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Potential changes in forest composition could reduce impacts of climate change on boreal wildfires

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Abstract. There is general consensus that wildfires in boreal forests will increase throughout this century in response to more severe and frequent drought conditions induced by climate change. However, prediction models generally assume that the vegetation component will remain static over the next few decades. As deciduous species are less flammable than conifer species, it is reasonable to believe that a potential expansion of deciduous species in boreal forests, either occurring naturally or through landscape management, could offset some of the impacts of climate change on the occurrence of boreal wildfires. The objective of this study was to determine the potential of this offsetting effect through a simulation experiment conducted in eastern boreal North America. Predictions of future fire activity were made using multivariate adaptive regression splines (MARS) with fire behavior indices and ecological niche models as predictor variables so as to take into account the effects of changing climate and tree distribution on fire activity. A regional climate model (RCM) was used for predictions of future fire risk conditions. The experiment was conducted under two tree dispersal scenarios: the status quo scenario, in which the distribution of forest types does not differ from the present one, and the unlimited dispersal scenario, which allows forest types to expand their range to fully occupy their climatic niche. Our results show that future warming will create climate conditions that are more prone to fire occurrence. However, unlimited dispersal of southern restricted deciduous species could reduce the impact of climate change on future fire occurrence. Hence, the use of deciduous species could be a good option for an efficient strategic fire mitigation strategy aimed at reducing fire propagation in coniferous landscapes and increasing public safety in remote populated areas of eastern boreal Canada under climate change.

Key words: boreal forest; climate change; climatic envelope; deciduous species; future fire occurrence; mitigation management; multivariate adaptative regression splines.

Introduction

It is now well recognized that wildland fires are essential to boreal forest dynamics. They shape forest structure and composition, for instance by increasing landscape-level forest productivity (Johnstone and Chapin 2006, Lecomte et al. 2006) and by favoring the conservation of shade-intolerant species (e.g., *Pinus banksiana*; Johnson 1992). However, fires in boreal forests also have their negative effects owing to their high suppression costs, the infrastructure disasters they cause in remotely populated areas, and the loss of harvestable forests during extreme fire years. In 2010, the Russian boreal forest was affected by several hundred fires due to exceptional drought conditions. A state of emergency was declared and damages were estimated at $15 billion (US$; HuffPost World 2010). Eastern boreal Canada was also hit in 2011 by major fires that forced a state of emergency and the evacuation of communities. In May 2011, for example, wildland fires spread in Alberta, and the Slave Lake fire (western boreal Canada) forced the evacuation of 15 000 residents, causing damage totaling over $700 million (Canadian$; Flat Top Complex Wildfire Review Committee 2012).

The processes governing wildland fire activity operate at several time scales (days, seasons, interannual, decadal) and are influenced by several climatic and environmental factors such as temperature, precipitation, wind, and the structure and composition of forests. From 1905 to 2005, rising concentrations of carbon dioxide in the atmosphere have contributed to a global warming estimated at 0.74°C (± 0.18°C; IPCC 2007).

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The response of the boreal forest to further warming is a major concern because high-latitude boreal regions are likely to be most affected by these changes (IPCC 2007), and the expected response of fire activity is closely linked to warmer and drier weather (Balshi et al. 2009). Recent temperature increases have been associated with increasing fire activity in Canada since about 1970 (Gillett et al. 2004) and exceptionally warm summer conditions in Russia during the 2010 fire season (Rahmstorf and Coumou 2011). This increased warming will likely result in more fire-prone weather conditions by the end of the 21st century (Yang et al. 2011) that will directly impact the number of fire occurrences (Girardin and Mudelsee 2008, Flannigan et al. 2009, Wotton et al. 2010) and area burned (Amiro et al. 2009, Balshi et al. 2009, Flannigan et al. 2009, Le Goff et al. 2009, Bergeron et al. 2010). Fire management capacity will eventually be overwhelmed (Podur and Wotton 2010).

Predictions of future fire activity are generally obtained using empirical models calibrated with variables describing the processes of drying in organic soil layers and fire behavior (Flannigan et al. 2005, Bergeron et al. 2006, Girardin and Mudelsee 2008, Lafleur et al. 2010). Most of these models do not consider feedback effects on fire ignition and spread resulting from changes in vegetation and fuel types (Flannigan et al. 2001, Hély et al. 2001, Krawchuk et al. 2009, Hessl 2011). Models assume that the vegetation component will remain relatively static in the course of the next few decades. However, deciduous species are less flammable than coniferous species (Päätalo 1998, Campbell and Flannigan 2000, Hély et al. 2001, 2010, Lefort et al. 2004, Arienti et al. 2006, Krawchuk et al. 2006) and thus one could expect that the increasing risks brought about by more fire-prone climatic conditions could be offset by an increasing deciduous component in boreal landscapes. While in the long term a potential northward migration of limited temperate deciduous species and an expansion of other deciduous species in the boreal forest are expected to occur in response to climate change (Iverson and Prasad 1998, McKenney et al. 2007, O'ishi and Abe-Ouchi 2009, O'ishi et al. 2009, Berteaux et al. 2010, McKenney et al. 2011), it might not be the case in the medium term owing to low species migration and dispersal rates. Nevertheless, the question of potential vegetation feedback on fire activity is important when placed in the following context: to what extent can changes in vegetation offset predicted increases in fire risk driven by more severe and frequent drought conditions if species dispersal is unlimited or facilitated through, for instance, strategic forest management planning aimed at mitigating increasing climate risks? Such planning could include fuel treatments through prescribed burning and modification of vegetation composition around forest communities to reduce the fuel load (e.g., Hirsch et al. 2004).

The objective of this study is to determine whether increases in the occurrence of boreal fires predicted to occur by the end of the 21st century can be mitigated by changing the vegetation composition. We used a previously published method of predicting future fire behavior coupled with ecological niche models that take into account the effect of changing tree species distribution. Ecological niche models present correlative descriptions of the current environment and species distribution and, based on predicted future environmental conditions, they can be used to project future species' suitable ranges (Franklin 2009). In order to fulfill data requirements for the wildfire and ecological niche models, we focused our modeling effort on eastern boreal North America. Three hypotheses related to wildfire risk response to climate change were statistically tested. These hypotheses were formulated on the basis of the widely accepted evidence that temperatures will be rising in boreal regions over the present century (IPCC 2007) and that fire activity will be increasing (e.g., Flannigan et al. 2009, Bergeron et al. 2010). The hypotheses are that (1) weather and tree composition are both important explanatory variables of fire occurrence in boreal forests; (2) future climate conditions will be more conducive to fire; and (3) changes in tree composition may limit the increase in fire occurrence.

Study Area

Our study area is located in the province of Quebec, Canada (Fig. 1). The climate is predominantly continental, with warm and short summers, and cold, long, and snowy winters (Natural Resources Canada 2007). Temperature and precipitation differ across the province due to maritime effects, latitude, topography, and the presence of the Labrador Current along the east coast (Richard 1987). Mean annual temperature decreases with latitude and elevation, ranging from 7°C in the south to −3.1°C in the north (Natural Resources Canada 2007). Total annual precipitation varies from 800 to 1600 mm, with maximum values in the eastern part due to maritime effects (Richard 1987, Natural Resources Canada 2007). Forests located south of 47° N are dominated by deciduous and mixed stands. Dominant tree species include, but are not restricted to, sugar maple (*Acer saccharum* Marsh.), yellow birch (*Betula alleghaniensis* Britt.), beech (*Fagus* sp.), balsam fir (*Abies balsamea* (L.) Mill.), red pine (*Pinus resinosa* Ait.), and white pine (*Pinus strobus* L.). Coniferous species cover small areas (Bérard et al. 1996, Saucier et al. 1998) and the region is rich in vascular plants (>1600 species) and tree species (~40 species) (Richard 1987). Forests located between 47° and 58° N are dominated by conifers, including black spruce (*Picea mariana* (Mill.) BSP.), white spruce (*Picea glauca* (Moench) Voss.), and balsam fir. However, deciduous species such as trembling aspen (*Populus tremuloides* Michx.), paper birch (*Betula papyrifera* Marsh.), and yellow birch also occupy extended areas in these forests (Bérard and Côté 1996, Saucier et al. 1998). Vascular plants are abundant in deciduous stands, and abundance, diversity, and evenness decrease with

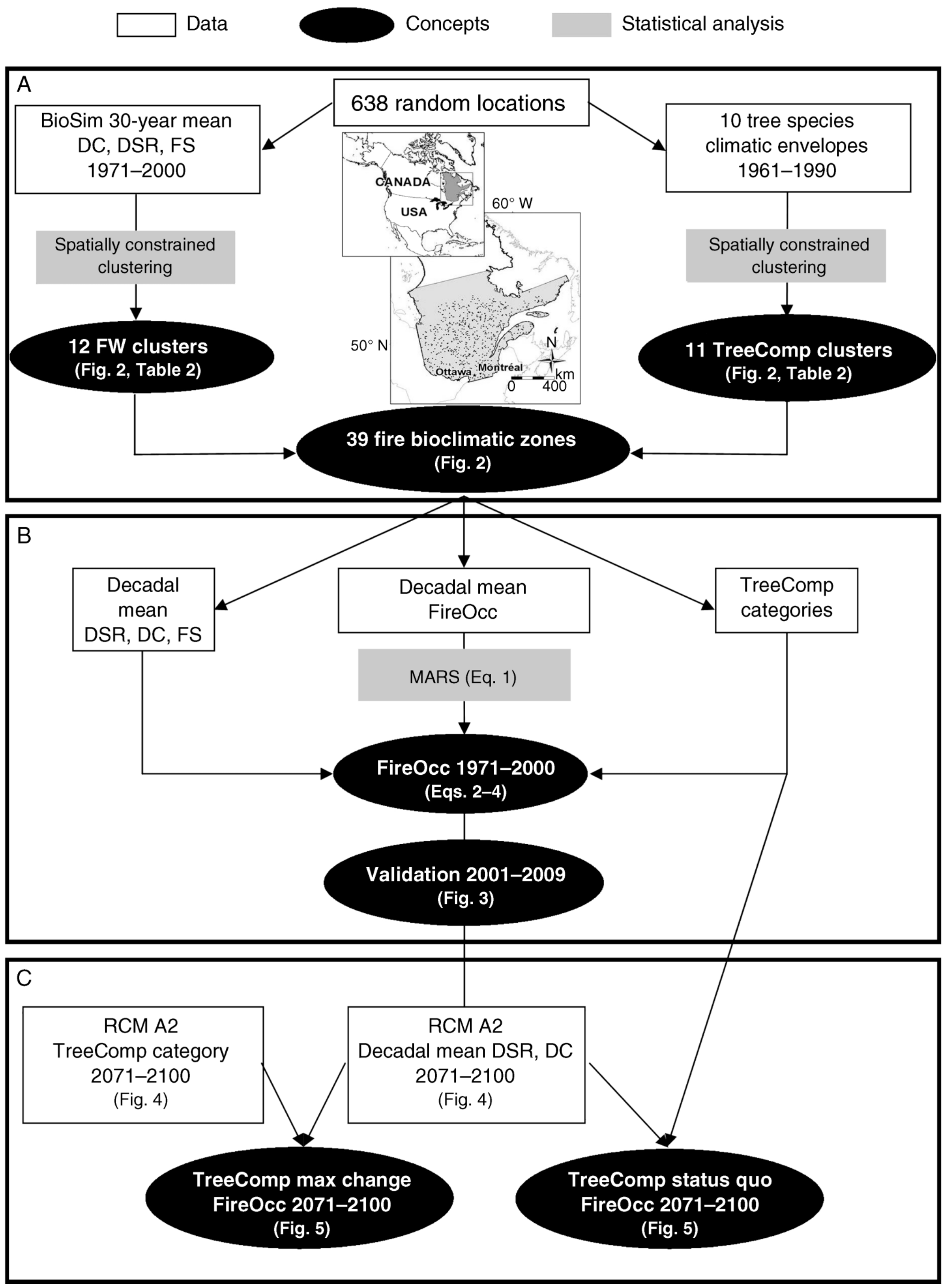

FIG. 1. Diagram of the statistical analyses conducted in this study. (A) Quebec regionalization based on conditions (fire weather and tree composition) prone to fire from 1971 to 2000. (B) Modeling of fire occurrence with fire weather variables and tree composition categories from 1971 to 2000. (C) Fire occurrence projections for 2071–2100.

increasing coverage of conifer species (De Grandpré et al. 1993, Hart and Chen 2006). Where coniferous trees dominate, lichens and mosses occupy the subarctic forest floor (Gauthier et al. 2000, Bergeron et al. 2002). Above 58° N, the subarctic tundra forest extends from the continuous forest limit to the northernmost limit of tree growth (Payette 2001). Plant diversity is poor and landscapes are generally occupied by scattered black spruce individuals (Richard 1987, Payette 2001).

METHODS

The structure and organization of the methods used are illustrated in Fig. 1. Hereafter, we formulate the FireOcc quantity as follows:

$$\text{FireOcc}_j = \sum (c_1\text{BF}_1 \times \text{FW}_j + c_2\text{BF}_2 \times \text{TreeComp}) \quad (1)$$

where FireOcc is the number of fires per year per 1000 km^2 for a given fire size class (≥ 1 ha; ≥ 10 ha; ≥ 200 ha) for the decade j. Predictor variables are sets of fire bioclimatic zones determined from fire weather (FW) variables and tree species composition (TreeComp). Finally, c_1 and c_2 correspond to constants, while BF_1 and BF_2 are basis functions for nonlinear interactions (see *Methods: Parameterization of the FireOcc model*). For the purpose of developing predictions of future FireOcc that take into account regional climate and tree composition changes, the following intermediate analyses were undertaken: (1) division of the study area into fire bioclimatic zones using a clustering method applied to gridded FW variables and tree species distributions; (2) parameterization of the FireOcc model using a nonlinear regression technique relating FireOcc to FW variables and TreeComp; and (3) inclusion of simulation outputs of a regional climate model into the FireOcc models. Data and methods are described in detail in the following sections.

Fire statistics

Forest fire data from the Ministère des Resources Naturelles et de la Faune du Québec were used for this study. The database contains information on the location, date of detection, size (ha), and cause (lightning or human) of all fires recorded in the province of Quebec. The period covered by the data encompasses that during which systematic fire detection was made by detection planes. We considered lightning fires from 1971 to 2009 only. Fires of size <1 ha were not included as the database for these fires is considered incomplete (Boulanger et al. 2012). Fires <10 ha of unknown origin were removed from the analysis. Remaining fires were grouped in the following size classes: ≥ 1 ha, ≥ 10 ha, and ≥ 200 ha. These classes correspond to the first, second, and third percentiles of the fire size distribution.

Climate data and fire weather (FW) variables

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. Briefly, the FWI System calculates three fuel moisture codes at different forest floor levels based on daily temperature, precipitation, relative humidity, and wind velocity. These codes are the fine fuel moisture code (FFMC), duff moisture code (DMC), and drought code (DC). FFMC estimates the moisture contents of the litter and other fine fuels in a forest stand in a layer of ~ 0.25 kg/m^2 dry mass. It is an indicator of sustained flaming ignition and fire spread. DMC represents the average moisture content of loosely compacted, decomposing organic layers of moderate depth weighing ~ 5 kg/m^2 when dry. It relates to the probability of lightning ignition and fuel consumption. DC represents the average moisture content of deep, compact organic layers (about 10–25 cm from the surface) weighing ~ 25 kg/m^2 when dry. It relates to the consumption of heavier fuels and the effort required to extinguish a fire. For a temperature of 25°C, relative humidity of 30%, and wind speed of 10 km/h, the response times of the FFMC, DMC, and DC are ~ 0.5, 10, and 50 days, respectively (Wotton 2009). These moisture codes feed into other codes related to fire behavior, including a 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). It is important to note that the FWI System estimates fire behavior without regard for fuel types and, hence, forest composition (Van Wagner 1987, Wotton 2009). All indices are unitless, with the zero value indicating low fire risk and high values indicating high fire risk. Winter precipitation is included in the algorithms of the FWI so fire behavior indices also depend on snow accumulation (Girardin and Wotton 2009). Additionally, we also considered the length of the fire season (FS) as a potential predictor of FireOcc. The start of the fire season was assumed to begin either on the third consecutive day with noon temperature above 12°C, or three days after snowmelt (i.e., after seven consecutive days with <2 cm of snow on the ground; Brown et al. 2003), depending on which of the two criteria came first. The end of the season was set based on the accumulation of >2 cm of snow on the ground for seven consecutive days or on the occurrence of three consecutive days with daily minimum temperature <0°C, whichever happened first (Brown et al. 2003).

To begin with the computation of the FW variables, a set of 1000 locations was randomly selected across the study area using a random location list generator, 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 differences in latitude, longitude, and elevation between the data sources and the location, and averaged using a $1/d^2$ weight, where d is distance. Data for the 1971–2009 period were interpolated from Environment Canada's historical climate database (Environment Canada 2011). Data for 2071–2100 were obtained from gridded simulation outputs of the Canadian regional climate model, version 4.1.1 (CRM4.1.1 acs and act runs; Biner et al. 2007, Music and Caya 2007). The simulations were carried out on a horizontal grid-size mesh of 45 km. Simulations were performed using the IPCC SRES A2 emission scenario (Nakićenović et al. 2000). The A2 storyline from which it is developed represents a very heterogeneous world with continuously increasing global population and regionally oriented economic growth that is more fragmented and slower than in other SRES storylines. The CO_2 concentration therein increases from 476 ppm in 1990 to 880 ppm by the late 21st century. To account for differences between model development data and RCM predictions, the delta method was applied. This method involves calculating differences in temperature and ratios of precipitation, wind speed, and specific humidity projected by the RCM model in relation to the model's average climate during the time period for which historical climate data are available (i.e., 1971–2000). Those changes are then added (for temperature) or multiplied (for precipitation, wind speed, and specific humidity) to historical climate data averages. Specific humidity was converted into relative humidity.

Mapping of tree species' potentially suitable habitats

Previously published tree species distributions predicted by ecological niche models for the baseline and future periods were used to estimate tree species presence and absence at each of our locations (Berteaux et al. 2010). The potentially suitable habitats predicted by the models correspond to areas that species can occupy under current and changing climate conditions without any dispersal constraint (McKenney et al. 2007, Engler and Guisan 2009). Details on the data and methods can be found in Berteaux et al. (2010). Briefly, multiple statistical models of species-suitable habitats were parameterized using Quebec data on forest composition, altitude, soil types, soil drainage class, and climate (1961–1990 normals). These multiple statistical models were then averaged to obtain ensemble means, and parameter settings were applied to adjusted outputs of the Canadian regional climate models and scenarios described earlier for the projection of future suitable habitats. We retained the following 10 tree species, which are generally considered dominant in Quebec's bioclimatic domain (Saucier et al. 1998): balsam fir, sugar maple, yellow birch, paper birch, bitternut hickory (*Carya cordiformis* (Mill.) K.), black spruce, white spruce, jack pine (*Pinus banksiana* Lamb.), trembling aspen, and American basswood (*Tilia americana* L.). All results were centered on 2080 for future periods. Locations without tree information were deleted. We obtained a total of 638 random locations for analysis (Fig. 1).

Spatial clustering

Baseline reference conditions for the different FW variables were computed at each of our locations from the averages of the daily quantities over 1971–2000. Spatially constrained clustering (Legendre and Fortin 1989, Legendre and Legendre 2012) was then applied to the baseline FW variables, and to tree species distributions to divide the study area into homogeneous fire and tree composition zones, respectively (Fig. 1). For several of our locations, the fire weather variables were calculated from the same weather stations, which induces an overlap in location information and, hence, an inflated autocorrelation. Spatial clustering analysis made it possible to eliminate this autocorrelation. Owing to the high collinearity between some of the FW predictor variables, we restrained our application of the clustering method to DC, DSR, and FS variables; sensitivity analyses indicated that inclusion of other variables such as temperature and other FW variables did not improve model performance. We computed space-constrained agglomerative clustering using a multivariate dissimilarity (distance) matrix (Legendre 2011). Ward's minimum variance hierarchical cluster analysis was used as the clustering method (Ward 1963). Hellinger transformation was applied to the tree species distribution data set followed by calculation of Euclidian distance; the result was a matrix of Hellinger distances among sites (Legendre and Gallagher 2001). The number of clusters that minimized the cross-validated residual error (CVRE) was retained. One hundred cross-validation iterations were also conducted to calculate the CVRE. The clustering analysis was performed with the "const.clust" package (Legendre 2011) included in the R freeware (R Development Core Team 2010).

Fire bioclimatic and future tree composition zone delimitation

Fire bioclimatic zones were delimited using FW variables and TreeComp spatial clustering results. This was done to obtain homogeneous zones of both FW variables (FW clusters) and tree composition (TreeComp clusters). Spatial clustering analysis made it possible to assign to each location two values corresponding to its cluster membership (FW variables and TreeComp). The "factor" function in the R freeware was used to encode each location with a cluster code (one fire bioclimatic zone). A new cluster was created when two points belonged to the same FW cluster, while their TreeComp cluster was different, or vice-versa.

To reflect that tree-suitable habitats would change in response to climate change, new TreeComp clusters were assigned. Potential presence or absence of the 10 tree species over the 2071–2100 period was attributed to each of the 638 locations from habitat suitability models centered on 2080. New TreeComp clusters were obtained for each location by calculating Hellinger distances between locations and centroids of TreeComp clusters for the 1971–2000 period. The cluster with the shortest Hellinger distance was assigned to each location. Maps of fire and vegetation zones were then delimited by agglomerating the Thiessen polygons of the locations for each cluster using ESRI ARCGIS 9.3 (ESRI, Redlands, California, USA).

Parameterization of the FireOcc model

Development of a predictive model for FireOcc was carried out using multivariate adaptative regression splines (MARS; Friedman 1991). Descriptions of the method are provided by Leathwick et al. (2006) and by Balshi et al. (2009). MARS is a nonparametric spline regression approach that models nonlinear relationships between a response variable (e.g., FireOcc) and predictor variables (e.g., FW variables and TreeComp). The main principle is the division of the space of explanatory variables into regions. A set of linear regressions, named basis functions (BF in equations), are then fitted for each region to describe the relationships between the response and explanatory variables. Knots separate regions and correspond to positions where the slope of basis functions changes. The procedure builds models in a parsimonious manner by minimizing mean square error (MSE), while selecting combinations of variables and the number and location of knots in a forward stepwise manner (Friedman and Roosen 1995). It starts with a maximum of candidate knot locations and all predictor variables. Progressively, knots and variables that contribute the least to the fitting are removed. Generalized cross-validation (GCV) selects the model with the best predictive fit. Various studies have illustrated the strong performance of MARS models in various ecological studies (De Veaux et al. 1993, Abraham and Steinberg 2001, Moisen and Frescino 2002, Leathwick et al. 2006), particularly under moderate sample sizes ($50 < N < 1000$; Friedman and Roosen 1995). This method has previously been used in the modeling of burned areas in Canada (Balshi et al. 2009, Bergeron et al. 2010).

Decadal averages (1971–1980, 1981–1990, 1991–2000, and 2001–2009) of the FW variables were computed at each of our locations from the averages of daily quantities. FW predictors were the same as those used for the cluster analysis. These decadal averages were then aggregated to the level of the fire bioclimatic zones using averaging and used as input in Eq. 1 (Fig. 1). Decadal averages of annual FireOcc for the different size classes ($\geq$1 ha; $\geq$10 ha; $\geq$200 ha) were also computed at the level of fire bioclimatic zones and used as the response variable in Eq. 1. Decadal averages were used instead of annual or long-term averages to avoid having too many zeros in the response matrix and to satisfy variance requirements. TreeComp was entered in the form of binary variables to indicate the presence of a given vegetation category. Models were computed with the 1971–2000 FW decadal data and baseline TreeComp data ($n = 117$ observations), and verified with independent FW decadal data covering the 2001–2009 and baseline TreeComp data ($n = 39$ observations). Note that here we made the assumption that TreeComp did not change from one decade to the other. Goodness of fit over the independent period was measured using the regression R^2 of observed data as a function of predicted data. MARS models were computed using Salford System Software (Salford Systems 2001). The nonparametric method of this software has the advantage to support zero values. A maximum of two interactions was allowed, but interactions between TreeComp and FW variables were disabled. Other software parameters were set by default.

Predictions of FireOcc in a changing climate

Our final goal was to produce a map of the potential response of fire occurrence (FireOcc; Eq. 1) to climate change over the 2071–2100 horizon. Two assumptions were made to account for the migration ability of each tree species. The no migration scenario assumed that species dispersal was null. It represents the status quo, in which TreeComp 2071–2100 is the same as TreeComp 1971–2000. In the second scenario, unlimited dispersal assumes that climate, topographic, and edaphic conditions are the only factors that limit dispersal of tree composition. Scenarios of no migration vs. unlimited dispersion are widely used to interpret predictions from species distribution models (Araújo et al. 2005, Thuiller et al. 2006, McKenney et al. 2007, Engler and Guisan 2009, Meier et al. 2011). We applied the FireOcc models to the 1971–2000 decadal averages of the FW variables and to the TreeComp variables across the 638 locations. Prediction of future FireOcc was done by substituting historical FW and TreeComp conditions by the future ones obtained from the RCM simulations. Decadal results were averaged to obtain one value for a 30-year period. FireOcc 1971–2000, FireOcc 2071–2100, and FW ratio of change ([FW 2071–2100]/[FW 1971–2000]) were interpolated from the 638 locations to obtain continuous maps. We used ordinary kriging interpolation in ESRI ARCGIS 9.3 with 1-km grid mesh and a spherical model to fit the variograms. Significant differences between future projections and current values were tested using Student's *t* test; points that passed the 5% significance level were projected on the map.

Results

Analysis of baseline conditions

Constrained spatial analysis led to the delimitation of 39 fire bioclimatic zones (Fig. 2). FireOcc varied spatially with values ranging from 0.02 to 0.90, 0 to 0.39, and 0 to 0.15 fires per year per 1000 km^2 for FireOcc $\geq$1 ha, $\geq$10 ha, and $\geq$200 ha, respectively (results not shown). The lowest fire occurrences were observed in the eastern fire bioclimatic zones (Fig. 2C). Zones of low FireOcc $\geq$1 ha are coherent with the relatively low DC and DSR indices observed in eastern zones (I, III, VIII, XI; Fig. 2A, Table 1).

FireOcc $\geq$10 ha was low or null in the hotter and drier southern fire bioclimatic zones (Fig. 2C). The majority of these fires occurred in regions where habitats are unsuitable for American basswood, bitternut hickory, and sugar maple (Fig. 2B, Table 2). FireOcc $\geq$200 ha was concentrated in the northwestern part of Quebec, where climatic conditions are suitable for fire-prone coniferous species and, to a much lesser extent, *Populus tremuloides* (Fig. 2B, Table 2). Values of DSR and FS are higher in the south than in the north (Fig. 2A, Table 1), indicating that fire risk should be higher in the south. Values of DC did not show the similar north–south trends.

Predictive models of FireOcc

We regressed the FireOcc quantity for each fire size class against FW variables and TreeComp variables using

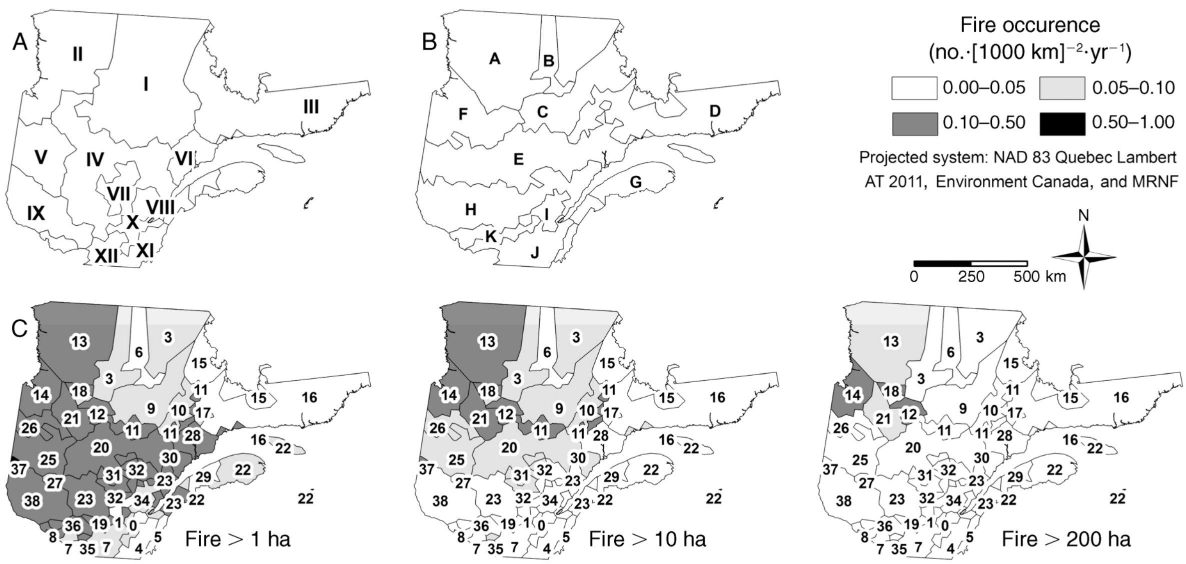

FIG. 2. Map of fire bioclimatic zones in Quebec. Fire bioclimatic zones were obtained by intersecting (A) the fire weather (FW) variables clustering and (B) the tree species distribution clustering. Annual natural forest fire occurrences (number of fires per year per 1000 km^2) of all (fire $\geq$ 1 ha), medium (fire $\geq$ 10 ha), and large fires (fire $\geq$ 200 ha) were calculated for (C) each fire bioclimatic zone. The period of analysis is 1971–2000.

MARS. Models explained 64%, 79%, and 30%, respectively, of the deviation between FireOcc $\geq$1 ha (generalized cross-validation, GVC $R^2 = 0.62$), FireOcc $\geq$10 ha (GVC $R^2 = 0.76$) and FireOcc $\geq$200 ha (GVC $R^2 = 0.22$). Verification of model performance on independent data (2001–2009 period) indicated good predictive skills for size classes $\geq$1 ha and $\geq$10 ha (Fig. 3) with R^2 of 0.32 and 0.51, respectively. Predictive skills for size class $\geq$200 ha were low with $R^2 = 0.17$. DSR was selected as a predictor of FireOcc for fire size classes $\geq$1 ha and $\geq$10 ha, while DC was selected as a predictor for fire size class $\geq$200 ha. Predictors and the model for fire size class $\geq$1 ha took on the following form:

$$\text{FireOcc} \geq 1\ \text{ha} = \max(0; 0.11 + 9.29 \times \text{BF}_2) \quad (2)$$

$$\text{BF}_1 = \Big(J(0)\Big) \quad (2a)$$

$$\text{BF}_2 = \max(0; \text{DSR} - 0.92) \times \text{BF}_1 \quad (2b)$$

so that FireOcc progressively increases as DSR increases above 0.92 units; if DSR is smaller than 0.92, BF_2 of Eq. 2b takes on the value of 0 (no fire). On the other hand, the presence of tree composition category J contributes significantly to a decrease in FireOcc (via Eq. 2a, in which BF_1 takes on a value of 0 in the presence of this compositional group). This compositional group contains all tree species included in the analysis, except jack pine. More specifically, it is the only group that combines sugar maple, bitternut hickory, and American basswood (Table 2).

Model results for size class $\geq$10 ha are more complex and include several compositional groups:

$$\text{FireOcc} \geq 10\ \text{ha} = 0.25 - 0.11 \times \text{BF}_3 - 0.10 \times \text{BF}_4 + 9.51 \times \text{BF}_6 \quad (3)$$

$$\text{BF}_3 = \Big(F(0)\Big) \quad (3a)$$

$$\text{BF}_4 = \Big(A(0)\Big) \quad (3b)$$

$$\text{BF}_5 = \Big(J(0)\Big) \quad (3c)$$

$$\text{BF}_6 = \max(0; \text{DSR} - 1.02) \times \text{BF}_5 \quad (3d)$$

so that FireOcc progressively increases as DSR increases above 1.02 units (Eq. 3d). The presence of tree compo-

TABLE 1. Summary of 30-year averages (1971–2000) of daily drought code (DC), daily severity rating (DSR), and fire season length (FS, in days) for each region of Quebec (FW clusters) computed from the fire weather (FW) variables (DC, DSR, and FS).

FW clusters	DC	DSR	FS
I	89.38	0.33	162.68
II	130.03	0.58	167.75
III	102.04	0.37	157.26
IV	101.34	0.51	174.56
V	108.04	0.71	180.04
VI	122.54	0.73	171.46
VII	118.69	0.80	181.26
VIII	61.19	0.39	164.21
IX	120.74	0.73	187.73
X	97.16	0.68	192.56
XI	74.94	0.41	193.03
XII	127.93	0.96	199.43

TABLE 2. Tree species composition in each region of the province of Quebec based on tree species occurrences.

	Site clusters										
Species	A	B	C	D	E	F	G	H	I	J	K
Coniferous											
Balsam fir		X	X	X	X	X	X	X	X	X	X
Jack pine	X		X	X	X	X	X	X	X		X
White spruce				X	X		X	X	X	X	X
Black spruce	X	X	X	X	X	X	X	X	X	X	X
Deciduous											
Sugar maple							X	X	X	X	X
Bitternut hickory										X	
Yellow birch					X		X	X	X	X	X
White birch					X		X	X	X	X	X
Trembling aspen				X	X	X	X	X	X	X	X
American basswood								X		X	X

Note: Spatially constrained site clusters (columns) are identified by letters A–K.

sition category J contributes significantly to a decrease in FireOcc (Eq. 3c, in which BF_5 takes on a value of 0 in the presence of this compositional group). In contrast, compositional groups F (Eq. 3a) and A (Eq. 3b) contribute significantly to increasing FireOcc as BF_3 and BF_4 take on a value of 0 in their presence. Both compositional groups are dominated by coniferous species (Table 2).

Finally, the following model was selected for fire size class $\geq$200 ha:

$$\text{FireOcc} \geq 200\text{ha} = 0.08 - 0.07 \times BF_7 + 0.004 \times BF_9 \quad (4)$$

$$BF_7 = \Big(F(0)\Big) \quad (4a)$$

$$BF_8 = \Big(K(0)\Big) \quad (4b)$$

$$BF_9 = \max(0; DC - 125.14) \times BF_8 \quad (4c)$$

so that FireOcc progressively increases as DC increases above 125.14 units (Eq. 4c). The presence of tree composition category K (essentially deciduous; Eq. 4b) contributes significantly to a decrease in FireOcc (via BF_8, which takes on a value of 0 in the presence of this compositional group). In contrast, compositional group F contributes significantly to increasing FireOcc as BF_7 (Eq. 4a) takes on a value of 0 in its presence.

FireOcc projections over 2071–2100

Projections of explanatory variables and FireOcc under the A2 IPCC scenario (greenhouse gas and aerosol projected changes) are shown in Figs. 4 and 5. Increases of DSR and DC are predicted under the A2 IPCC climate change scenario over almost all of the study area (Fig. 4). More specifically, the predicted increases of DSR are higher, with ratios of (DSR 2071–2100)/(DSR 1971–2000) = 1.5–2.5, with a maximum reaching above 2.5. Higher increases are predicted for the northern and eastern parts of the study area. In response to DSR change, FireOcc $\geq$1 ha and FireOcc $\geq$10 ha are predicted to increase (Fig. 5, status quo scenario). The southern areas of the boreal forest will be affected by higher increases of FireOcc $\geq$1 ha and FireOcc $\geq$10 ha. Small DC increases are predicted for the north. DC increases get increasingly important in southern regions of the boreal forest (Fig. 4; [DC 2071–2100]/[DC 1971–2000] = 1–1.5). Predicted changes in the frequency of large fires ($\geq$200 ha) are generally not significant under the status quo scenario, except for some southern locations (Fig. 5).

The A2 IPCC unlimited tree dispersal scenario of tree composition showed that future climatic conditions in the north could be suitable for the expansion of southern tree species, especially for tree composition categories containing sugar maple, American basswood, and bitternut hickory (expansion of categories G, H, I, J; Fig. 4). Increases of FireOcc $\geq$1 ha and FireOcc $\geq$10 ha are predicted to be less important in the boreal forest, and trends could even be reversed under unlimited dispersal scenarios. More specifically in the southern boreal, FireOcc $\geq$1 ha is predicted to be similar to baseline conditions and FireOcc $\geq$10 ha is expected to decrease. Changes in FireOcc $\geq$200 ha should be more significant because northwestern regions that are currently affected by frequent large fires are predicted to undergo a change toward a lower frequency of small- and medium-size fires if a change in tree composition category occurs (Fig. 4)

DISCUSSION

This study is the first that we are aware of to report on the integration of ecological niche models and fire weather indices in empirical fire models with the objective of projecting future spatial patterns of wildland fires in boreal forests. Our description of fire occurrence distributions was based on clustering of fire weather and vegetation components. This approach differs from previous studies in which fire properties were projected at the scale of ecozone and ecoregion classifications of the National Ecological Framework of Canada (NEFC; e.g., Bergeron et al. 2004, Lefort et al. 2004, Flannigan et al. 2005, Girardin and Mudelsee 2008). Boulanger et al. (2012) recently showed that the use of these NEFC zones could prevent researchers from capturing the real fire spatial variability. Spatially constrained clustering of fire weather indices should address some of the limitations reported by Boulanger et al. (2012). In addition, our approach made it possible to circumvent the problem of spatial dependence in locations induced by our modeling experimentation, notably that resulting from the spatial interpolation of weather data in the fire weather calculations. Finally, we used RCM output instead of global climate models. Unlike global climate models, RCMs simulate climatic characteristics at a fine scale; the boundaries are clearly defined and include realistic simulations of orographic effects (Plummer et al. 2006).

In agreement with Balshi et al. (2009), our results show that fire can occur only if particular weather conditions are reached (drought and wind speed through DC and DSR). For the whole study area, we found that decadal averages of fire occurrence increase significantly over a tipping point of DSR equaling about 0.92 and 1.02 units for all fires and medium fires (Eqs. 2b and 3d), respectively, and of DC equaling 125 units for larger fires (Eq. 4c). A strong statistical relationship between seasonal DC and annual large fire occurrences was previously found by Girardin and Mudelsee (2008). Also, the importance of DSR as a predictor of Canadian area burned has previously been highlighted (e.g., Flannigan et al. 2005, Balshi et al. 2009). Models developed by Wotton et al. (2010) to explain lightning-ignited fires in Canadian ecoregions selected DC, DMC, and FFMC as explanatory fire weather variables and did not identify east–west differences in fire occurrence within Quebec. The latter study was slightly different from ours in that it included fires smaller than 1 ha and encompassed different periods (1985–2000). Moreover, the delimitation of fire occurrences along an east to west gradient is coherent with a previously published moisture map (expressed using July DC) presented by Girardin and Wotton (2009), suggesting that moister regions are less prone to fire than dry regions (Hély et al. 2001).

Our study also suggests that north–south distributions of various classes of fire occurrence in Quebec are governed by differences in tree composition. Generally, fire control is more effective in the southern part of the province due to the ease of detection and accessibility. However, fire suppression should not affect the analysis of fire size classes $\geq$10 ha and $\geq$200 ha because when a large fire occurs, weather conditions are extreme and the human capacity to control a fire is reduced. Fires of $>$3 ha (Arienti et al. 2006) or 4 ha (Podur and Wotton 2010) are attributed to escaped fires. Their distribution can also be assigned to tree composition change. Previous

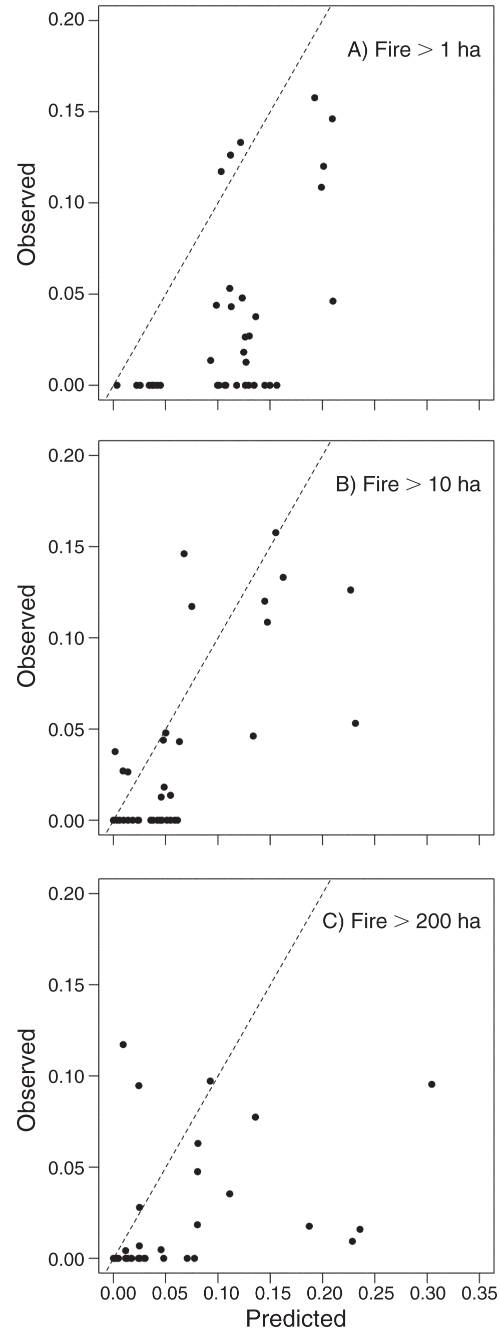

Fig. 3. Observations vs. MARS (multivariate adaptative regression splines) model predictions from 2001 to 2009 of (A) all fire occurrences (fire $\geq$ 1 ha), (B) medium fires (fire $\geq$ 10 ha), and (C) large fires (fire $\geq$ 200 ha).

comparisons between coniferous stands and deciduous or mixed stands in boreal forests highlighted the importance of tree composition in fire regimes (Hély et al. 2001, 2010). Lower fire activity in deciduous-dominated stands and landscapes has already been documented (Quinby 1987, Päätalo 1998, Campbell and Flannigan 2000, Hély et al. 2001, 2010, Lefort et al.

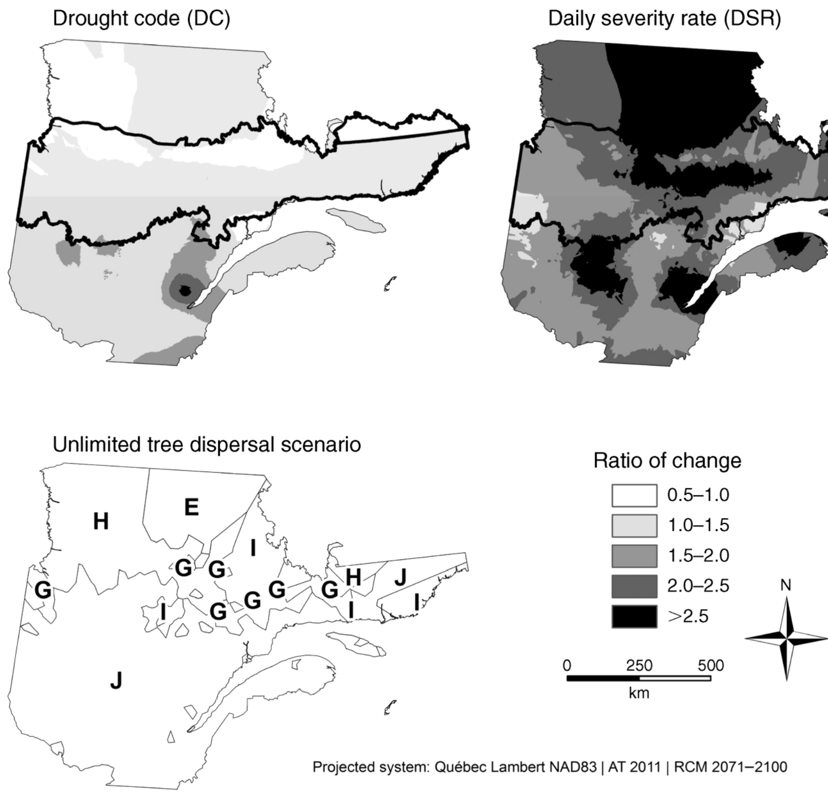

FIG. 4. Maps of changing drought code (DC), daily severity rating (DSR), and unlimited tree dispersal projected with the regional climate model (RCM, scenario A2) for the 2071–2100 period. DC and DSR changes are expressed as ratios of change ([30-year average 2071–2100]/[30-year average 1971–2000]). The unlimited tree dispersal scenario represents the dispersion of tree species clusters.

2004, Arienti et al. 2006, Krawchuk et al. 2006, Drever et al. 2008). These differences come from higher coniferous species flammability in comparison with deciduous species. Coniferous species contain highly flammable oils and resin, and moisture content in the needles is low, while deciduous species have leaves with a higher moisture content that acts as a fire break. Quinby (1987) compared temperate tree species flammability in laboratory experiments. Pine species showed high ignition probability, whereas sugar maple and poplar species showed the lowest probabilities of flammability. Our study highlighted the fire break role of sugar maple forests, largely because the presence of these forests is associated with a reduced frequency of medium and large fires, and because of offsetting effects in FireOcc models of tree composition categories that include sugar maple forests. This study confirmed that weather and tree composition are both important explanatory variables of fire occurrence in boreal forests.

Our analyses are consistent with previous studies indicating that future warming will create climatic conditions more prone to fire occurrence (Girardin and Mudelsee 2008, Amiro et al. 2009, Drever et al. 2009, Flannigan et al. 2009, Le Goff et al. 2009, Wotton et al. 2010). Across our study area, increases in fire occurrence will vary spatially, with the most important changes projected to occur at the eastern and southern limits of the boreal forest. The projected changes (an increase of 10–25% by 2090) are in the range of those predicted in an earlier study by Wotton et al. (2010), with the exception of two regions where the magnitude of change was predicted to be higher by Wotton et al. (2010). That being said, our experiment indicates that the projected increase in fire-conducive weather conditions could be offset by changing tree species distributions. Regions in which this offsetting effect holds true include the western fire bioclimatic zones and the southern limit of the boreal forest. It is important to remember that tree composition changes in this study are governed by climatic, edaphic, and topographic conditions. However, other factors will influence tree migration in addition to these environmental variables. Notably, the fire regime itself is an important factor affecting species distribution (Flannigan and Bergeron 1998, Tremblay et al. 2002, Asselin et al. 2003). For example, fire frequency was a barrier in the past for jack pine expansion (Asselin et al. 2003). Red maple (*Acer rubrum* L.) (Tremblay et al. 2002) and red pine (*Pinus resinosa* Ait.) (Flannigan and Bergeron 1998) are limited to the southern limits of their predicted climatic envelopes because the fire regime prevents these species from spreading farther north. Competition could also

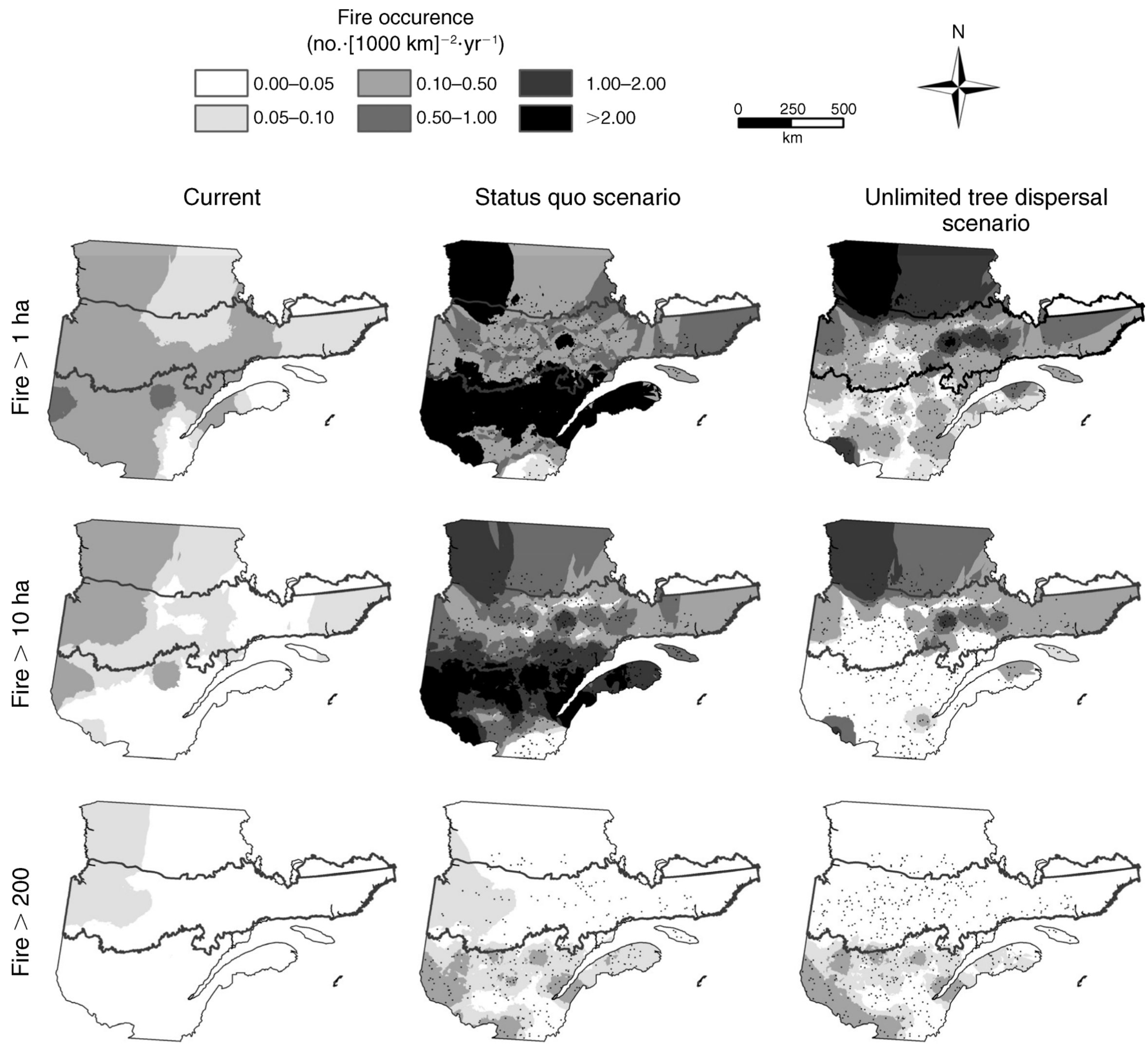

FIG. 5. Map of Quebec showing baseline and 2071–2100 projections (status quo and unlimited dispersal scenarios) for the three classes of fire occurrence (FireOcc $\geq$ 1 ha; $\geq$ 10 ha; $\geq$ 200 ha). Interpretation of the results was limited to the closed-canopy boreal forest owing to biases associated with the application of kriging to northern regions (buffer effect, rarity of data). In the southern part of Quebec, future species composition should be different from the projected one owing to the potential northward migration of southern-limited species not included in our analysis. Points correspond to locations that showed significant differences between current and future projections (Student t tests, $P < 0.05$).

play a role in limiting tree species migration (Engler and Guisan 2009). The unlimited dispersal scenario of tree composition change shows that northward migration of tree composition clusters dominated by southern limited species (e.g., sugar maple, American basswood, and bitternut hickory) offsets the impact of climate change on fire occurrence. However, these species are unlikely to migrate to such high latitudes in the short to medium terms, even though areas of sugar maple can be found north of its specific climatic range, for example in the western boreal forest of Quebec (F. Tremblay, *personal communication*). Further studies are needed, but future fire regimes and climatic conditions could lead to increasing sugar maple abundance at its northern limit. Our results should be used preferentially when building scenarios of future fire occurrence that take into account a potential expansion of deciduous northern zones in boreal forests.

Uncertainties and limitations

Many uncertainties lie in the vegetation data. First, tree composition was represented here by the presence and absence of 10 major tree species; uncertainties could be reduced by using abundance data. Forest type limits do not correspond to an abrupt transition from the presence of a species to an absence; exact species limits may be unknown (Berteaux et al. 2010). The lack of empirical data for the baseline period prevented the use of abundance data. Second, climate change would not imply the sudden appearance of a species in a region where present climatic conditions are not suitable (for example, sugar maple in the boreal forest). Climate change will have an impact on the relative abundance of deciduous compared with coniferous species (Bergeron and Danserau 1993, de Groot et al. 2003, Lecomte et al. 2006) and is also likely to change the understory vegetation. Future projections of fire occurrence should integrate the impacts of vegetation change in forest types as a whole (species abundance, including understory vegetation) rather than the presence/absence of species. On the other hand, the forest as a whole does not constitute a continuous set of forest; it should rather be seen as a fragmented landscape (such as from lakes and human infrastructures) that acts as a firebreak (Parisien et al. 2005). Finally, projections developed in this work assume that the current boreal vegetation distribution is governed only by climatic, edaphic, and topographic factors. However, the climatic envelope observed does not necessarily translate into a potential climatic envelope (Morin and Thuiller 2009). Wildfires play a major role in the distribution of species, and vegetation can respond faster to indirect impacts from shifts in fire regimes than to direct climate change effects (Bergeron and Archambault 1993, Weber and Flannigan 1997). It has been suggested that the combined use of process-based models, including feedback effects of fire activity, in addition to niche-based models could reduce uncertainties related to species distribution (Morin and Thuiller 2009).

The characterization of fires also brings uncertainties related to random effects on fire distribution. Even if the conditions are favorable to fire, the ignition source (lightning, human) was not included in our predictive models. A prediction may therefore be incorrect if a significant change takes place in the frequency of fire occurrences (Hessl 2011), particularly in connection with the increased use of forestland by humans. Other uncertainties are related to human control. Climate change may exceed our ability to control fires (Podur and Wotton 2010); however, this control ability influences fire size (Martell and Sun 2008). Improvements and a better knowledge could make it possible to better control fire and reduce burned areas in the future.

Finally, future projections are always associated with uncertainties because of the chaotic nature of climatic systems (Rind 1999) and future anthropogenic greenhouse gas emissions. The Intergovernmental Panel on Climate Change (IPCC 2007) recommends the use of multiple climate models and emission scenarios in a context of climate change impact assessment (IPCC 2000). Only one climate model (CRM4.1.1) and one climate change scenario (IPCC A2) were used for this study. The RCM runs used in this study were driven with atmospheric and oceanic data from the coupled Canadian General Circulation Model version 2 (CGCM2; Flato and Boer 2001). Balshi et al. (2009) showed that the Canadian General Circulation Model ranks among the best IPCC models with respect to the level of predictability at high northern latitudes. On the other hand, the A2 scenario is at the higher end of other SRES emission scenarios (Nakićenović et al. 2000); while it is not the highest, it is quite realistic in terms of greenhouse gas emissions estimate (Raupach et al. 2007). Within an ensemble of 19 GCM experimentations, the CGCM3 A2 ranks third in terms of level of increase in seasonal drought severity for western boreal Quebec from the late 20th to the late 21st century (Appendix). From an impact and adaptation point of view, if adaptation to a larger climate change is possible, then adaptation to the smaller climate changes at the lower end is also possible (NARCCAP 2007). Finally, a model correction (delta method) was used to reduce bias in modeled climate data. Although this method is considered a very robust method (Déqué 2007), it has the disadvantage of constraining the same frequency and magnitude of extreme weather events relative to the mean climate throughout all periods under study. Nevertheless, a sensitivity analysis on DC projections in which data were treated using different correction methods showed results similar to those reported in this study (T. Logan, *personal communication*). Future projections should use an ensemble approach with multiple corrections, models, and scenarios.

Conclusion

The potential influence of changing forest composition on the impacts of climate change on fire activity in Quebec was examined. Both climate and forest compositions were important factors explaining the distribution of fire occurrences in this province. Each factor had its own relative importance with regard to fire size classes, with large fires being more influenced by forest composition. These results have important implications for fire management in a context of climate change adaptation. In fact, our results indicate that climate change will increase fire occurrence in boreal forests (Girardin and Mudelsee 2008, Amiro et al. 2009, Drever et al. 2009, Flannigan et al. 2009, Le Goff et al. 2009, Wotton et al. 2010). A change in tree composition toward an increasing deciduous component has the potential to significantly offset the impact of increased fire risk in many areas, particularly in areas affected by fires that are difficult to control ($\geq$200 ha). These results suggest that the presence of deciduous species, and more specifically of forest types dominated by temperate species, should be promoted in fire management strategies that attempt to reduce communities' long-term vulnerability to climate change in eastern Canadian boreal forests. Given the uncertainties associated with the various assumptions inherent to the use of ecological niche models, these results should be seen as first estimates of the impacts of changing tree distributions on boreal wildfires in the context of global warming. Future studies should include feedback effects of fire on vegetation distribution and an ensemble modeling approach that integrates several anthropogenic gas emission scenarios and models. Notably, there is potential for expanding this study to the scale of the North American boreal forest using recently mapped tree distribution projections simulated using an ensemble approach of general circulation models (McKenney et al. 2011). The spatial resolution may be coarser, but analysis could gain robustness.

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Supplemental Material

Appendix

Climate model and emission scenario ranks based on the ratio of mean seasonal drought code index of 2071–2100 to 1961–1990 (*Ecological Archives* A023-002-A1).