting and industrialization during recent decades have already contributed to similar changes of the proportion of broadleaf species in boreal landscapes.

Keywords: burn rates, climate change, fuel management, broadleaf species, boreal forest, Multivariate Adaptative Regression Splines

Introduction

The 20th century was a pivotal period at high northern latitudes as it marked the onset of a rapid climate warming associated with major anthropogenic changes in global atmospheric composition (PAGES 2k Consortium 2013). Prominent effects include more persistent weather patterns with an increased probability of long-lasting extreme weather events, such as summer heat waves and droughts (Francis and Vavrus 2012). Already, these changes have impacted boreal forests. Indeed, wildfire activity has been increasing in many Canadian regions since fires have been recorded, in association with climate warming (e.g. Gillett et al. 2004). The phenomenon of enhanced fire activity is projected to continue throughout this century for almost all of boreal Canada, as atmospheric greenhouse gases attain unprecedented levels (Wotton et al. 2010; Boulanger et al. 2014). Anticipated consequences from the increasing fire activity are the alterations of wildlife habitats, the increase in the carbon emissions, increasing threats for population safety, and more economic damage to the forest sector, which may include the reduction of commercial products and timber supplies (de Groot et al. 2013; Rupp et al. 2006; Harden et al. 2000; Gauthier et al. 2015). There is a need for preparedness and for exploring forest management strategies that aim at reducing fire risks brought about with climate change.

Amongst adaptation options covering the strategic and silvicultural aspects of forest management, manipulative vegetation treatments in which boreal tree compositions are shifted toward an increasing broadleaf component are increasingly being considered (Hirsch et al. 2004; Krawchuk and Cumming 2011; Terrier et al. 2013). Broadleaf stands are characterized by higher leaf moisture (Johnson 1992) and lower flammability and rate of fire ignition than coniferous stands (Van Wagner 1983). Their introduction into coniferous stands as strategic barriers, or increasing abundance in landscapes, could decrease the intensity (i.e. energy output) and rate of spread of fires, improve suppression effectiveness, and reduce fire impacts (Amiro et al. 2001; Hirsch et al. 2004). The realization of this "biotic effect" (i.e. the offsetting effect of fuel composition; Krawchuk and Cumming 2011) on wildfire activity has recently been brought to the attention with the publication of studies demonstrating that past fire risks during post-glacial warm episodes were offset by a higher broadleaf component in landscapes (Girardin et al. 2013; Kelly et al. 2013; Brown and Giesecke 2014). That being said, the practical implementation of this fire management option has challenges. For instance, the limit of the biotic effect for a given magnitude of climate change needs to be determined: under excessive drought, the biotic effect might not be strong enough to attain the fire reduction objectives (e.g. Krawchuk and Cumming 2011; Alexander 2010). Additionally, given the complexity of climate and vegetation settings across Canada, the biotic effect is likely to depend on a rate of forest conversion that is region-specific: a given rate of forest conversion may be sufficient to offset climate change impacts on fire at some locations, and not at others. Modelling experiments may provide significant insights into these issues.

Here we examine the potential biotic effect on future fire activity with the aid of model experiments applied through boreal Canada. We focus on the analysis of the burn rate metric, that is, the proportion of burned areas per year (% yr^{-1}) in a given region (e.g. the typical scale at which the burn rate metric is applied in forest management planning, Gauthier et al. 2009). Specifically, our work aims at searching for the optimal forest conversion rate to keep the regional burn rate metric constant, and an upper bound for the maximum potential biotic effect, under given climate change scenarios and regions. To achieve this, the following tasks were undertaken: I) the Canadian boreal forest was divided into fire bioclimatic regions on the basis of fire climatology and vegetation datasets, II) a burn rate model was parameterized using the segmented regression technique, and III) future burn rates were projected over 1971–2100 using simulation outputs from a regional climate model and forest composition change scenarios.

Material and Methods

Fire data

Two fire datasets were used in this study. First, yearly national area burned maps constructed from MODIS summer and winter composite imagery at 250 × 250 m (6.25 ha) spatial resolution, and covering 2001 to 2011, were used to develop the burn rate model. These maps are the results of classification using data mining algorithms such as decision and regression trees. They represent a uniform and standardized dataset across Canada that captures both fire and timber harvest events on an operational basis (Guindon et al. 2014). Accuracy of burned area detection is hindered in some regions by persistent haze or cloud cover, insect outbreaks and salvage logging. Ecozones with the least amount of topography (Boreal Shield, Boreal Plain, Hudson Plain and Taiga Shield; Fig. S1) had higher fire detection accuracies in a data validation procedure (Guindon et al. 2014) and were retained for this study (the Boreal and Montane Cordillera were excluded). Secondly, the Canadian National Fire Database (CNFDB) was used for verifying the predictive skills of the burn rate model. The CNFDB is a collection of forest fire data, covering the 1950s to the present, provided by Canadian fire management agencies (provinces, territories, and Parks Canada; Canadian Committee on Forest Management, 2012).

Climate data and fire weather indices

We used the Canadian Fire Weather Index (FWI) System (Van Wagner 1987) to estimate fuel moisture and generate a series of relative fire behavior indices based on weather observations and simulations (Supplementary Information). The moisture indices are the fine fuel moisture code (FFMC), duff moisture code (DMC), and drought code (DC). Fire behavior indices include the numerical rating of fire spread (initial spread index, ISI), the fuel available for combustion (build-up index, BUI), and an approximation of the difficulty of controlling fires (daily severity rating, DSR). Winter precipitation was included in the algorithms of the FWI, so fire behavior indices and length of the fire season (FS) also depend on snow accumulation (Terrier et al. 2013). All indices are unitless, with the zero value indicating low fire risks and high values indicating high fire risks.

A set of 3000 locations was randomly generated across the boreal forest, and weather data (maximum daily temperature, precipitation, wind, and relative humidity) were obtained for each location using the BioSIM software (Régnière and Bolstad 1994). As part of the procedure, daily data were interpolated from the four closest weather stations, adjusted for elevation and location differentials with regional gradients, and averaged using a $1/d^2$ weight, where d is distance. Data for the 1971-2011 period was interpolated from Environment Canada's historical climate database (Environment Canada 2013). Data for 2031–2060 and 2071–2100 were obtained from bias-corrected gridded simulation outputs of the Canadian Fourth Generation Regional Climate Model (CanRCM4); de Elia and Côté, 2010; Supplementary Information). Simulations were performed using the 'representative concentration pathways' RCP 4.5 and RCP 8.5 scenarios used in the IPCC Fifth Assessment Report (Van Vuuren 2011). In the RCP 4.5 scenario, total radiative forcing is stabilized shortly after 2100 at 4.5 W/m^2 (~ 650 ppm CO_2 eq.). The RCP 8.5 scenario expresses a rising radiative forcing pathway leading to 8.5 W/m^2 (~1300 ppm CO_2 eq.) by 2100.

Forest composition

Published maps of tree species relative proportions (Beaudoin et al. 2014) were used to estimate forest composition at each of our locations. Briefly, the maps describe 127 forests attributes (land attribute, stand structure, species proportions, etc.) estimated using the '*k* Nearest Neighbours' (*k*NN) method from photo-plot observations of Canada's National Forest Inventory and explicative variables (MODIS reflectance, climatic, topographic variables and permanent cover class). Maps were built at 250 × 250 m resolution for the 2001 base year. Species' relative proportions (% coverage) were extracted to each location and summed at these genus levels, which are dominant ones in Canada's boreal forest: *Balsam* spp., *Betula* spp., *Picea* spp., *Pinus* spp. and *Populus* spp. Locations without tree information were discarded. We obtained a total of 2714 random locations for our analyses (Fig. S1).

Statistical Analysis

Spatial clustering and fire bioclimatic region delimitation

Baseline reference conditions for the different fire weather indices were computed at each of our locations from the averages of the daily quantities over 2001–2010. Space-constrained agglomerative clustering (Legendre and Legendre 2012) was then applied to these indices, and to forest proportions data, to divide the study area into homogeneous fire weather and forest composition zones, respectively (Supplementary Information). The clustering analysis was performed with the "const.clust" package (Legendre 2011) included in the R freeware (R Development Core Team 2010). The number of clusters that minimized the cross-validated residual error (CVRE) was retained. Prior to clustering, we used Pearson correlation analysis to exclude collinear fire weather variables from the data matrix. Accordingly, we limited our analysis to the DC, DSR, and FS variables. Fire bioclimatic regions were created from overlapping of the fire weather and vegetation zones.

Parameterization of the burn rate model

The effect of vegetation on fires was first evaluated by comparing forest composition in burned and non-burned areas in each fire bioclimatic region. Forest composition datasets were aggregated from the 250 × 250 m values to the level of regional burned and non-burned areas through spatial averaging with ESRI® ArcGIS 10.1. A two-sample Student's *t*-test (one-sided) was applied to test for differences in coniferous and broadleaf genus proportions between burned and non-burned areas. The national area burned maps were used for these analyses.

The burn rate model was parameterized using Multivariate Adaptative Regression Splines (MARS) (Friedman 1991). MARS is a non-parametric spline regression approach that models non-linear relationships between a response variable and explanatory variables. The main principle is the division of the space of explanatory variables into regions. A set of linear regressions, named basis functions, are then fitted for each region to describe the relationships between the response and explanatory variables (segmented piecewise regression). The MARS model was formulated as:

$$(1)\ BR_j = \sum(c_1BF_1 \times FW_j + c_2BF_2 \times \rho),$$

where BR_j is the burn rate (% $year^{-1}$), FW are the fire weather indices, and ρ are the genus proportions (%). Finally, c_1 and c_2 correspond to constants, while BF_1 and BF_2 are basis functions for non-linear interactions. All variables were averaged for the decade *j* at the level of fire bioclimatic regions. Decadal averages were used instead of annual or long-term averages to avoid having too many zeros in the response matrix. Herein, the period used for calibration of the MARS model was 2001–2010; the national area burned maps were used. The MARS model was calibrated using the Salford System Software (Salford Systems 2013). Model selection was done using the generalized cross-validation (GCV) criteria. The degrees of freedom penalty was set using a *v*-fold cross validation analyses ($v = 10$). Other software parameters were set by default.

Two independent verifications were conducted to ensure that the MARS model had adequate predictive skills. First, the burn rate model was applied to FW indices covering the year 2011, and the predicted burn rates were correlated to corresponding values derived from the 2011 national area burned map. Second, the burn rate model was applied to 1971–2000 averages of the FW indices, and the predicted burn rates were correlated to corresponding values derived from the CNFDB. In both cases, the goodness-of-fit was assessed by the Spearman rank correlation between observed and predicted data. Note that genus proportions were herein kept constant: we thus assume that vegetation composition did not change within the calibration-to-verification interval. This assumption may cause a portion of the variance in verification data to remain unexplained if the fraction of flammable vegetation changed from one period to another.

Projections of future burn rates and impact of changes in the composition of forests

The burn rate model was applied to the 1971–2000 decadal averages of the FW indices and to the genus proportions across the 2714 locations. The procedure was repeated after substituting the historical FW indices by the future ones (2031–2060 and 2071–2100 horizons) obtained from the CanRCM4 simulations. Decadal results were averaged to obtain burn rates for each 30-year period.

The biotic effect was examined by incrementing the percent cover of broadleaf species in eq. 1 through a conversion rate λ (expressed as % yr^{-1}) starting in 2020. Proportions of coniferous species were lowered accordingly. Two iterative approaches contributed to locally defining this λ metric. First, burn rates were computed iteratively for each location after incrementing λ until the difference between the projected burn rate (2071–2100) and the baseline burn rate (1971–2000) equaled zero. In the second approach, burn rates were computed iteratively for each location l by incrementing λ until saturation of the biotic effect. This maximum biotic effect was determined using the β value of eq. 2:

$$(2)\ \beta_l = \lambda_l - \left(BR_{status-quo_l} - BR_{treatment_l}\right),$$

where λ is the conversion rate of forest composition and $BR_{status-quo}$ and $BR_{treatment}$ are the burn rates projected by status-quo and forest composition change scenarios, respectively, at location *l*. The maximum biotic effect is reached when the difference between $BR_{status-quo}$ and $BR_{treatment}$ is no longer greater than λ (i.e. $\beta \geq 0$). Our analysis thus provides an optimal λ value for keeping the burn rate metric constant, and an upper bound of λ for the maximum potential biotic effect, for each location *l*.

Results

Analysis of baseline conditions

Constrained spatial analysis led to the delimitation of 35 fire bioclimatic regions (Fig. 1). Therein burn rates varied from <0.1% yr^{-1} to a maximum of 1.9% yr^{-1} (mean of 2001–2010; Fig. 1c). The highest burn rates were observed in the FW zones I, II, IV, and XI where high fire weather indices prevailed along with relatively low proportions of broadleaf species (Fig. 1; Tables S1 and S2). Some northern coniferous dominated regions showed low burn rates despite having high fire weather indices (FW zones X and VII), likely owing to short fire seasons that characterizes these high latitudes.

Globally, the coverage of *Abies* spp., *Picea* spp. and *Pinus* spp. was significantly greater (Student's *t*-tests, $P < 0.001$) in burned areas in comparison with non-burned areas (Fig. 2a). The reverse holds true in regard to the proportions of broadleaf species: the coverage of *Betula* spp. and *Populus* spp. was significantly lower in burned areas in comparison with non-burned areas (Student's *t*-tests, $P = 0.05$) (Fig. 2b), except in northernmost regions where the proportions of these species were low (Fig. 1b and Table S2).

Predictive models of burn rates

The burn rate metric was regressed against decadal averages of FW indices and genus proportions using MARS. The model explained 58% of the deviation in the burn rates (GCV $R^2 = 0.29$; Fig. S2), and took on the following form:

$$BR = 0.328 - 0.014 \times BF_2 + 0.003 \times BF_3 - 0.020 \times BF_4 \qquad (3)$$

$$BF_2 = \max(0, 174.137 - FS) \qquad (3a)$$

$$BF_3 = \max(0, DC - 98.005) \qquad (3b)$$

$$BF_4 = \max\left(0, \rho_{Populus} - 0.477\right) \qquad (3c)$$

Therein, the burn rate progressively increases as DC increases above 98.005 units; if DC is lower, then there is no effect and BF_3 of eq. 3b takes on the value of 0. Similarly, the burn rate increases as fire season length (FS) increases up to 174 days; if FS is longer, then there is no effect and BF_2 of eq. 3a takes on the value of 0. A high proportion of *Populus* spp. contributes significantly to decreasing the burn rate. Verification of model performance on data withheld from calibration (2011) indicated reasonable predictive skills of this model with a Spearman coefficient of 0.53 ($P = 0.001$; Fig. S3). The model also predicted reasonably well BR computed from the CNFDB over the interval 1971–2000, with a Spearman coefficient of 0.62 ($P < 0.001$; Fig. S4). Other calibration methods led to the selection of the same variables (e.g. General Linear Model), thus confirming the robustness of the MARS model presented here.

Burn rates projections over 2031–2060 and 2071–2100

Increases of DC and FS were projected by 2031–2060 at the northern limits of the Canadian boreal forest under both RCP 4.5 and 8.5 scenarios (Fig. 3), with slightly more areas affected under the RCP 4.5 scenario. By the end of the 21st century, much of the boreal forest was projected to be affected by increases in the DC and FS under the RCP 8.5 scenario, and less so under the RCP 4.5 scenario. Projected changes in FS were particularly important in eastern and northwestern Canada under the RCP 8.5 scenario. In response to these changes, and assuming constant vegetation composition, burn rates were projected to increase across large areas of eastern Canada and extreme northwestern Canada (Fig. 3). Specifically, a threefold increase in the burn rate was predicted for 36% of the locations under the RCP 4.5 scenario by the end of the 21st century, and 42% under the RCP 8.5. Decreases in the DC, and by extension in the burn rates, were projected in some areas of southern and northern Canada's boreal forest (22% of locations under RCP 4.5 had a 10% or more reduction of their burn rate, versus 14% in the RCP 8.5). That being said, locations affected by a decrease in the burn rate are mostly located in areas where the baseline is already low (70% of locations affected by a decrease in the RCP 4.5 have a baseline burn rate <0.1% yr^{-1}, versus 75% in the RCP 8.5).

Examining the impact of changes in the composition of forests

Determination of the forest conversion rates (λ) necessary to maintain the projected burn rates constant from 1971–2000 to 2071–2100 at each of our locations was made by varying the proportion of *Populus* spp. through an iterative procedure implemented in eq. 3. As one can expect from such analysis, the greater was the climate change at location *l*, the higher was the required λ needed to attain the fire objectives (Fig. 4a). First, 31% of locations had a λ value of 0 in the RCP 4.5 scenario (18% in the RCP 8.5) and this is in accordance with the observation that many regions are projected to record either a constant or a decreasing burn rate. Besides, a conversion rate λ set at 0.1% to 0.2% $year^{-1}$ starting in year 2020 was sufficient to keep burn rates constant across much of the southern boreal forest and also within northern regions east and west of Hudson Bay (37% and 38% of locations in RCP 4.5 and 8.5 scenarios, respectively); a λ set at 0.2% $year^{-1}$ equals incrementing the proportion of *Populus* spp. by 5.2% and 13.2% at the mid (2045) and end (2085) of the 21st century, respectively. At the southern margin of the boreal forest, the proportion of *Populus* spp. was already close to the critical limit for saturation of the biotic effect in eq. 3 (Fig. 4b; Table 2). As for the northern regions, fire weather conditions were projected to remain rather low, and therein relatively small increases in the proportion of *Populus* spp. were sufficient to offset the climate change impacts on the projected burn rates. The maximum λ value for these southern and northern regions was also low (from 0 to 1.0), indicating that for many of these regions there would be no to little benefit gained by changing the forest composition (Fig. 4b).

Much larger λ were required to meet the fire objectives across a belt stretching west to east (Fig. 4a). Whereas a λ set at 0.3% $year^{-1}$ was effective in most of the northern boreal forest, in 9% of locations the optimal λ attained 0.4% $year^{-1}$ and over in the RCP 4.5 scenario (25% of locations in the RCP 8.5). That is, to keep burn rates constant, the proportion of *Populus* spp. had to be incremented by at least 10.4% and 26.4% at the mid (2045) and end (2085) of the 21st century, respectively. For most of these locations, the proportion of *Populus* spp. during the baseline period was low (vegetation zones B and I, Fig. 1 and Table S2), thus large increments were permitted before attaining the threshold at which the prescribed biotic feedback was no longer effective. The maximum λ notably attained values up to 0.6% $year^{-1}$ in large parts of northwestern Canada (Fig. 4b).

Discussion

Our study explores the role of forest composition in determining the wildfire risk in boreal Canada under climate change and the potential for forest composition alterations as an emerging adaptation strategy. Our projections of potential fire activity are consistent with previous studies indicating that continuing atmospheric greenhouse gas forcing will create climatic conditions that are more prone to fire in many Canadian regions (Boulanger et al. 2014; Flannigan et al. 2013). Climate change will notably lead to an expansion of regions affected by burn rates > 0.5% by the end of the century, particularly around the Hudson Bay area and at the northwestern limits of the boreal forest (Fig. 3). These trends are a consequent of more frequent fire-conducive days that translate into fire propagation (Wang et al. 2014), an effect that arises from the longer fire seasons and severe summer drought conditions that are projected by the CRCM5 simulations.

However, climate is not the sole driver of fire activity in Canadian boreal forests. The driest climate conditions and longest fire seasons do not match the highest burn rate quantities therein; the occurrence of a high fire season also depends on vegetation flammability (Alexander 2010; Krawchuk and Cumming 2011; Terrier et al. 2013; Wang et al. 2014). In our analysis, burn rates were higher in fire bioclimatic regions characterized by high percent covers of *Picea* spp. and *Pinus* spp. (vegetation zone A) in comparison with the rest of the Canadian boreal forest (Fig. 1, Table S2). The importance of vegetation was also highlighted in our analysis of the pre-burned composition: the proportion of conifers was significantly greater in burned areas in comparison with non-burned areas. Finally, the relative proportion of *Populus* spp. accounted for a significant amount of variability in burn rates across boreal Canada. *Populus* is amongst the most widespread broadleaf genus in Canada, and is mostly represented by the shade-intolerant *Populus tremuloides* Michx. (trembling aspen), and its biotic influence on fire activity in the summer following green-up or flushing of the overstory canopy and understory vegetation is well recognized (Taylor et al. 1996; Alexander 2010). A reasonable case can therefore be made about how burn rate responds to changing vegetation types and how alterations of forest cover can affect burn rates associated with projected changes in climate.

Accordingly, our results suggested that the required optimal forest conversion rate (λ) metric was highly variable across Canada. Notably, many southern and northern boreal regions are predicted to record either a constant or a decreasing burn rate and, therefore, will not require a change in the forest composition (λ = 0.0% year^{-1}). We also found that relatively modest forest composition changes could alter future burn rate trajectories in many areas of the southern boreal forest. Therein, an increase of roughly 13% in the percent cover of *Populus* spp. by the late 21st century (λ = 0.2% year^{-1}) could be sufficient to keep future burn rates at current levels. Higher rates of change of the forest composition were required in northern regions to attain the fire objectives. Therein the required λ metric was generally estimated at 0.3% year^{-1}, with a few locations reaching 0.4 year^{-1} and over.

It should be pointed out that biotic feedbacks may naturally contribute to reducing future burn rates, and to some extent, lowering the magnitude for human intervention. Landscape forest cover and structure are tied to the disturbance regime of a particular landscape, and vegetation shifts to a larger broadleaf component in landscapes can be expected in response to increasing fire activity (Chen et al. 2009; Barrett et al. 2011;; Johnstone et al. 2010; Landhäusser et al. 2010). Notably, in stands where broadleaf species already exist, increases in depth of burning of surface organic layers in black spruce forests can lead to increased recruitment and growth of broadleaf trees (Shenoy et al. 2011). Harvesting and industrialization during recent decades have also already contributed to an increase in the proportion of broadleaf species in the provinces of Ontario and Quebec (e.g. Grondin et al. 2003; Pinto et al. 2008; Laquerre et al. 2011; Boisvert-Marsh et al. 2014) and northern Minnesota (Hanberry et al. 2012; Nowacki and Abrams 2014). For instance, according to Pinto et al. (2008), the relative proportion of conifers decreased by roughly 10-20% in Ontario’s northeastern forests over the past 50 years. In western Quebec, from the 1970s to the 1990s, the proportion of stands that underwent an increase in the cover of broadleaf species is about 30%. These changes in forest composition are largely supplanted by changes in the cover of *P. tremuloides*, an early successional species that regenerates vigorously after disturbances, notably by root suckering in stands where it is already present (Brown and DeByle 1987, Johnstone and Chapin 2006). It is expected that these landscape-cover changes, which tend to follow intense and recurrent site disturbances, will continue to occur if no silvicultural intervention is applied to counter them (Laquerre et al. 2011).

Additionally, northward migration of southern broadleaf species and an expansion of other broadleaf species in the boreal forest are expected to occur over long horizons in response to climate change (Iverson and Prasad 1998, O'ishi and Abe-Ouchi 2009, McKenney et al. 2011; Boisvert-Marsh et al. 2014). This being said, plant species require specific edaphic (soil texture, drainage) characteristics for their establishment and survival on a given site: the potential for expansion and migration (naturally or through management) could be limited by the availability of these sites within a region (Laquerre et al. 2011). Also, an increase in fire frequency and a decrease in fire free interval will increase the amount of immature vegetation present on the landscape and feedback negatively on the amount of burnable areas (Héon et al. 2014). In some areas, limitations from fuel availability may further be exacerbated by a decrease in forested areas and increase in shrublands and woodlands because of regeneration failures (Boiffin and Munson 2013). Finally, broadleaf tree species will not grow in conditions of poor drainage and deep-organic layers such as commonly found in eastern Canada. There, conservation of these attributes could help to reach fire and carbon management goals because moist conditions encountered in these forests provide a level of resistance to the increasing fire activity (Terrier et al. 2015).

While parsimonious, the MARS model used for projecting future burn rate omits many processes that also contribute to fire ignition and spread. Notably, the ignition source (lightning, human) was not included in the calibration of the MARS model. A projection of future burn rate may therefore be misleading if a significant change takes place in the frequency of ignition, particularly in connection with the increased use of forest land by humans or changes in lightning activity. Also, in many regions there is either a distinctly thin or absent deep duff layer and vegetation grows mostly on rock or sand. Under such conditions, the use of the DC may be questionable, and indices with shorter drying lags are likely to be better surrogates for fuel moisture content and for evaluating climate change impacts on burn rates at these locations. Iterative analyses did yield, occasionally, models that included the DSR as a predictor variable. However, these models were less parsimonious than the one retained for our fire predictions, and more difficult to replicate under different statistical approaches. Similarly, our model did not select the broadleaf genus *Betula*, likely owing its low proportions in the sampled dataset and perhaps its limited distribution in the study area (restricted mostly to southeastern Canada). These two factors may have contributed to reducing the detection accuracy of this genus' control on fire activity at the scale at which our analyses were undertaken. Moreover, our analyses only considered the tree species composition rather than forest types as a whole (species abundance, including understory vegetation and forest floor litter and herbs). They do not account for fuel structure, which is an important issue as the vertical continuity from the organic layer to the tree crown is an important aspect for the conductivity to intense crown fires (Terrier et al. 2015).

Affecting species composition over a large forest landscape such as Canada's boreal forest is challenging given the size of areas involved and the relatively low levels of intervention from forest management, particularly in northern regions (Amiro et al. 2001). Therein, the fires will continue to grow despite the presence of broadleaf species as there are no fire management activities to take advantage of the slower moving and lower intensity fires. In southern regions, with appropriate management it can be expected that implementation will be constrained by forest management preferences for conifers. Currently, broadleaf species only account for a fraction of the total planted areas despite increases in the relative proportion of harvested broadleaf volume since the 1990s (Canadian Committee on Forest Management 2012; see Fig. S5). However, there is concern that relatively high burn rates combined with long exposures of stands to fire risk will significantly compromise the amounts of allowable cuts in parts of the boreal forest over the next century, as coniferous stands will unlikely achieve sufficient volume to be harvested prior to burning (Gauthier et al. 2015; Gauthier et al. unpublished). Management practices that tend to favor high stocking of coniferous species, such as broadleaf control strategies or afforestation of woodlands, should be applied with caution (Laquerre et al. 2011; Mansuy et al. 2012). While changes in forest cover are not necessarily desirable for forest conservation and industry, in the long term and with the aid of other fire suppression measures, they may contribute to the achievement of fire management objectives in these impacted regions. Impact scenarios combining forest conversion and disturbance rates with productivity metrics, and a diversity of simulated climate change outcomes, will support decision-making in relation to forest and fire management.

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Figure 1

Map of Canadian fire bioclimatic regions. Fire bioclimatic regions were obtained by intersecting the (A) fire weather and (B) vegetation zones. Mean annual burn rates [% year^{-1}] were computed for each fire bioclimatic region (C) using the 2001–2010 MODIS burned area estimates. Histograms show 10-year averages of the daily drought code (DC, unitless), daily severity rating (DSR, unitless), and fire season length (FS, in days) for each fire weather zone (period 2001–2010), and average proportions of *Populus* spp., *Betula* spp., *Abies* spp., *Picea* spp., and *Pinus* spp. for each vegetation zone (as of 2001).

Figure 2

Comparison in the averaged proportions of a) coniferous (*Abies* spp., *Picea* spp. and *Pinus* spp.) and b) broadleaf (*Betula* spp. and *Populus* spp.) species in burned and non-burned areas for each fire bioclimatic region. Box plots show the distributions in proportions of coniferous and broadleaf species in each fire bioclimatic region (the boundaries of the boxes indicate the 25th and 75th percentiles, the line within the boxes marks the median, and the error bars indicate the 10th and 90th percentiles).

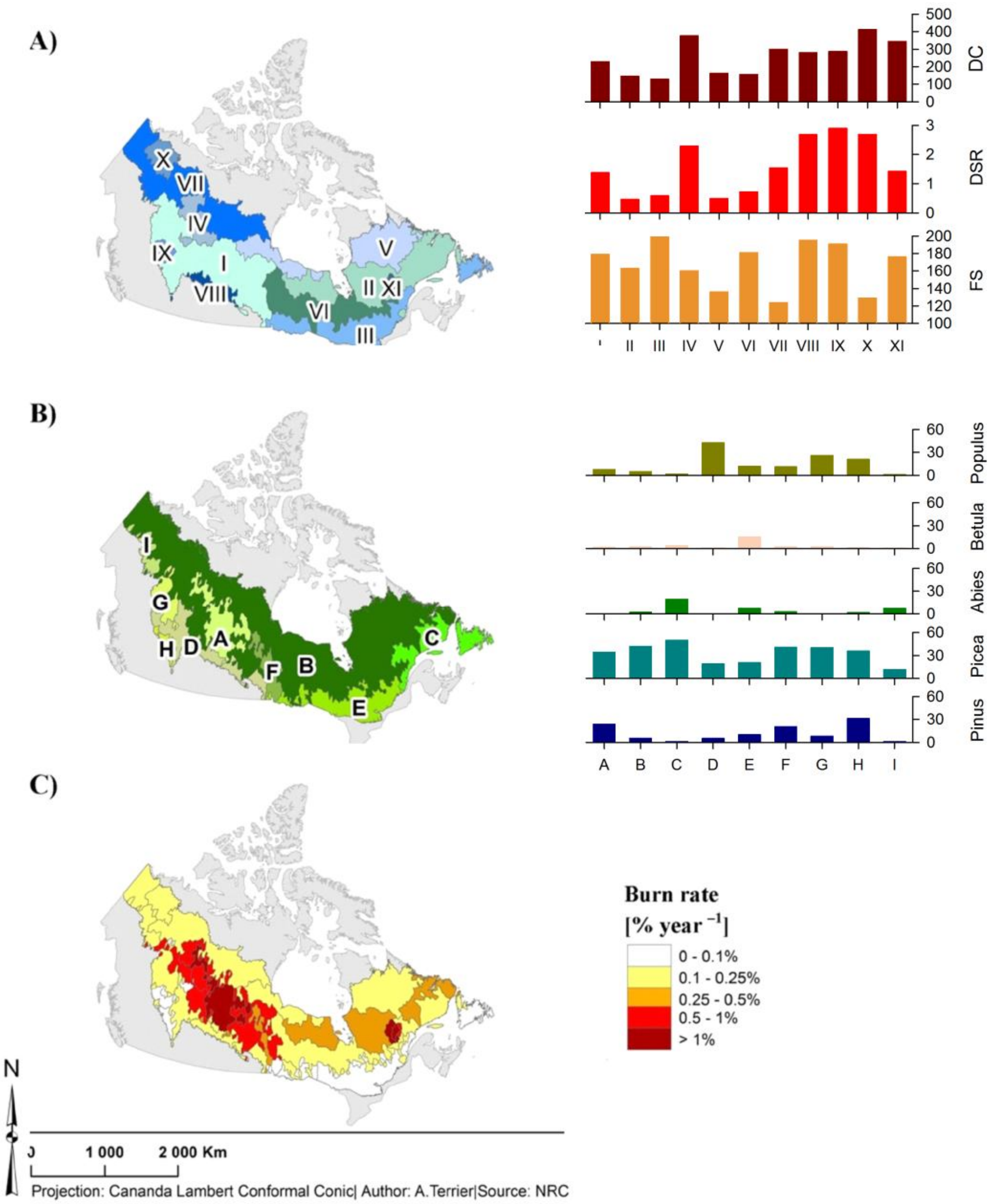
Figure 3

Maps of changing annual means of the daily drought code (DC), fire season length (FS), and burn rates (BR) projected with the Canadian Fourth Generation Regional Climate Model (CanRCM4) under representative concentration pathways (RCP) scenarios 4.5 and 8.5.

Figure 4

A) Optimal conversion rate of forest composition (λ, expressed as % yr^{-1}) that keeps the burn rate metric constant from the baseline (1971–2000) to the future (2071–2100) horizons under the RCP 4.5 and RCP 8.5 scenarios and for each location *l*. A high λ value indicates that a substantial increase in the proportion of *Populus* spp. is required to attain the fire objectives in the simulation, and vice versa. B) Upper bounds of λ for the maximum potential biotic effect for each location *l* under the specified RCP scenario.

Figure 1



A)
X
VII
IV
IX
I
VIII
V
II XI
VI
III
DC
0
100
200
300
400
500
DSR
0
1
2
3
FS
100
120
140
160
180
200
II III IV V VI VII VIII IX X XI
B)
I
G
H D A
F B
C
E
Populus
Betula
Abies
Picea
Pinus
0
30
60
A B C D E F G H I
C)
Burn rate
[% year $^{-1}$]
0 - 0.1%
0.1 - 0.25%
0.25 - 0.5%
0.5 - 1%
> 1%
N
1 000
2 000 Km
Projection: Cananda Lambert Conformal Conic| Author: A.Terrier|Source: NRC

Figure 2

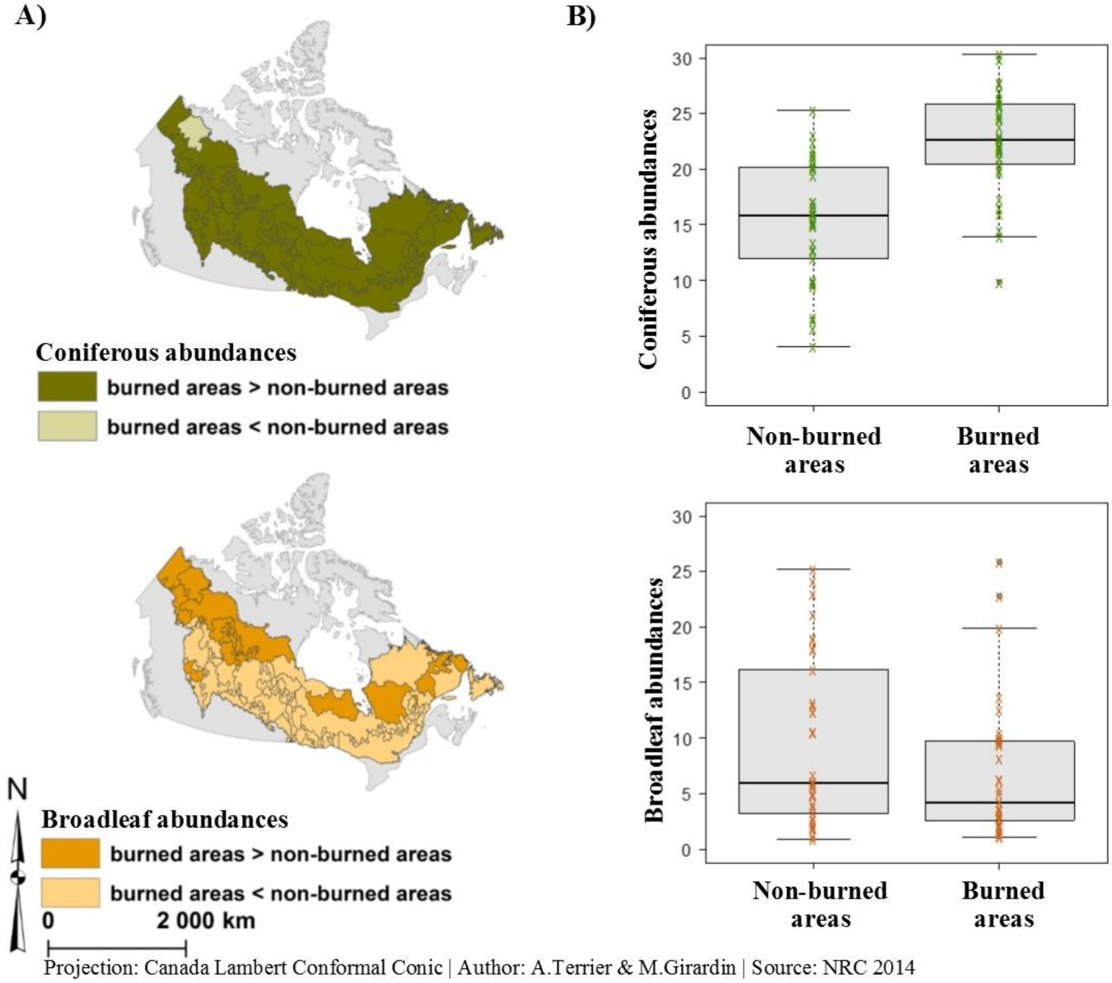

A)
B)
Coniferous abundances
burned areas > non-burned areas
burned areas < non-burned areas
Broadleaf abundances
burned areas > non-burned areas
burned areas < non-burned areas
N
0
2 000 km
Projection: Canada Lambert Conformal Conic | Author: A.Terrier & M.Girardin | Source: NRC 2014
Coniferous abundances
Broadleaf abundances
0
5
10
15
20
25
30
Non-burned areas
Burned areas

Figure 3

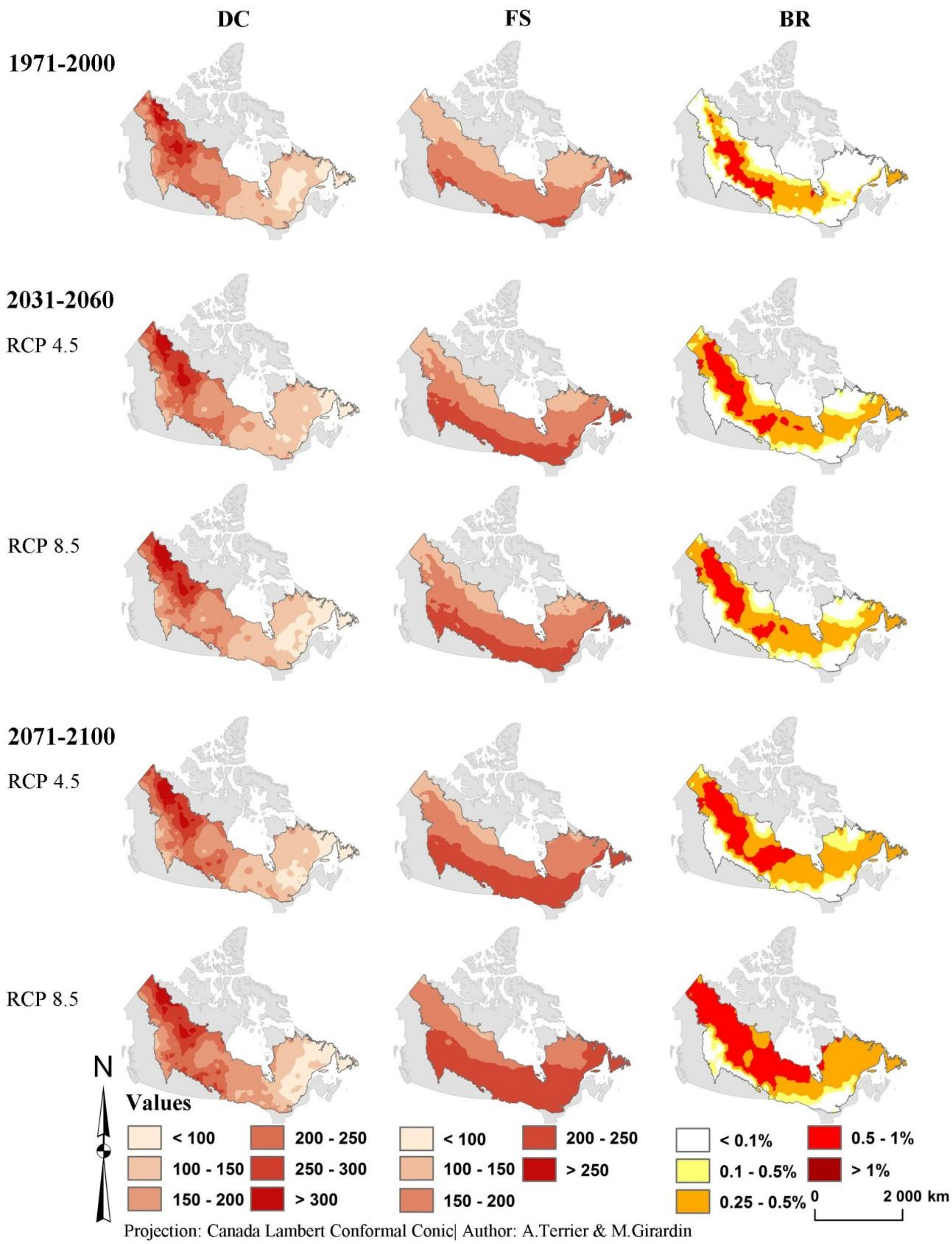
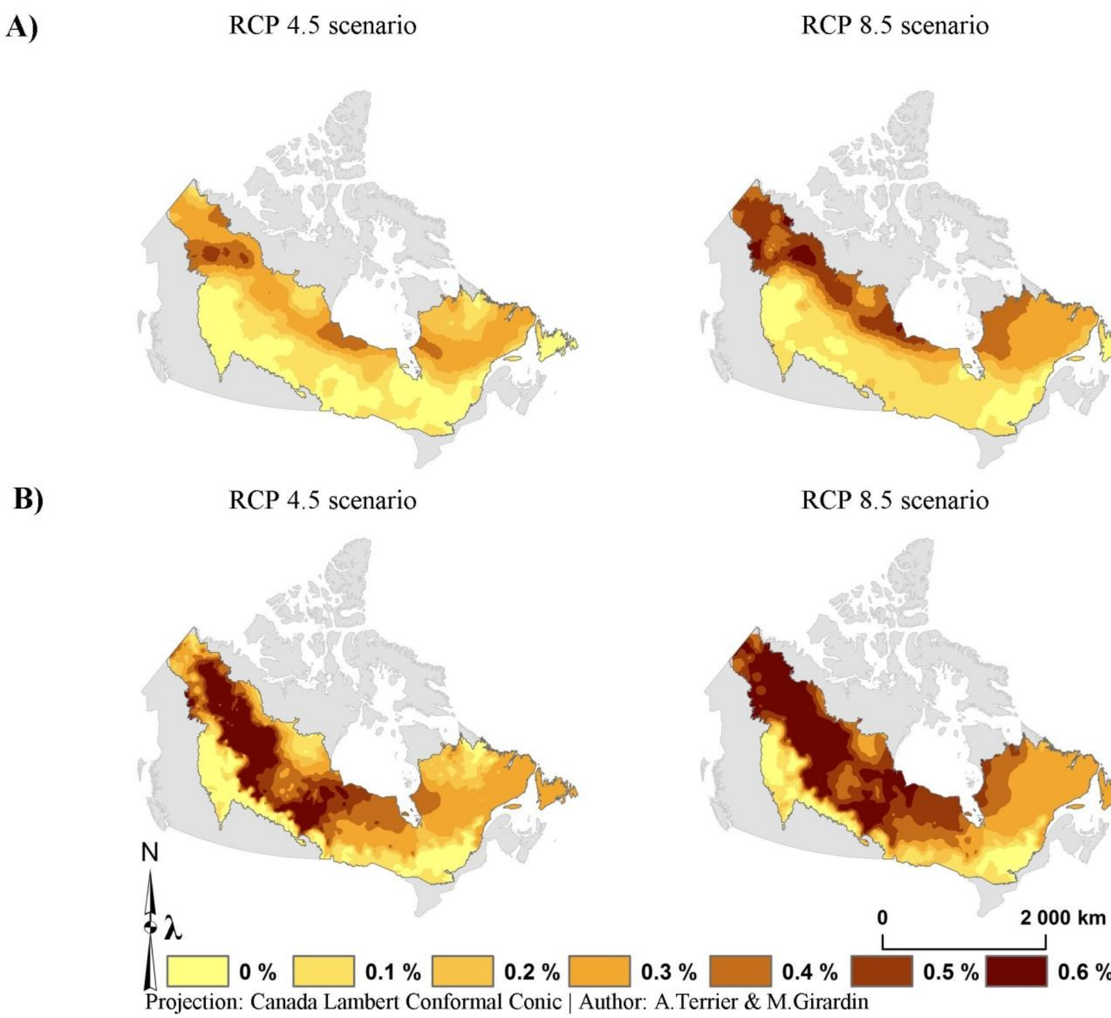

DC
FS
BR
1971-2000
2031-2060
RCP 4.5
RCP 8.5
2071-2100
RCP 4.5
RCP 8.5
N
Values
< 100
100 - 150
150 - 200
200 - 250
250 - 300
> 300
< 100
100 - 150
150 - 200
200 - 250
> 250
< 0.1%
0.1 - 0.5%
0.25 - 0.5%
0.5 - 1%
> 1%
0
2 000 km
Projection: Canada Lambert Conformal Conic| Author: A.Terrier & M.Girardin

Figure 4


A)
RCP 4.5 scenario
RCP 8.5 scenario
B)
RCP 4.5 scenario
RCP 8.5 scenario
N
0
2 000 km
0 %
0.1 %
0.2 %
0.3 %
0.4 %
0.5 %
0.6 %
Projection: Canada Lambert Conformal Conic | Author: A.Terrier & M.Girardin