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Disturbance legacies and paludification mediate the ecological impact of an intensifying wildfire regime in the Clay Belt boreal forest of eastern North America

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Boreal forest dynamics; Burned area; CanFIRE model; Climate change; Depth of burn; Open black spruce–*Sphagnum* forests

Nomenclature
Departmental SIFORT database (Pelletier et al. 1996)

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Abstract

Questions: High moisture levels and low occurrences of wildfires have contributed during recent millennia to the accumulation of thick layers of organic soil and to a succession into open black spruce (*Picea mariana*)–*Sphagnum*-dominated forests in the Clay Belt boreal landscapes of eastern North America. In these forests, the anticipated increase in drought frequency with climate change could lead to a shift in forest structure and composition via increased fire disturbance. Here, we quantify the expected changes in fire behaviour, biomass burning and vegetation composition in the Clay Belt forest of North America that could arise under climate change over the next century.

Location: A managed forest unit in the Clay Belt boreal forest of eastern North America.

Methods: The impact of a changing climate from 1971 to 2100 on fire regime characteristics (i.e. rate of spread, fuel consumption, fire intensity, type of fire and depth of burn) and vegetation dynamics (mortality and recruitment) was investigated using the Canadian Fire Effects Model (CanFIRE). Vegetation dynamics were governed by the fire danger and behaviour that affect tree mortality and post-fire recruitment of species, and by long-term successional pathways that are driven by post-fire recruitment and forest age. An ensemble of two climate models forced by three scenarios of greenhouse gas emissions was used to drive CanFIRE simulations.

Results: Results from multiple scenarios suggested that fire danger will increase significantly during the 21st century in the Clay Belt forest. The burn rate was projected to change from 4.2% decade^{-1} during 1971–2000 to 18.6% decade^{-1} during 2071–2100. Stand mortality, fire intensity and areas affected by crown fires were also projected to increase. A shift in forest composition did not occur over the simulation period across most of our fire regime scenarios. Dominance of open black spruce–*Sphagnum* forests was projected to remain in future landscapes.

Conclusions: Moist and cool conditions in these forests prevent high depth of burn and contribute to the ecological resistance of these forests to increasing fire danger.

Introduction

In the context of global warming (IPCC 2013), boreal forest vulnerability to climate change is a major concern. High-latitude boreal regions are projected to be amongst the most affected by climatic changes (IPCC 2013). There, warming is projected to result in more frequent and severe drought conditions by the end of the 21st century (Yang

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et al. 2011) that will directly impact wildfire activity (e.g. Flannigan et al. 2005; Bergeron et al. 2010; Wotton et al. 2010; Turetsky et al. 2011; Boulanger et al. 2013). Shifts in landscape forest cover and structure are expected in response to these changes (Weber & Flannigan 1997; de Groot et al. 2003; Johnstone et al. 2010b; Barrett et al. 2011), altering wildlife habitat diversity (Weber & Flannigan 1997; Rupp et al. 2006) and reducing the carbon stocks (Harden et al. 2000; Amiro et al. 2009; Turetsky et al. 2011; de Groot et al. 2013). These changes may also result in economic damage for the forest sector, which may include the reduction of commercial products and timber supplies (Weber & Flannigan 1997; Kirilenko & Sedjo 2007; Gauthier et al. in press).

The boreal forest of the Clay Belt region in eastern North America, at the border of the provinces of Quebec and Ontario (Fig. 1), constitutes one of the world's largest carbon stocks estimated at 201–250 $t{\cdot}ha^{-1}$ with a large proportion of forests reaching values up to 1050 $t{\cdot}ha^{-1}$ (Scharlemann et al. 2009). The physiographic unit of the Clay Belt (Vincent & Hardy 1977) originated from the retreat of the proglacial Lake Barlow-Ojibway, which left a thick deposit of clay (Fig. 1). Poor drainage conditions induced by the presence of an impermeable clay substrate, a flat topography and a cold climate facilitated the accumulation of thick layers of organic soil, a process often described as paludification (Fenton et al. 2005; Lavoie et al. 2005b). In parts of the region, peat mosses accumulate on initially mesic soils independently of topography or drainage, and are primarily related to forest succession (Simard et al. 2007). Once *Sphagnum* species increase on the forest floor, fluctuations in water saturation of the organic layer decrease (Bergeron & Fenton 2012). The water table moves from the mineral soil into the organic forest layer, and organic layer depth becomes the dominant factor explaining the water table position (Fenton et al. 2006). Additionally, tree roots are unable to reach the mineral soil, inducing humid, colder and less nutrient-rich environments that result in a drop in tree productivity (Payette 2001). Therefore, in the prolonged absence of fire, forests tend to converge to open or partially less productive spruce–*Sphagnum* forests regardless of the initial species composition (Harper et al. 2003; Lecomte et al. 2006a; Fenton et al. 2007; Simard et al. 2007; Lafleur et al. 2010; Belleau et al. 2011). A decrease in the fire activity in southeastern North American boreal forests during the past three millennia in association with an orbitally-driven climatic cooling (Carcaillet et al. 2001; Girardin et al. 2013a,b) likely contributed to an acceleration of peat accumulation in the Clay Belt forest and a decrease in its productivity (Simard et al. 2007; Girardin et al. 2011).

Burned area and residual organic layers (i.e. layers not consumed by the fire) jointly control forest structure and composition (Lecomte et al. 2006b). Shallow residual organic layers on the ground lead to the establishment of dense forests composed of black spruce (*Picea mariana* (Mill.) BSP), trembling aspen ((*Populus tremuloides* Michx.), white birch (*Betula papyrifera* [Marsh.]) or jack pine (*Pinus banksiana* Lamb.) on mesic sites (Lecomte et al. 2005, 2006b; Johnstone & Chapin 2006; Greene et al. 2007; Johnstone et al. 2010a). In contrast, thick residual organic layers favour black spruce self-replacement (Van Cleve et al. 1983; Johnstone et al. 2010a) and accelerate the process of paludification (Lecomte et al. 2006b; Simard et al.

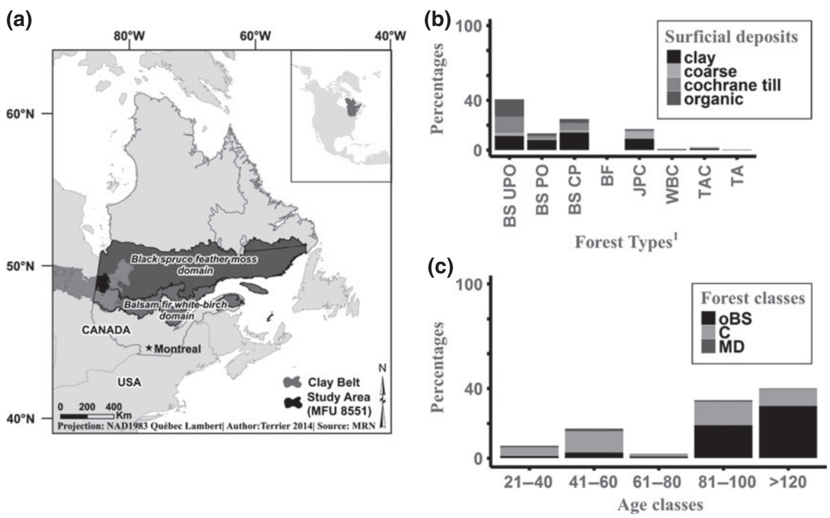

Fig. 1. **(a)** Study area in the Clay Belt boreal forest of eastern North America. **(b)** Relative percentages of cells grouped in different forest types according to the dominant surficial deposits. Forest types and abbreviations are described in Table 1. **(c)** Relative percentages of cells grouped in age classes according to the forest classes (oBS, open black spruce forests; C, coniferous forests; MD, mixed and deciduous forests).

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2009). As burned areas may be positively related to depth of burns in boreal forests (Turetsky et al. 2011), increasing fire activity over the next century under global warming could contribute to the removal of the organic layer, thereby promoting a decrease in black spruce–*Sphagnum*-dominated forests and a shift to closed coniferous and deciduous forests. Although black spruce–*Sphagnum*-dominated forests tend to be moister and resistant to high depth of burn (Terrier et al. 2014), paleoecological analyses of peat sediments indicated that such properties do not necessarily prevent the organic layer from burning (Cyr et al. 2005; Simard et al. 2007). There is therefore justification for an assessment of the risks posed by future climate changes specifically in the Clay Belt forest. The purpose of this paper is to quantify the expected changes in fire behaviour, biomass burning and vegetation dynamics in the Clay Belt forest of North America over the next century that could arise under climatic change.

Study area

The study area is a managed forest unit (MFU 8551; 49°00′–50°00′ N, 78°30′–79°50′ W) in northwestern boreal Quebec, Canada, covering ~ 400,000 ha (Fig. 1). The topography is flat; however, small rocky hills are present. The area lies mostly in the black spruce–moss bioclimatic domain, with the southern part lying in the balsam fir–white birch domain (Saucier et al. 1998). Landscapes are dominated by black spruce forests with the prominent presence of jack pine, trembling aspen, white birch and balsam fir forests on various surficial deposits from coarse till to clay (Harper et al. 2003; Fig. 1b). As of year 2000, 48% of the area was occupied by open black spruce forests (Ministère des Forêts, Faune et Parcs (MFFP) 2013). The climate is sub-polar and sub-humid continental, characterized by long, harsh and dry winters and short, hot and humid summers (Environment Canada 2013). The average annual temperature from 1970 to 2009 was 0.3 °C, ranging from monthly means of – 22.2 to 17.4 °C; the mean total annual precipitation was 862 mm (Environment Canada 2013).

Methods

The impact of a changing climate on fire behaviour and vegetation in the study area was investigated using the Canadian Fire Effects Model (CanFIRE, formerly the Boreal Fire Effects Model, BORFIRE; de Groot et al. 2002, 2003, 2007; de Groot 2006). CanFIRE is a collection of Canadian fire behaviour models that are used to estimate first-order fire effects on physical characteristics, and to estimate ecological effects, at the stand level. Feedback and interactive effects between fire characteristics and vegetation are considered in the CanFIRE model, which makes it an interesting tool for addressing our research question (Flannigan et al. 2001; Hély et al. 2001; Keane et al. 2004, 2013; Krawchuk et al. 2009; Hessl 2011). Fire danger is therein described using the Canadian Forest Fire Weather Index (FWI) System (van Wagner 1987) and the type of fuel available. The vegetation dynamics are governed by fire behaviour (fire rate of spread, fuel consumption, intensity, type of fire-crowning or surface fire and depth of burn), which affects mortality and post-fire recruitment of species based on fire ecology traits. Long-term successional pathways are driven by potential recruitment and natural mortality induced by intra- and interspecific competition (de Groot et al. 2003).

Several tasks were required before having the CanFIRE model applied to the Clay Belt boreal forest. These are illustrated in Fig. 2. Essentially they involved three undertakings: (i) the construction of driving data sets for the CanFIRE model, (2) the determination of recruitment and successional rules, and (3) the simulation of fire-driven vegetation changes. In the following section, we describe our approaches to these tasks.

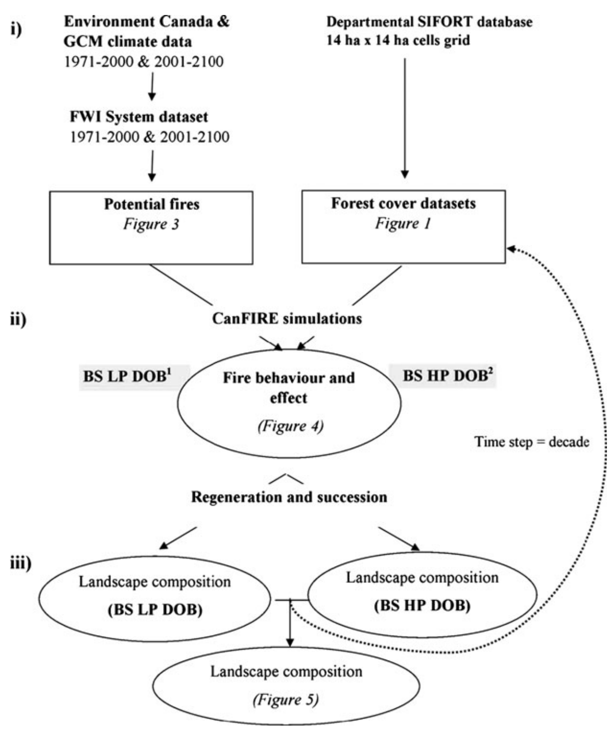

Fig. 2. Tasks conducted in this study to simulate fire regimes and landscape forest composition: **(i)** the construction of driving data sets for the CanFIRE model, **(ii)** the determination of recruitment and successional rules, and **(iii)** the simulation of fire-driven vegetation changes. [1]BS LP DOB: black spruce low potential depth of burn; [2]BS HP DOB: black spruce high potential depth of burn.

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Driving data sets

Forest data sets

For CanFIRE simulation work, we used the SIFORT (Système d'Information FORestière par Tesselle) forest cover data derived from classification of aerial photography and plot-level ground surveys from the first (1970–1980) decadal forest inventories (Pelletier et al. 1996). Data from the third forest inventory (1990–2000) were also used to complete the age class data. The SIFORT forest cover data are represented in raster format at a resolution of 14 ha. Our study area therefore consists of 29527 grid cells. Each grid cell was characterized by the species' cover percentages, forest type and forest age (Table 1, Fig. 1b,c; see supplementary materials for details). Site index (i.e. height at 50 yr old) was also assigned to each cell. It was determined for each species within a grid cell using growth and yield curves compiled by MFFP during their last timber supply inventory (Pelletier et al. 1996). These yield curves were calculated using the model developed by Pothier & Savard (1998). Site index and cover percentages were averaged by species and by each forest type to obtain a single value of species in each forest type for the entire landscape (Table 1).

Table 1. Forest types, forest cover and site index describing the study area (adapted from Belleau et al. 2011). Percentage of species covers and site indices were averaged by forest type.

Forest Cover Type	Name	Code	Species	% Cover	Site Index
Coniferous	Black spruce	BS	–	–	–
	Closed	BS CP	BS	100	14
	Partially open	BS PO	BS	100	12
	Unproductive	BS UPO	BS	100	12
	Balsam fir	BF	BF	100	14
Deciduous	Trembling aspen	TA	TA	100	19
Mixed	Trembling aspen coniferous	TAC	WB	10	16
			TA	30	18
			BS	35	15
			JP	10	16
			BF	15	14
	White birch coniferous	WBC	WB	30	15
			TA	10	18
			BS	25	14
			JP	5	16
			BF	30	14
	Jack pine coniferous	JPC	WB	5	16
			TA	5	18
			BS	45	15
			JP	40	16
			BF	15	14

WB, white birch; TA, trembling aspen; BS, black spruce; JP, jack pine; BF, balsam fir.

The CanFIRE model required quantitative information on the organic layer depth (cm) for the fuel load quantification and for the depth of burn algorithm. For estimation of total organic layer content, we made use of a data set comprising 4217 temporary sample plots (circular 0.04 ha) selected from the third inventory programme of the MFFP (Anyomi et al. 2014). These measurements include a characterization of the forest strata and the site features (e.g. soil texture, depth of B and C horizons, drainage class, surface deposit type, humus type). From these data we deduced the model linking measured organic layer depth (OL depth) within each plot with stand age, forest type and surficial deposit (more details in supplementary materials). In parallel, the depth of the fibric layer (cm) was required for the black spruce to aspen post-fire recruitment rules (see *Recruitment and successional rules*). Herein, we deduced fibric layer depth (cm) using a linear functional relationship to the total organic layer depth (more details in supplementary materials). This relationship was determined using measurements of fibric and total organic layer depths sampled in sites that originated from one recent fire (1997) and eight older fires (1725–1880) (Leroy, pers. comm.). Characterization of the fibric layer was based on the Von Post scale (Canada Soil Survey Committee 1978).

Finally, forest floor fuel loads (litter, moss, lichen, duff fermentation, duff humus) and dead woody debris loads (coarse and medium; kg·m^{-2}) were required to describe biomass in the CanFIRE model. Forest floor and dead and downed woody fuel load information were estimated using Paquette's equations (2011). Therein standard fuel (litter, moss, lichen, total duff) inventories were established in sites originating from different fire years and for four successional pathways (trembling aspen, jack pine, severe black spruce, non-severe black spruce). Total duff was separated into duff-fibric and duff-humus using fibric layer equations (more details in supplementary materials). Open black spruce stands <100 yr old were considered as stands originating from non-severe fires; black spruce stands 100 yr old and over were considered as originating from severe fires for the start of our simulation (Belleau et al. 2011). We used trembling aspen fuel loads for all mixed wood stands, including white birch–coniferous stands, and productive black spruce fuel loads for balsam fir stands.

Forest fire data

In this simulation work, the disturbance rate is guided by a random generator of fire sizes. To obtain an approximation of the size and frequency of fires in our region, we used fire data from the MFFP. The database contains information on the location, the date of detection, the size (ha) and the

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cause (lightning or human) of all fires recorded in the province of Quebec from 1971 to 2011. The data set is provided in point-based format and as polygons. Fires that occurred in the eastern part of the black spruce–moss and balsam fir–white birch bioclimatic domains were extracted and used to generate the fire events distribution needed for the random generator. A total of 613 fires occurred between 1971 and 2000, and 97% of the total area was burned by fires larger than 1000 ha.

Fire danger data

Daily FWI System component values were calculated for every fire starting point encountered during the interval 1971–2000 using BioSIM software (Régnière & Bolstad 1994). The FWI System is used in Canada to evaluate the severity of fire weather conditions by providing indices of fuel moisture and fire behaviour (Wotton 2009). All indices are unitless, with the zero value indicating low fire danger and high values indicating high fire danger. As part of the procedure, daily weather data were interpolated from Environment Canada's historical climate database (Environment Canada 2013) from the four closest weather stations, adjusted for differences in latitude, longitude and elevation between the data sources and stand location, and averaged using a $1/d^2$ weight, where d is distance. Winter precipitation was included in the algorithms to correct the onset and end of the fire season using snow accumulation (Terrier et al. 2013). Finally, the Daily Severity Rating (DSR) was computed. The DSR is a numerical rating of the difficulty of controlling fires.

To determine future fire danger at fire starting points, we used a procedure similar to that described by Flannigan et al. (2013) and de Groot et al. (2013). We used monthly temperature, precipitation, relative humidity and wind speed collected from two General Climate Models (GCMs), namely the Canadian CGCM3.1 (Scinocca et al. 2008) and the United Kingdom HadCM3 (Gordon et al. 2000). Simulations were performed using the IPCC A2, A1B and B1 Special Report on Emission Scenarios (Nakicenovic et al. 2000). In total, our work includes six GCM simulation experiments. CGCM3.1 and HadCM3 were selected among other GCMs to include a wide spectrum of climate change possibilities in our study area. The GCM data consist of simulated climate data by cells whose size depends on the model resolution. Here GCM data were collected from four to eight GCM cells, depending on model resolution, and averaged for the whole study area. Monthly weather changes between the GCM baseline data and future GCM data was summarized by decade until 2100. These monthly changes were then added (for temperature, relative humidity and wind speed) or multiplied (for precipitation) to Environment Canada's historical daily data for each decade. Generally, increases in temperature and precipitation were projected with both GCMs for the periods 2031–2060 and 2071–2100 in comparison with the reference period (1971–2000; Table S1). For the period 2031–2060, the HadCM3 model under the A2 storyline projected the highest temperature increase (+3.1 °C) and the lowest increase in precipitation (+13%). A slight decrease in temperature (−0.9 °C) and increase in precipitation (+13%) were projected with the CGCM3.1 under the B1 storyline (Table S1). Fewer changes were projected for the period 2031–2060.

Simulated fires and burnable cells

Projections of potential fire occurrences in the Clay Belt forest were done using published empirical models (Terrier et al. 2013) describing the decadal distribution of wildfire occurrences for a given class (≥1 ha, ≥10 ha, ≥200 ha) in eastern Canada as a function of a set of fire bioclimatic zones determined from fire weather (*FW*) components and tree species composition (*TreeComp*). The model for fires of size class >10 ha was selected for this study for its high predictive capabilities (Terrier et al. 2013) and to ensure that approximately the entire grid cell (14 ha) would be burned during a fire event. Using these equations, we calculated the total number of fires that were encountered during each decade of simulation (1971–2100), and this was repeated for each GCM experiment. The fire sizes, dates-of-ignition (day of year) and associated fire danger were randomly selected from the fire events distribution discussed earlier. Fires were prescribed to burn the first year of each decade under analysis (e.g. 1971–1980, fires occurred in 1971). The total area burned during a decade was calculated, and then cells were randomly selected until this total area burned was reached. From this, we obtained a large range of fire characteristics (fire occurrences, total burned areas, dates-of-ignition, types of fuel, etc.) with several fire impact scenarios (e.g. fire depth of burn, species turnover, etc.). This modelling approach of 'non-contiguous burned cells' was validated by comparing predicted burned areas from 1971 to 2000 with historical fire data from the MFFP.

Vegetation feedback on total burned area was finally evaluated by calculating the percentages of forest types in cells that underwent tree mortality (>60% of tree mortality) (see details in supplementary materials). To further evaluate the impacts of disturbance legacies and succession on our results, a no-fire scenario was included in our Can-FIRE model experiments. Therein, fires occurring from 1971 to 2000 were prescribed as in other simulations; from 2001 to 2100 it was run without fire.

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Modifications to the CanFIRE model

Depth of burn

In our study area, high soil moisture conditions in black spruce forest dominated by *Sphagnum* spp. imply lower depth of burn in comparison with other regions of the boreal forest (Terrier et al. 2014). Therefore, depth of burn algorithms in CanFIRE needed to be modified to reflect the regional specificity. We herein include depth of burn equations of Terrier et al. (2014) obtained by model calibration in black spruce–*Sphagnum* spp.-dominated stands. Therein the moisture content of the organic layer was predicted using the DC component of the FWI System, depth of the organic layer (cm) and site conditions (dominated by *Sphagnum* species or dominated by feather mosses). In accordance with Lafleur et al. (2010) and Fenton et al. (2007), site conditions were determined using the depth of the organic layer: the criteria for the switch from black spruce–feather mosses to black spruce–*Sphagnum* stands was set at ≤30 cm and >30 cm, respectively. Depth of burn was defined as depth where extreme gravimetric moisture contents of 140% and 500% were reached. The 140% limit represents an extremely low potential (LP) depth of burn; the 500% limit expresses an extreme high potential (HP) depth of burn. LP and HP scenarios were intended to capture the inherent variability in burning potential attributed to heterogeneity in the organic bulk density (Benscoter et al. 2011). Simulations were conducted separately for each of these scenarios of black spruce (BS) potential depth of burn.

Post-fire recruitment and successional rules

Post-fire recruitment in the CanFIRE model is based on tree species' adaptation to the fire environment (i.e. fire ecology using a plant functional type or PFT approach). In the absence of fire, vegetation dynamics are governed by recruitment of annual (or non-disturbance) seedling species and natural mortality due to competition (de Groot et al. 2003). However, tree post-fire recruitment and forest successional pathways in the Clay Belt forest are also affected by the total organic layer depth and surficial deposits (Lavoie et al. 2005a; Lecomte et al. 2006a; Lecomte & Bergeron 2005; Johnstone & Chapin 2006; Greene et al. 2007; Simard et al. 2009; Belleau et al. 2011). Here we modified the CanFIRE simulation setup by applying external post-fire recruitment and succession algorithms. Post-fire recruitment was based on pre-fire forest type, post-fire residual organic layer depth and composition of neighbouring grid cells. Post-fire tree recuitment was also made possible only when tree mortality occurred during the fire. A severe depth of burn was defined as a fire that burned sufficient organic material to generate productive stands of coniferous and hardwood species. These conditions were achieved when residual organic layers were less than 3 cm (Greene et al. 2007), 20 cm (Simard et al. 2009) and 6 cm (Moore & Wein 1977) in jack pine, black spruce and white birch stand types, respectively. For post-fire recruitment in trembling aspen, complete consumption of the fibric layer was required (de Groot et al. 2003). A severe depth of burn occurring in a pure black spruce stand was allowed to induce a shift in forest composition toward jack pine or hardwood stand types when neighbouring cells were dominated by these forest types. The neighbouring analysis was performed with the *k*-nearest neighbouring (*k*nn) function of the 'class' package (Venables & Ripley 2002) included in the R freeware (R Foundation for Statistical Computing, Vienna, AT). In this procedure, neighbouring forest types were assigned to each grid cell by selecting forest types that occupied the majority of neighbouring cells. The algorithm computed distance between points and returned the most frequent forest types in the *k* nearest neighbours. The classification is determined by the majority composition. No change in composition occurred after a non-severe fire in black spruce forests. Once recruitment was completed, cell age was changed to 1 yr and a new forest type was attributed to the recruited grid cells. New organic layer depths were calculated by subtracting the depth of burn to pre-fire organic layer depths and we used depth of burn information to determine fire severity. Fires on the Clay Belt generally involve high mortality and thus are stand replacing (Bergeron et al. 2004). A validation analysis verified beforehand that the CanFIRE model yielded reasonable stand mortality and recruitment outcomes by comparing the modelled mortality rates with the vegetation structure of the third SIFORT inventories (see supplementary materials for details).

To include successional and structural changes with the time since fire (Lecomte & Bergeron 2005; Belleau et al. 2011; Bergeron & Fenton 2012), we applied the successional pathways of Belleau et al. (2011) to our cells. Belleau et al. (2011) estimated for each stand type and surface deposit type the transition age, transition rate and probability of transited stands (their Table III). For each decade, we selected a percentage of cells, expressed by the transition rates, among cells that reached transition age. The transition rate reflects the fact that replacement of species is not synchronous with forest transition: not all forests transit at the same time even if the age criterion is reached. The probability of transited stands given by the transition matrix provided information on the forest types affected by changes and on the transition proportions. Once succession was completed, forest attributes (organic layer depths,

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species age) were updated using age class and forest composition as described in the forest data sets section.

Data analysis

Simulated fire weather conditions, fire regimes and behaviour, and landscape forest type compositions were analysed as follows. Fire weather conditions were summarized for the periods 1971–2000, 2031–2060 and 2071–2100. FWI System components were averaged by GCM and IPCC scenarios. The significance of increases in FWI System components was tested using the two-sample Student's *t*-test (one-sided) comparing the 1971–2000 with 2031–2060 and 2071–2100 intervals (n = 6; (3 + 3) decades per analysis). Student's *t*-test was performed in the R freeware (v 3.1). Decadal percentages of burned cells, percentages of cells affected by severe depth of burn, mean fire intensities ($kW{\cdot}m^{-1}$) and the percentages of burned cells affected by crown fires provided information on fire regime and behaviour changes. Landscape composition was described by calculating the decadal relative percentage of cells grouped into three forest type categories: open black spruce, coniferous and mixed/deciduous forest types. Open black spruce forest types included open and partially open black spruce forest types. This class reflected late successional black spruce (*Picea mariana*)–*Sphagnum*-dominated forests. Productive black spruce, jack pine and balsam fir forest types were grouped into coniferous forest types. Mixed and pure deciduous forest types were considered as one group because of their low relative proportions in the study area.

Fire activity and landscape composition results of all experiments and black spruce (BS) potential depth of burn scenarios were summarized using visually-weighted regressions (Hsiang et al. 2013). The analysis was performed in the R freeware (v 3.1) using the vwREg function. The visually-weighted regressions procedure consists of computing 1000 bootstrap smoothed regressions from the original sample points, computing a density estimate for each decade through the bootstrapped regressions, and illustrating the uncertainty using gradient-shaded confidence intervals (darker colour for higher density). Sample points in our case corresponded to a total of 12 values per decade (six experiments × two BS potential depth of burn scenarios + one no-fire scenario for landscape composition). Regressions were conducted using time (decade) as a predictor variable.

Results

Future fire weather conditions

Table 2 illustrates current (1971–2000) and future (2031–2060, 2071–2100) FWI System components projected using two GCMs (Canadian CGCM3.1, Hadley HadCM3) and three IPCC scenarios (A1B, A2, B1). For the current period, the Drought Code (DC), the Fire Weather Index (FWI) and the Daily Severity Rating (DSR) components equalled 113, 4.1 and 0.68 units, respectively. These values

Table 2. Comparison of empirical current (1975–2000) and future projected (2031–2060, 2071–2100) means of the Canadian Forest Fire Weather Index (FWIs) System components for the study area. Future indices were projected using two global climate models (GCM) (Canadian CGMCM3.1, Hardley HadCM3) and three IPCC scenarios (A1B, A2, B1). FWI System components that showed significant increases between current levels and future projections are denoted by an asterisk (Student's *t*-tests, $P \leq 0.05$). All FWI System component values are unitless.

		1975–2000	2031–2060			2071–2100		
			A1B	A2	B1	A1B	A2	B1
FFMC	*CGCM3.1*	68.9	69.3	68.8	69.7*	69.6	69.6	69.1
	HadCM3		71.1**	71*	71.3**	73.5**	74.3***	71.6**
DMC	*CGCM3.1*	10.1	10.1	9.9	10.5***	10.6	10.7	10.1
	HadCM3		11.4**	11.4	11.4***	13.3**	14.7**	11.8**
DC	*CGCM3.1*	113	108	106	124*	116	123	114
	HadCM3		110	116	118	127	161*	119*
ISI	*CGCM3.1*	2.8	2.9*	2.9*	2.9	3	3.1	2.9
	HadCM3		3.3	3.3	3.3*	3.9**	4.1**	3.4*
BUI	*CGCM3.1*	14.6	14.4**	14	15.3*	15	15.3	14.5
	HadCM3		16.1	16.1	16.1**	18.5*	21*	16.6**
FWI	*CGCM3.1*	4.1	4.2**	4	4.3	4.4	4.6	4.1
	HadCM3		5	4.9	4.9*	6.12**	6.9*	5.1**
DSR	*CGCM3.1*	0.68	0.7**	0.72	0.72	0.78	0.81	0.69
	HadCM3		0.93***	0.89	0.89*	1.3**	1.53*	0.98**

FFMC, Fine Fuel Moisture Content; DMC, Duff Moisture Content; DC, Drought Code; ISI, Initial Spread Index; BUI, Buildup Index; FWI, Fire Weather Index; DSR, Daily Severity Rating.
Student's *t*-test: $*P < 0.05$; $**P < 0.01$; $***P < 0.001$.

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were projected to rise significantly from the middle to end of the 21st century in the HadCM3 simulations and less so in the CGCM3.1 simulations. From 1971–2000 to 2071–2100, the least and most severe case scenarios (upper and lower bounds) are, respectively, the CGCM3.1 B1 and HadCM3 A2 simulations.

Projected number of fires and burned areas

A total of eight fires occurred between the first and the third SIFORT inventories in the study area, burning 13.6% of the area (burn rate equivalent = 4.5% decade^{-1}). Three fires exceeded 200 ha, two of which reached, respectively, 24,168 and 30,876 ha in 1976 and 1997 (Fig. 3b).

For the baseline period from 1971 to 2000, nine fires were predicted from the FireOcc equation with DSR as input (Fig. 3a). The equivalent burn rate (which combined the FireOcc and the random generator of fire sizes) was 4.2% decade^{-1} for the whole period (vs the observed burned rate of 4.5% decade^{-1} reported earlier; Fig. 3b). Decadal fire occurrences were projected to remain constant (three fires decade^{-1}) from 1971–2000 to 2071–2100 in the CGCM3.1 simulations (Fig. 3a). This is explained by the fact that the minimum DSR threshold for a significant increase in FireOcc was never reached in these simulations (Table 2). The opposite was true in the HadCM3 simulations: under the A1B and A2 IPCC scenarios, FireOcc was projected to increase to 162 and 296 fires in 2091–2100. Projected burnable cells were estimated at 73% and 15% decade^{-1}, respectively, for the A1B and A2 scenarios. Of note is that such high values resulted from the extremely high rate of change in the mean DSR over the end of the next century as projected by HadCM3 (Table 2; also see Wotton et al. 2010). While a burned area of 15% decade^{-1} is realistically plausible for the region on the basis of stand-replacing fire history studies (see Girardin et al. 2013a; their Fig. 3a), the same thing cannot be said for the most extreme value (discussed later). It was however kept in our simulation experiment to study the behaviour of the CanFIRE model runs when inducing extreme perturbations. Altogether, the number of burnable cells averaged across all available simulations changed to 18.6% decade^{-1} from 2071 to 2100 (i.e. a four-fold increase relative to the baseline period).

CanFIRE simulated fire regimes and fire effects

Almost all burnable cells were predicted by the CanFIRE model to record fire-induced mortality. The number of cells affected by mortality for all simulations was on average 4.1% decade^{-1} (vs 4.2% decade^{-1} burnable cells) for the whole 1971–2000 period and changed to 17.9% decade^{-1} (vs 18.6% decade^{-1} burnable cells) for 2071–2100. Differences resulted from cells that were predicted to escape fire. Trends in the percentage of cells affected by mortality were projected to remain relatively constant from 1971 to 2060 and then increase for the remainder of the 21st century (Fig. 4a, white line). Greater dispersion of projections (lighter red colour) was observed during the 2061–2100 period, reflecting variability in the HADCM3 and CGCM3.1 runs. Fire intensity and percentage of cells affected by crown fire were also projected to increase. Fire intensities averaged 300 kW·m^{-1} in 1971–2000 and increased up to an average of 1200 kW·m^{-1} at the end of the century (Fig. 4c). Again, a high dispersion of projections occurred toward the end of the century. As for the average percentage of cells affected by crown fire, it increased from 5% to 15% decade^{-1} from the start to the

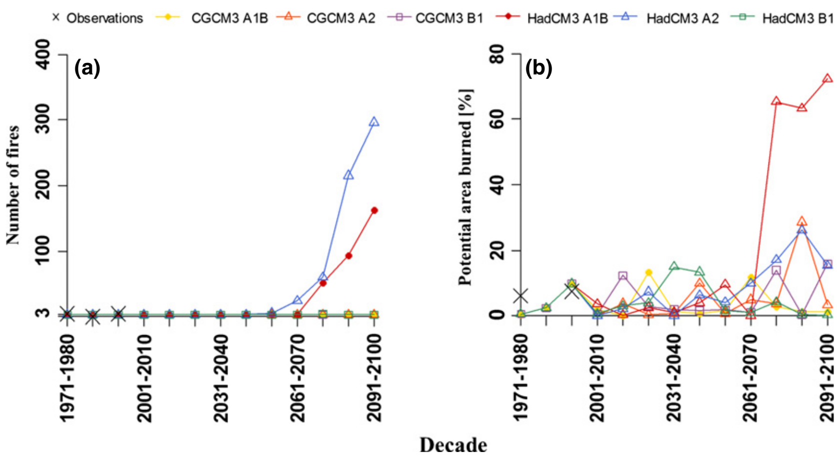

Fig. 3. Observed and projected decadal (**a**) number of fires and (**b**) percentage of burnable cells from 1970 to 2100. Current number of fires (1971–2000) was predicted using Environment Canada's historical climate database (Environment Canada 2013). Future number of fires (2001–2100) was projected using weather data from the CGCM3.1 and HadCM3 global climate models with forcing scenarios A1B, A2 and B1.

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end of the simulations (Fig. 4d). In contrast, the percentage of cells affected by severe depth of burn decreased from around 50–30% decade^{-1} during the 1971–2030 period, then stabilized at ~30% decade^{-1}, and remained at that level until 2100 (Fig. 4b). Exclusion of the uppermost extreme fire simulation (e.g. HADCM3 runs under the A1B scenario) from computation of the conditional means did not alter our results.

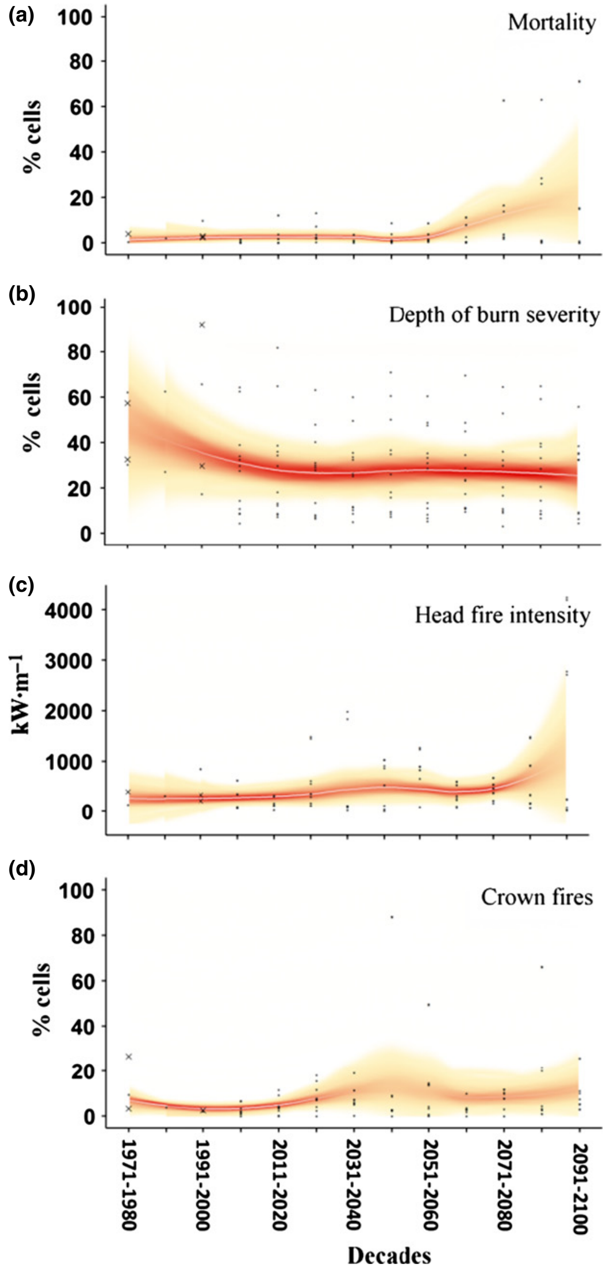

Fig. 4. Fire regime and behaviour simulated using the Canadian Fire Effects (CanFIRE) model. Shown are trends in (**a**) the decadal percentages of cells affected by more than 60% of mortality, (**b**) percentages of cells affected by severe depth of burn, (**c**) head fire intensities [kW·m^{-1}], and (**d**) percentages of cells affected by crown fires. Each dot expresses a GCM experiment for a given decade and black spruce potential depth of burn scenario. Trends are shown using the Visually-Weighted Regression, where colour intensity depicts dot density (darker = lower dot dispersion). The white line in each panel denotes the conditional mean. Crosses express values simulated with observed fires.

Simulated future landscape compositions

Simulated landscape composition was presented as the relative percentage of each forest type (Fig. 5). Simulations with observed fire projected a mean of 57% open black spruce forests (vs 48% estimated by the MFFP). The percentage of open black spruce forests was projected to increase by roughly 30% from 1971–2000 to mid-21st century, while coniferous and mixed/deciduous forests were projected to decrease by 20% and 2%, respectively. The overall composition was then projected to stabilize until 2080. After 2080, the proportion of cells occupied by open black spruce forests was projected to begin on average, a slight decrease. Coniferous percentages showed a negligi-

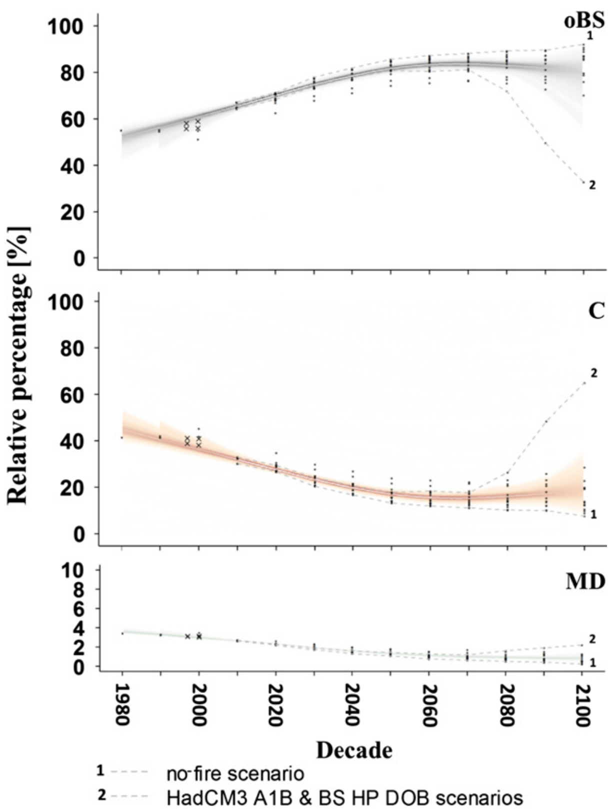

Fig. 5. Relative percentages of each forest class projected from 1971 to 2000. Each dot expresses a GCM experiment for a given decade and black spruce potential depth of burn scenario. Grey lines denote the upper and lower bounds of the multiple simulation runs that are given by the no-fire and extreme HadCM3 A2 scenarios. Crosses express values simulated with observed fires.

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ble increase. The largest change in composition was simulated under the scenario projecting the largest climate change. The no-fire scenario (grey line) likewise showed stabilization from 2050 to 2080, and open black spruce percentages increased slightly from then onwards. In contrast, landscapes recovered high coniferous proportions and an abrupt open black spruce decrease was observed under the extreme HadCM3 simulations (grey line, A1B scenario and BS HP DOB).

Discussion

This study used an ensemble-mean of six GCM experiments and two scenarios of potential black spruce depth of burn to project the impacts of climate change on fire activity and landscape composition in the Clay Belt boreal forest of eastern boreal North America. Succession and fire were incorporated into the modelling setup to simulate interactions between disturbance and vegetation (Keane et al. 2013). Our simulations analysis suggested that depth of burn will decrease from 1971 to 2050 and level off afterward with the rapid increase in the dominance of open black spruce–*Sphagnum* forests. Landscape productivity and composition will be primarily driven by succession until 2100, despite global climate change impacts on fire. This interpretation is supported by the fact that both no-fire and the conditional mean of all fire scenarios yielded similar outcomes in relation to levels of trends in percentages of open black spruce–*Sphagnum*-dominated forests.

We attribute our results to two processes, that is, the influence of disturbance legacies on the future response of the Clay Belt forest to climate change, and the low potential for severe depth of burns in open black spruce–*Sphagnum*-dominated forests. The Clay Belt forest was affected by pulses of major fires during the periods 1830–1850 and 1910–1920 (Bergeron et al. 2004). In future decades, the stands affected by these fires will reach suitable ages for the transition into open black spruce–*Sphagnum*-dominated forests (Lecomte & Bergeron 2005; Lecomte et al. 2006a; Belleau et al. 2011). This transition implies higher soil moisture conditions (Fenton et al. 2006; Simard et al. 2007; Terrier et al. 2014) induced by the water retention characteristics of *Sphagnum* species. Forest soils will consequently be protected from high organic layer consumption during future wildfires (Benscoter & Wieder 2003; Harden et al. 2006; Shetler et al. 2008; Kasischke et al. 2010; Benscoter et al. 2011; Turetsky et al. 2011; Magnan et al. 2012). Depth of burn severity will decrease in the landscape, fires will burn the above-ground biomass, but peat deposits will remain relatively intact (Magnan et al. 2012) and the probability of a return to closed forests will be reduced (Simard et al. 2007; Terrier et al. 2014). A reversal of these phenomena by the end of the 21st century is unlikely according to our experiment, but might be possible under an extreme climate pertubation scenario (i.e. our HadCM3 A1B run).

Our projections of climate change impacts on fire activity in the study area are consistent with earlier assessments. Previous projections were obtained by empirical models describing the relationship between area burned and the variables describing the drying of fuels and fire behaviour processes (Amiro et al. 2009; Balshi et al. 2009; Flannigan et al. 2009). These studies were generally conducted with the assumption that vegetation would remain constant over time. Here, area burned simulations were based on fire occurrence projections and random selection of fire sizes in the forest fire database to meet the CanFIRE model data requirements (date of burning, burned area). Unlike in previous assessments, our simulations included a vegetation feedback mechanism. Although they do not apply to the exact same periods and territories, our burn rate estimate for the reference period (4.2% decade^{-1}, 1971–2000) was within the range estimated by Boulanger et al. (2013; burn rates 2–5% decade^{-1}). As for the fourfold increase in burn rate projected in this study, it is also consistent with projections obtained in previous studies. Bergeron et al. (2010) projected a twofold increase in the burn rate from 1961–1999 to 2081–2100. An increase of 150–200% in the burned areas was projected by Flannigan et al. (2005), and of 286–409% by Boulanger et al. (2014). Note that, in our experiments, cells affected by mortality followed the same trends as found in fire weather data. The same correlation occurred between predicted burnable cells (only climate effects) and cells affected by fire-induced mortality (climate + vegetation effects). Hence, potential vegetation feedback on burned areas can be excluded. In the absence of a changing landscape composition, and particularly without an increase in deciduous proportions arising from forest management, climate will remain the main driver of burned areas in the Clay Belt forest over the next century (Carcaillet et al. 2001; Girardin et al. 2013a).

Accompanying the increase in drought conditions and burned areas were increases in fire intensities and crown fires, as projected in other parts of the Canadian boreal forest (Flannigan et al. 2005, 2009; de Groot et al. 2013). Values in our study area were nonetheless lower. For instance, de Groot et al. (2013) projected values reaching 6047 kW·m^{-1} in the west and 57–69% of crown fires in coniferous forests at the end of the century (against up to 3000 kW·m^{-1} and 15% of crown fire for 2100 in our study). These differences result from higher rainfall amounts in our area (Environment Canada 2013) and milder fire weather conditions. In our simulations, trends in fire intensities and crown fires also followed the overall climatic trends. The projected increase in black

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spruce–*Sphagnum*-dominated forests will unlikely influence these fire characteristics. Despite their low potential fuel consumption, under intense fires the vertical and horizontal fuel structures of open black spruce forests is particularly conducive to intense crown fires. These forests are composed of a high density of small trees (height <3 cm; Paquette 2011), and black spruce layering further implies a fuel vertical continuity from the organic layer to the tree crown. Fire can therein propagate with forest floor litter and herbs and move to the crown (Paquette 2011). The projected increase in fire intensities and crown fires under climate change inevitably will imply a greater difficulty for fire control in these forests in the future (Podur & Wotton 2010; de Groot et al. 2013).

Uncertainties and limitations

The limited temporal coverage of the future climate simulations used here prevented us from projecting fire behaviour, biomass burning and vegetation composition beyond 2100 (IPCC 2013). Our simulation period was probably too short to observe the long-term impact of the increasing burned areas. Changes in landscape structure require one half to two rotations of a new disturbance regime to adjust (Baker 1995). Although a higher proportion of black spruce–*Sphagnum*-dominated forests was projected for the end of the 21st century, a slight decreasing trend was observed starting from 2080. This reduction may be larger after 2100.

We believe that open black spruce forests from the Clay Belt may not be completely protected from fire: a return to productivity was simulated under our extreme climate change scenario HadCM3 A1B. This scenario reflects potential fire effects under extreme drought conditions and high fire intensities. Under such conditions, fire generates particularly high amounts of radiative heat drying ahead of the fire, and fuel can ignite without direct contact with the main front (Cyr et al. 2005). Future simulation experiments should be extended to forthcoming centuries to fully capture these effects.

The modelling setup and assumptions made in this study are not without limitations. Cells occupied by more than 75% of black spruce coverage were rounded to 100%, implying that the coverage of deciduous and mixed forests may be underestimated at the landscape level. A low proportion of deciduous species could favour mixed forest recruitment in pure black spruce forests. Future studies should thus look into more details about the contribution of small patches of deciduous species to the long-term dynamics of coniferous forests. We think, however, that this limitation may not change the projected proportion of coniferous vs deciduous forests at the landscape scale for the end of the 21st century, because this horizon is too short for a reversal of the transition from mixed to coniferous forests.

Direct impacts of climatic change on vegetation were not considered in our simulation work. This includes the potential increase in tree growth with the lengthening of the growing season (Dunn et al. 2007; Grant et al. 2009), decomposition (Wickland & Neff 2008) and northward migration of temperate species on peat accumulation (Iverson & Prasad 1998; McKenney et al. 2007, 2011; O'ishi & Abe-Ouchi 2009; O'ishi et al. 2009; Berteaux et al. 2010). The particularly high organic layer thickness and high soil moisture content may prevent such impacts by limiting tree growth (Simard et al. 2007) and establishment of southern species (Lafleur et al. 2010). Hence, these two factors are not expected to significantly affect our conclusions. However, smoldering combustion can burn for hours to weeks after flaming (Ryan 2002) and decomposition may be accelerated by increasing temperatures. Residual post-fire organic layer depths could therefore be overestimated in our simulations, implying a faster paludification process (an overestimation of the proportion of open black spruce forests) under the low potential depth of burn scenario.

Conclusion

We have evaluated how future increases in wildfire activity might affect the forest structure and composition of the Clay Belt forest in eastern North America through their impact on the depth of burns, mortality and post-fire recruitment. Previous studies indicated that, in boreal forests, increasing depth of burn and/or total area burned will lead to shifts from coniferous-dominated cover to a mixed/deciduous cover (de Groot et al. 2003; Bonan 2008; Johnstone et al. 2010b; Barrett et al. 2011). Dynamic drivers could eventually shift from vegetation to climate-driven fire processes under future warmer and drier climates (Keane et al. 2013). Our results suggest the opposite for the Clay Belt forest, as the proportion of conifers is therein projected to increase. The legacy of past fire pulses implies a rapid succession of closed forests into open black spruce forest in upcoming decades, and moist conditions encountered in these forests will provide a level of resistance to some adverse impacts of the increasing fire activity (Johnstone et al. 2010b; Jafarov et al. 2013; Terrier et al. 2014). That said, future shifts in landscape composition and structure are unlikely to offset climatic effects on fire behaviour (Podur & Wotton 2010; de Groot et al. 2013): increased fire intensity and area burned are likely outcomes of continued warming in the Clay Belt boreal forest. Managers should consider that practices that favour the development of hardwood and productive black spruce forest, or that

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reduce *Sphagnum* establishment (such as summer clear-cut or site preparation), enhance forest productivity, but also increase potential vulnerability to severe depth of burn and decrease long-term carbon storage (Terrier et al. 2014). Vulnerability of these forests to climate change will inevitably depend on current and future forest management practices (Loudermilk et al. 2013). Inclusion of land use in future simulation work should allow guidance for development of forest management practices that minimize climate change impacts.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Supplementary materials.