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Disturbance history affects lightning fire initiation in the mixedwood boreal forest: Observations and simulations

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ABSTRACT

The emulation of natural disturbances such as fire is a prominent harvest management strategy for ecosystems in Canada's boreal forest region, but the effect of harvesting on subsequent lightning fire occurrence has not been studied systematically in the mixedwood boreal forest. We quantified the relationship between annual patterns of lightning fire initiation, forest composition, lightning, and fire weather conditions over eight years (1994–2001) in a 60,000 km^2 region of actively harvested mixedwood boreal forest in western Canada. Our analyses illustrated that forest harvesting and burning had opposite effects on subsequent fire initiation, so harvest was not a surrogate for fire. Fire initiation increased in landscapes with more area harvested and decreased with area recently burned. Our data suggested that increased fire initiation was most pronounced in harvested stands up to a decade old, and there was some evidence that the effect might last as long as 30 years. We then used a dynamic fire-succession simulation model to quantify the long-term effects of these fuel-based relationships using two metrics. As expected, the first metric demonstrated that the number of years between disturbances was significantly less in stands that were harvested and then burned, than those that were burned and then burned again. However, the more revealing component of the simulations was an illustration that despite the strong, positive relationship between harvested areas and fire initiation, the area affected over the long-term by a reduced disturbance interval was relatively small. Accordingly, our study shows that spatiotemporal regulation of lightning fire initiation through harvesting activity results in a systematic accelerated frequency of disturbance that is novel to the mixedwood boreal system, but the area affected by these events amount to local peculiarities rather than broad-scaled regularities.
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1. Introduction

The emulation of natural disturbance regimes as a coarse-scaled strategy for sustainable ecosystem management (Hunter, 1993) is a prominent goal in Canada's boreal forest regions (Paine et al., 1998; McRae et al., 2001; Song, 2002; Armstrong et al., 2003; Bergeron et al., 2007). However, implementation of this strategy is not simple, and there is no clear agreement on its efficacy (Song, 2002; Belle-Isle and Kneeshaw, 2007; Thiffaut et al., 2007). Where fire is the dominant natural disturbance, harvest prescriptions established by forest management companies have been guided by derived indices such as burn rate or frequency, pattern of forest structure produced by fires, and the residual material left post-fire (Cumming, 2001a; Reed, 2001). The rationale being that the spatial and temporal patterns of harvesting should reflect the natural range to which the ecosystem is adapted, so that harvesting acts as a surrogate for wildfire (Hunter, 1993; McRae et al., 2001; Gillson and Willis, 2004; Bergeron et al., 2007).

Stand-replacing fires initiated by lightning strikes are the dominant natural disturbances in Alberta's mixedwood boreal forest region (Alberta Sustainable Resource Development [ASRD], 2004) and similarities of forest harvest to wildfire have been widely studied in the region (Song, 2002). Short-term comparisons between fire and harvest treatments for ecological indicators such as soil properties, forest regeneration, bird communities and landscape structure show that current harvest practices cannot comprehensively emulate fire, but is similar at some spatial and temporal scales (Song, 2002). But how well does harvesting emulate fire from the perspective of wildfire itself? Do harvesting and burning have systematically different effects on ensuing fire events? It is imperative to quantify how harvest activity interacts with subsequent fire events to understand the development of the heterogeneous forest structure we currently see in managed landscapes, and to comprehend the long-term effects that ongoing forest management could exert on this ecosystem.

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Landscape heterogeneity in managed mixedwood boreal forests arises from harvesting and burning, the latter regulated by the interaction of weather, fuel, and ignition agents. The relative influence of fuel type (forest structure) and weather/fuel moisture on fire activity is controversial, since these factors contribute to fire behaviour and regime differently among scales and systems (Renkin and Despain, 1992; Bessie and Johnson, 1995; Hély et al., 2001; Schoennagel et al., 2004; Bigler et al., 2005; Krawchuk et al., 2006). However studies on mixedwood boreal forests have demonstrated that burn patterns (Cumming, 2001a), area burned (Hirsch et al., 2001), fire initiation (Krawchuk et al., 2006), fire behaviour (Quintilio et al., 1991) and the efficacy of fire management (Arienti et al., 2006) all differ among forest types. These effects can largely be explained by variation in the structural properties of fuel among mixedwood forest types, including coniferous, mixed, and deciduous stands (Hély et al., 2000; Krawchuk et al., 2006). Harvesting wood alters the structure of the forest and therefore changes fire behaviour but the degree to which this occurs in the mixedwood forest, and how it differs from burning, is not fully understood.

Simulation experiments have suggested that harvesting in mixedwood boreal forests might reduce the hazard of crown fires in the long term through an increase in the resultant area of deciduous dominated stands (Hirsch et al., 2001). However, these simulations did not account for short-term increases in fine fuel loads in recently harvested stands and Hirsch et al. (2001) speculate that post-harvest fire hazard might increase. Alexander (1982) and Baxter (2004) also suggest that forest harvesting increases fire hazard in the short term through piling of logging debris such as tops, limbs, un-merchantable stems and logs, and exposing this debris to atmospheric desiccation (Kiil, 1968). Fire operations personnel classify harvested stands in the mixedwood boreal forests of Alberta as a slash fuel type (Forestry Canada Fire Danger Group, 1992) for the first few years after logging, then as a grassy fuel type (Forestry Canada Fire Danger Group, 1992); in combination these types result in a high fuel load for up to 20 years post-harvesting (Kurt Frederick, ASRD, personal communication). In contrast, recently burned areas have depressed re-burn potential that has been shown to persist for up to three decades (Schimmel and Granstrom, 1997; Lavoie, 2004). A systematic comparison of how harvesting and burning influence subsequent fire occurrence in the boreal forest is necessary to quantify the extent to which these disturbances differ.

The objective of our study was to assess the relationship between lightning fire initiation and landscape composition under a mixed disturbance regime of wildfire and forest harvest in the mixedwood boreal forest region of Alberta, Canada. We were specifically interested in comparing the probability of initiation in areas that had been recently burned relative to areas recently harvested. In this study, we defined 'recent' as within the preceding 30 years. We used two general approaches for the study. We first used regression models to quantify the annual pattern of lightning fire initiation observed over eight years (1994–2001), within ≈10,000 ha landscapes in a 60,000 km^2 study region with respect to spatiotemporal variation in lightning activity, fire weather conditions and forest composition. We then included parameter estimates from our regression model in the dynamic fire-succession model, Tardis (Cumming and Armstrong, 2004) to characterize long-term effects generated by the regulation of lightning fire initiation by previous forest harvest and wildfire activity.

2. Methods

2.1. Study region

The study was located in the mixedwood boreal forest (Rowe, 1972) of central-eastern Alberta, Canada (Fig. 1). The regional climate is continental with mean historical winter and summer temperatures of −13.2 °C and +13.5 °C, respectively, and total precipitation in summer and winter averages of 240 and 64 mm, respectively (Strong and Leggat, 1992). Glaciation resulted in a surficial geology of rolling moraine deposits on the uplands and flat glaciolacustrine deposits on the lowlands (Rowe, 1972). There is limited relief with elevation varying from 200 to 1000 m.

The mixedwood is the most productive and diverse of boreal forest ecosystems in North America (Chen and Popadiouk, 2002). Forest tree communities are stratified by soil moisture (Strong and Leggat, 1992). Upland mesic sites are dominated by mixtures of trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.) or white spruce (*Picea glauca* (Moench) Voss). Post-fire community development at these sites depends on

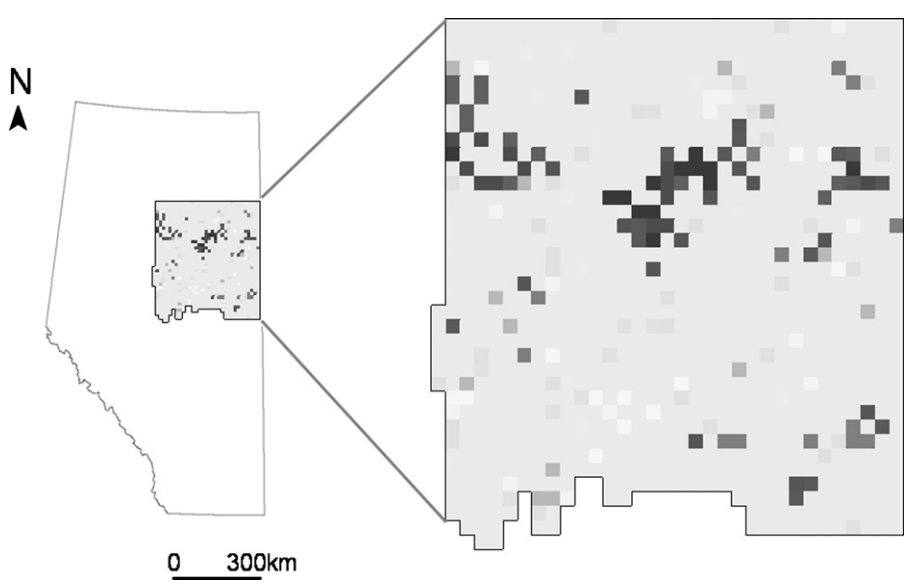

Fig. 1. The study region in central-eastern Alberta, Canada. Squares in the map show the size of ≈10,000 ha landscapes. The region presented here shows the 1118 landscapes used in simulations; we used empirical data from 588 of these landscapes in regression models. The gridded inset shows an example of simulated spatial pattern of area burned, where lighter colours indicate little or no burning and dark colours indicate greater area burned.

Table 1
The forest composition of the study region based on classification of Alberta Vegetation Inventory polygons.

Forest description	Moniker	Range (ha/landscape)	Percent[a] (region)	Incidence[a] (# of landscapes)
Deciduous dominated mesic, >90 years	MD_{90+}	0–3397	4%	550
Deciduous dominated mesic, 31–90 years	MD	0–7174	16%	587
Conifer dominated mesic, >30 years (white spruce)	MC	0–2914	5%	580
Black spruce (including larch and some birch/black spruce mixes), >30 years	BS	161–8762	44%	588
Jack pine (including deciduous dominated stands where pine is the dominant conifer), >30 years	JP	0–6781	8%	580
Burns, 1–10 years*	B_{10}	0–7129	2%	77
Burns, 11–20 years*	B_{20}	0–7718	4%	148
Burns, 21–30 years*	B_{30}	0–7156	1%	112
All burns[b], 1–30 years	B_{ALL}	0–8256	7%	240
Conifer harvest, 1–10 years@	HC_{10}	0–1951	0.2%	93
Conifer harvest, 11–20 years@	HC_{20}	0–2003	0.2%	61
Conifer harvest, 21–30 years@	HC_{30}	0–760	0.1%	46
Aspen harvest, 1–10 years@	HD_{10}	0–1667	0.9%	196
Aspen harvest, 11–20 years@	HD_{20}	0–1219	0.1%	74
Aspen harvest, 21–30 years@	HD_{30}	0–1260	0.3%	114
All harvest[b], 1–30 years	H_{ALL}	0–3636	1.8%	264
Other (anthropogenic features, water, wetland, un-vegetated ground, unknown year or year one of burn or harvest, salvage harvest.	na	na	14.2%	na

The * and @ indicate variables collapsed to B_{ALL} and H_{ALL}, respectively, for regression analysis (see Section 2.3).
[a] Values are presented for 2001.
[b] All harvest and All burns simply summarize the finer classifications presented above them.

variation in disturbance history, composition, and the availability of propagules from neighbouring stands (Albani et al., 2005). Paper birch (*Betula papyrifera* Marsh) is present but pure stands are rare. White spruce and balsam fir (*Abies balsamea* (L.) Mill) dominate older stands but pure stands are not abundant. Xeric sites are vegetated by jack pine (*Pinus banksiana* Lamb), sometimes in association with aspen. Moist, poorly drained sites are vegetated by black spruce (*Picea mariana* (Mill.) B.S.P.). Extensive wetlands are common, including open areas of bog, fens, swamps and marshes. Tamarack (*Larix laricina* (Du Roi) Koch) is the dominant tree species in these wetlands. The composition of the region is shown in Table 1.

Burning and harvesting are common in the region and Table 2 summarizes the total area affected over the study period. Lightning fires are responsible for the majority of area burned but the frequency of human-caused fires is increasing with road and trail access. Small-scale conifer harvest has occurred since the 1950s; extensive commercial harvest of aspen in the region began in 1993. Clear-cutting with minimum retention of in-block trees is the dominant harvesting practice and harvest slash is not burned. The coniferous harvest is dominated by white spruce, accompanied by smaller volumes of jack pine and black spruce and these site types are typically re-planted. Renewal of deciduous sites relies on natural regeneration, primarily *via* suckering. Oil and gas development has resulted in forest harvest along lines cut for seismic exploration (Schneider et al., 2003). The influence of these linear features on fire behaviour is debated, though Arienti et al. (2006) indicate these features do not systematically affect the initial growth and suppression of fires.

2.2. Empirical data

The Province of Alberta is partitioned using an administrative grid system where each landscape unit measures approximately 9600–10,000 ha, varying due to geographic corrections. Data were collected from 588 landscapes (Fig. 1) over eight fire seasons (May 1 to September 30) between 1994 and 2001 resulting in 4705 space-time units called voxels. Spatial data sets were managed in ArcGIS 9.1 and statistical analyses were performed in R (R Development Core Team, 2005, version 2.2.0). We combined existing records of lightning-caused fires, fire weather, lightning activity and forest composition from the study voxels in a multiple regression analysis to isolate the relative effect of previous harvesting and burning on subsequent lightning fires.

Fire data were provided by Alberta Sustainable Resource Development (ASRD, 2006). Undetected or unrecorded fires may have occurred in the region, but detection efficiency appears constant over the study interval (Cumming, 2005). We defined lightning fire initiation as an ignition that becomes large enough to be detected and thus recorded by ASRD. Multiple ignitions detected on the same day in the same landscape were the result of the same conditions, so we aggregated these events. We expect that the implication of aggregation is minimal under the hypothesis that significant spread would coalesce the fires. As a result, our fire initiation data described the number of days per voxel wherein at least one lightning fire was detected (Table 3). Systematic fire patrols and the daily tracking of lightning and storm data across Alberta provide observers with information to discern between lightning- and human-caused initiations with minimal error.

We used two joint weather-lightning (JWL) metrics to describe the coincidence of lightning activity and fuel moisture conditions conducive to fire, as suggested by Nash and Johnson (1996) and developed by Krawchuk et al. (2006). The JWLs for each voxel were calculated from spatially referenced daily fire weather and lightning data, recorded during fire seasons throughout the eight-year study period. To calculate the JWLs, we quantified

Table 2
Observed area burned (ha) by lightning and human-caused fires, and area harvested (ha) in the study region from 1994 to 2001.

Year	Burned	Harvested
1994	1283	8925
1995	68991	13254
1996	318	5223
1997	13	10662
1998	37195	9057
1999	553	5126
2000	558	7937
2001	1173	2421
Total	110084	62605
Mean	13760	7826

Table 3
Summary of annual lightning fire initiations for the region, mean lightning fire initiations per voxel, and variance in lightning fire initiations per voxel.

Year	Initiations	Mean	Variance
1994	82	0.14	0.15
1995	77	0.13	0.15
1996	15	0.026	0.025
1997	39	0.066	0.062
1998	212	0.36	0.42
1999	289	0.49	0.62
2000	85	0.14	0.16
2001	67	0.11	0.12
1994–2001	866	0.18	0.24

daily weather conditions conducive to fuel ignition using two fire weather indices of the Canadian Forest Fire Weather Index System (Van Wagner, 1987): the Fine Fuel Moisture Code (FFMC) and the Duff Moisture Code (DMC). The FFMC and DMC represent moisture content of litter or fast-drying fine fuels and loosely compacted organic layers of moderate depth, respectively. The FFMC takes values from 0 to 100, and the DMC takes values from zero to infinity. High values represent dry forest fuels and a high risk of fire.

We calculated daily values of the FFMC and DMC from measurements of temperature, relative humidity and precipitation taken at an array of 65 provincial weather stations operated and calibrated by ASRD in the region. The two codes were then interpolated over the study region at a resolution of 1 km by inverse-distance weighting (Flannigan and Wotton, 1989) using the six nearest weather stations as informative neighbours. We then combined these fire weather codes with lightning data obtained from ASRD detection stations across the province. Assessment of the potential for spatial error in lightning detection (Nimchuk, 1989) is discussed in Krawchuk et al. (2006) and suggests minimal effect on JWL calculations. For each landscape in each year (a voxel), we tallied the number of days when FFMC and DMC exceeded critical values and lightning was detected, and used this co-occurrence to produce two JWLs: $FFMC_{JWL}$ and DMC_{JWL} (Table 4). The critical values for FFMC and DMC were set at 87 and 34, respectively; values above which a majority (85%) of observed lightning fires were detected over the period 1983–1993 in the same region (Krawchuk et al., 2006) and in data collected by Anderson and Englefield (2001) from a similar forest region in Saskatchewan, Canada.

We used the seasonal severity rating (SSR) from the Canadian Forest Fire Weather Index System to measure annual climate variation within the region. The SSR was calculated as the mean annual Daily Severity Rating of all weather stations over the fire season (Table 4). Severity ratings measure the difficulty of controlling a fire based on wind speed, rainfall, humidity, and temperature. The positive relationship between annual SSR (Table 4) and annual area burned (Table 2) emphasize additional links between this climate metric and fire activity.

Fuel composition for each voxel was quantified with the Alberta Vegetation Inventory (AVI), which catalogues vegetation structure (forest canopy species composition, density) and the estimated age of forested and non-forested stands. Stands are delineated as homogeneous polygons from 1:15,000 air photo interpretation and field checking with a minimum resolution of 2 ha. Disturbance type and year is also recorded in the AVI. Photography for the AVI was collected between 1992 and 1999, and we updated our database to 2001 with recent fire and harvest events.

We adapted an existing ecological classification developed by Cumming and Vernier (2002) to describe the fuel structure of each AVI polygon (Table 1). The algorithm used the first to fifth dominant species and the associated percentage within each polygon according to crown closure, estimated year of stand origin, natural but non-forested cover types, anthropogenic cover types, and the year, type and extent of disturbance. We tracked the area classified as harvested (H) and burned (B) in each voxel within the preceding 30 years using decadal classes (Table 1). We separated the harvested polygons as harvested conifer (HC) or harvested deciduous (HD) to capture potential variation caused by differences in harvest slash and post-harvest conditions such as needle/leaf, branch and bark flammability. Specific harvest days were not recorded in the AVI and so ordering of multiple disturbances within a given year was impossible to resolve. To avoid misclassifying the order of disturbances, for each year of fire data the areas burned or harvested in that year were not included in the calculations.

One final step was used to prepare data for analysis; this involved retrieval of additional historical forest stand information since original stand attributes are over-written in the AVI when disturbances occur. We developed processing algorithms to recover these early data based on the relative proportion of forest types for each landscape, available from a previous inventory that was updated until 1993. We calculated the proportion of MD, MC, MD_{90+} (Table 1) for each voxel from the 1993 inventory, and attributed the areas harvested in the current AVI according to these proportions, given that these three cover types are harvested most frequently. Since fires select different resources to burn than harvest operators select to cut, we used the proportion of MC, MD, MD_{90+}, BS, JP and O to replace areas burned. We weighted the replacement composition for burned areas according to their relative propensity to burn, estimated in Cumming (2001a).

2.3. Regression models

We ran exploratory data analyses, then used generalized linear regression models to quantify the relationship between counts of fire initiation per voxel and environmental variables. Overall, exploratory analyses of the data suggested a zero-inflated negative binomial regression model without explicit *a priori* concern for structured dependence or autocorrelation in errors. The variance-to-mean ratio of fire initiations was 1.33 (Table 3) suggesting overdispersion relative to a Poisson distribution, and the data

Table 4
Summary of joint weather-lightning indices (JWLs) for Fine Fuel Moisture Code (FFMC) and Duff Moisture Code (DMC), and the annual regional Seasonal Severity Rating (SSR).

Year	$FFMC_{JWL}$ mean days/voxel	$FFMC_{JWL}$ total days annually	DMC_{JWL} mean days/voxel	DMC_{JWL} total annual days	SSR annual
1994	2.3	1358	2.4	1399	1.8
1995	1.1	645	2.4	1428	2.5
1996	0.7	425	0.1	77	0.7
1997	2.1	1237	0.1	61	0.8
1998	4.5	2643	7.6	4495	3.3
1999	3.3	1910	1.4	807	2.0
2000	1.0	581	0.0	12	0.7
2001	2.0	1200	0.9	534	1.8

contained a large proportion (85%) of zeros suggesting a zero-inflated model (Martin et al., 2005). To assess if the assumption of independent spatial distribution in errors was likely to hold for regression analyses, we tested for spatial autocorrelation in the fire initiation data using a local Moran's I statistic (Legendre and Legendre, 1998). Test statistics were calculated separately for 10 km distance classes up to 60 km for each year of data; these distances represent tests for autocorrelation between first to sixth neighbours. We applied Bonferroni correction and a more conservative, progressive Bonferroni correction to tests of significance (Legendre and Legendre, 1998) with an initial critical $\alpha = 0.05$. The progressive Bonferroni tests indicated spatial autocorrelation in fire initiation between nearest neighbours (20 km), whereas the Bonferroni tests suggested no autocorrelation. When assessed together, these tests indicated raw fire initiation data were relatively independent in space. The time series of fire initiation data from each voxel were not inherently temporally autocorrelated because lightning activity – the cause of the lightning fire – is an externally generated process. As validation, we used box-and-whisker plots to detect any spatially stable pattern in lightning activity or JWLs through time within each voxel, and found no evidence for it.

We assessed the functional form of bivariate relationships between counts of fire initiation and each explanatory variable (JWLs and forest composition) using plotting functions provided in R in association with generalized additive models. Explanatory variables all had linear relationships with fire initiation with the exception of the composition variable, BS (Table 1), which we then fit with a quadratic term BS^2 due to its convex shape.

Highly correlated pairs of explanatory variables were screened using Pearson correlation coefficients. As expected, some burn (B) and harvest (H) categories presented in Table 1 (see * and @) resulted in correlation of up to 0.8. Inter-correlation of variables in regression analyses interferes with interpretation of estimates and so we identified the most parsimonious aggregation of the H and B data. According to selection using AIC from a global model that included all covariates for each disturbance group, the binning of six harvest ($HD_{(10,20,30)}$ and $HC_{(10,20,30)}$) variables to H_{ALL}, and of three burn ($B_{(10,20,30)}$) variables to B_{ALL} (Table 1) resulted in the most parsimonious description of fire initiation. Although the collapse of nine variables to two resulted in a loss of information describing the time since disturbance, the covariates contributing to H_{ALL} were all positive in a separate, bivariate analysis and those contributing to B_{ALL} were all negative, suggesting the procedure did not obscure any age-related differences in polarity. Of note, the variable HD_{10} explained the most variability within H_{ALL}, and was also the variable with the greatest area in the study region (Table 1). The variable HC_{30} ranked second, but had the lowest area. Baxter (2004) suggests that by eight years post-harvest there is no difference in the flammability of conifer and deciduous harvest slash, adding support to our treatment of deciduous and coniferous harvest as a single category of fuel/cover type in the H_{ALL} variable. The $FFMC_{JWL}$ and DMC_{JWL} variables had a high correlation of $r = 0.6$ but because we had no reason to maintain one over the other, we included both in the global regression model, with an *a priori* decision to maintain only one in the final model.

We fit four global regression models using Poisson, negative binomial, zero-inflated Poisson and zero-inflated negative binomial distributions to explain fire initiation with the 11 explanatory variables: $FFMC_{JWL}$, DMC_{JWL}, SSR, H_{ALL}, B_{ALL}, MC, MD_{90+}, MD, JP, BS and BS^2 (Tables 1 and 4). As suggested in exploratory analysis, the zero-inflated negative binomial (ZINB) model was a superior fit based on the likelihood ratio test ($\alpha = 0.05$) and the Vuong statistic. We pre-selected a mixture distribution for the ZINB model (see Martin et al., 2005) based on our *a priori* understanding of the processes responsible for fire initiation. The mixture distribution of the zero inflated models consists of two parts, the binomial and the negative binomial. We considered weather/lightning/moisture variables, $FFMC_{JWL}$, DMC_{JWL} and SSR, as preconditions for ignition in the binomial contribution to the overall number of zero counts. We considered weather/moisture and vegetation, $FFMC_{JWL}$, DMC_{JWL}, SSR, H_{ALL}, B_{ALL}, MC, MD_{90+}, MD, JP, BS and BS^2, in the negative binomial contribution to the overall count distribution.

We used a backward, stepwise AIC selection procedure from the global ZINB model to obtain the final, best model. In the final model, we also tested the statistical significance of parameter estimates using the Wald statistic ($\alpha = 0.05$). *A posteriori*, we tested for a significant interaction between a categorical variable for year and both H_{ALL} and B_{ALL} to determine if their relationship with fire initiation was consistent across the study interval.

We used a-spatial plots and variograms of Pearson residuals to assess the adequacy of the best model. We assessed the fit by looking at the difference between observed and predicted probabilities of counts, as prescribed by Lambert (1992). As an external validation, we systematically excluded each year of data to assess the difference between the observed (omitted data) and predicted probabilities of counts. Model specification and goodness of fit were tested using 1000 iterations of a parametric bootstrap (Efron and Tibshirani, 1998) where each iteration generated ZINB random deviates from the best model, and we compared the sum of squared errors (SS) from these simulated data with the SS of the observed data.

2.4. Simulation

We performed a simulation experiment to examine the magnitude of long-term, emergent effects of the relationships between fire initiation, recently harvested and burned area estimated in our statistical models. We used 100-year simulations in the dynamic vegetation model, Tardis (Cumming and Armstrong, 2001, 2004), a spatially explicit stochastic model of mixedwood boreal forest dynamics and management developed and calibrated for this region (Cumming and Armstrong, 2001, 2004; Cumming and Vernier, 2002). Tardis simulates the interactive processes of wildfire, harvesting, regeneration and succession. The model operates at a temporal resolution of five years using 10,000 ha landscape units (Fig. 1).

Each landscape unit in Tardis was initialized with variables describing vegetation composition using aspatial stand tables from the AVI data. As in the regression analysis, forest attributes were used to classify stands into ecologically meaningful cover types for the fire, harvest, regeneration and stand dynamics modules described below. Stand attributes changed over time as a function of simulated disturbance and stand dynamics. Fire and harvesting re-set the successional trajectory within forest stands and via regeneration rules. Non-forested cover types such as peatland or water were invariant throughout the simulation.

Wildfire was simulated by four independent stochastic processes: initiation, escape, final size, and effect (Cumming and Armstrong, 2004). The number of fires initiated per time step and landscape was simulated based on an additive function of weather conditions and vegetation composition using parameter estimates from the ZINB model we developed (Table 5). The initiation model was called five times within each five-year time step and each call used a different year of the observed JWLs obtained by a random draw of the annual, observed data used to fit the fire initiation model. Vegetation covariates for each landscape were calculated from Tardis stand attributes.

Fire initiations that exceed a threshold size of three ha are said to have escaped (Cumming, 2005), and for each time step and landscape, the number of fire escapes was sampled from a binomial distribution (p_e, N_i) where p_e was the probability of

Table 5
Parameters and standard errors estimated by the best ZINB model.

Parameter	Estimate	Standard error	*Z*-value	*P*-value	ΔAIC
Intercept (B)	+1.78	0.28	+6.27	3.78×10^{-10}	
SSR	−1.76	0.35	−5.01	5.54×10^{-7}	87
Intercept (NB)	−1.94	0.32	−6.07	1.29×10^{-9}	
H_{ALL}	+4.06	1.13	+3.59	3.30×10^{-4}	10
B_{ALL}	−1.24	0.45	−2.77	5.58×10^{-3}	6
MC	+1.87	0.86	+2.18	2.91×10^{-2}	3
MD	−2.31	0.47	−4.88	1.06×10^{-6}	22
BS	+2.45	1.029	+2.34	1.70×10^{-2}	4
BS^2	−2.50	1.026	−2.36	1.92×10^{-2}	4
$FFMC_{JWL}$	+0.13	0.018	+7.23	4.93×10^{-13}	50
Log(Theta)	0.85	0.33	+2.57	1.033×10^{-2}	10

The log of the likelihood = −2246.19. Calculated *p*-values are presented from the Wald test statistic. A change in the AIC with the removal of each term is indicated by ΔAIC. The binomial component of the model estimated the $\Pr(Y = 0)$ so the $\Pr(Y = 1)$ is the opposite sign giving a positive relationship between fire initiation and SSR.

escape, and N_i was the total initiation count. We used a value of $p_e = 0.12$, estimated following methods by Cumming (2001b) based on historical data and representing current fire management effectiveness (Cumming, 2005). Final sizes of escaped fires were sampled from a truncated exponential distribution developed for the study region from regional, empirical data from 1980 to 1998 (Cumming, 2001b); no fires larger than those recorded during this 19-year timeframe have occurred in the region since 1931 (Delisle and Hall, 1987). The largest fire recorded during this interval was 428,000 ha and the truncation point of the distribution allows a maximum fire size of 643,000 ha. The distribution's shape parameter was estimated from regional empirical data and varies as a function of vegetation composition of the landscape where the fire was initiated, generating smaller fires when they started in recently disturbed areas (areas harvested and burned within the last 20 years).

The area burned by a fire is strongly related to the composition of the landscape in which it started (Cumming, 2001a). Following Cumming (2001a), we modelled the burned forest types using a multivariate function of forest composition in each landscape and that of its neighbours. Crown fires and intense surface fires caused by lightning burn the majority of area in the boreal forest (Bonan and Shugart, 1989), and for simplicity we assumed that all fires were stand-replacing. We did not consider human-caused fires explicitly in our simulations.

Forest harvest was simulated to emulate existing forestry facilities and practices in the study region, but used a global forest management policy presented by Cumming and Armstrong (2001, 2004) that centralizes harvest planning. Harvest scheduling was simulated by greedy optimization (Cumming and Armstrong, 2001) whereby Tardis determined the cheapest harvest options within each time step. Timber volumes were calculated from the forest inventory using seven yield classes developed for the region by Timberline Forest Inventory Consultants Ltd. (Anon., 1994). When stands were harvested, all timber volume was removed and the stand age was set to zero.

Tardis simulated regeneration after disturbance and stand dynamics with a set of simplified rules based on the seven yield classes. The trajectory of individual stands within the mixedwood boreal forest is variable however we simplified stand dynamics so that when stands burned they simply regenerated to their pre-burn yield class. When peatlands burned they were regenerated as peatlands. Water and anthropogenic development never burned. However, harvested mesic forests were regenerated according to defined rules based on current management policy (Cumming and Armstrong, 2001) whereby mesic forest is reforested as either white spruce or aspen dominated stands.

We ran two scenarios in Tardis: one that included both harvesting and fire; the other where fire was the only disturbance. Ten replicates of each scenario provided a general distribution in results created by the stochastic representation of fire. We measured two indices to describe the regional effects of the mixed disturbance regime. The first index, ΔYEAR, was the number of years between two consecutive disturbances in a given stand; it tracked the frequency of disturbance for every AVI stand that was burned or harvested over the simulation. Within the simulation's disturbance history, a stand that was harvested then burned was labelled HARVESTED–BURNED (H–B), and a stand that was repeatedly burned was labelled BURNED–BURNED (B–B). We determined the magnitude of change in the frequency of disturbance for all forest types combined by comparing the distribution of ΔYEAR between H–B and B–B in scenarios with harvesting and fire, to that for B–B in the fire-only scenario. We also calculated ΔYEAR from observed disturbance intervals that occurred in the study region and were recorded in the AVI. The second index we calculated was AREA, the cumulative area affected by each combination of disturbance types (B–B or H–B). We expected a decrease in ΔYEAR for H–B relative to B–B, but had no *a priori* expectation of the magnitude of the effect and overall area that would be affected over time.

3. Results

3.1. Regression models

The ZINB model with covariates SSR, B_{ALL}, H_{ALL}, BS, BS^2, MC, MD, and $FFMC_{JWL}$ gave the most parsimonious quantification of lightning fire initiation (Table 5). The positive relationship with seasonal severity rating (SSR) was the only significant variable in the binomial part of the model, and although the effective sample size for SSR was eight (years) rather than 4704 voxels, the reduction in AIC was large (ΔAIC 87) suggesting SSR should be maintained in the model. Fire initiations decreased with the area burned in the preceding 30 years (B_{ALL}; Table 5) but increased with the area harvested (H_{ALL}). There was no interaction between year and H_{ALL} or B_{ALL}, except for between B_{ALL} and the year 1995, where the interaction was significant with a *p*-value of 0.02. A conditioning plot of fire initiation as a function of B_{ALL} suggested this effect was not strong relative to the remaining, insignificant interactions with year.

The variance of residuals was higher in 1998 and 1999 than in other sample years. These two years also had the greatest number of fire initiations, JWL days, and highest values for SSR suggesting additional covariates might be required to describe fire initiation in 'extreme' fire years. Overall, there was minimal difference between the observed and predicted probabilities of counts generated by the best model, with a maximum difference of 0.001. Our external validation, where each year of data was systematically left out of model estimation, showed higher differences between predicted and observed probabilities of counts than the model using all data. Differences between predicted and observed probabilities did not exceed ±0.01 except for 1999, where the model had reduced ability to discriminate between zero and one. The goodness of fit metric calculated using a parametric bootstrap indicated that the best ZINB model sufficiently described the distribution of the data (SS = 1143, $p = 0.42$, $n = 1000$).

To demonstrate the opposing effects of H_{ALL} and B_{ALL} on fire initiation, Fig. 2 illustrates changes in expected initiation when a featured forest type was either harvested or burned (H_{ALL} or B_{ALL}), according to parameter estimates from our best model. We present an abscissa ranging from 0 to 4000 ha which corresponds to the values of H_{ALL} observed across the data. We used a composite of data so that forest composition in each scenario was set to the

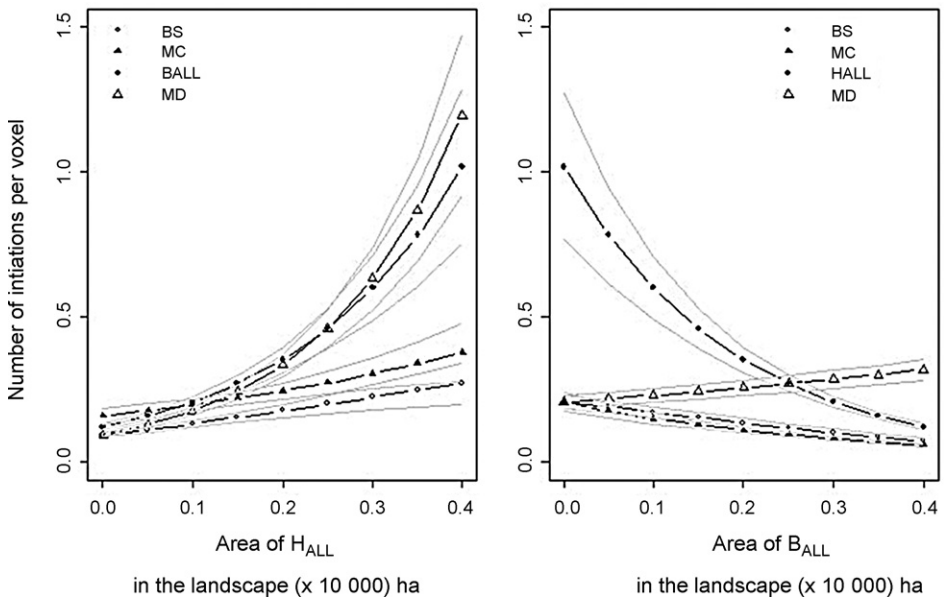

Fig. 2. Changes in expected fire initiation associated with changing landscape composition. Prediction bands are illustrated in grey. At left, the expected number of initiations increases as black spruce (BS), white spruce (MC), burned forest (B_{ALL}) and aspen forest (MD) are replaced by harvested forest (H_{ALL}). At right, the expected number of initiations decreases as MC, BS and H_{ALL} are replaced by B_{ALL}, and increases as MD is replaced by B_{ALL} (right). There is a large effect on predicted initiation associated with the replacement of B_{ALL} by $H_{ALL,}$ and vice versa, illustrated by '●' in both panels.

mean observed value of each covariate and weather covariates were set to their mean observed values and did not vary among scenarios. Prediction bands around the ZINB expectations were calculated using 1000 Monte Carlo simulations of standard errors from parameters estimated in the best model. The increasing curves in the left panel of Fig. 2 dramatize the increase in fire initiations expected as more of a landscape is harvested. Forest composition is altered via the abscissa as BS, MC, B_{ALL} or MD is replaced by H_{ALL}. For example, as the area of H_{ALL} (Fig. 2 left panel) increased, the area of MC, MD, BS, or B_{ALL} each decreased, while the other cover types were held constant. Of particular interest is the steeply increasing curve for MD because harvesting activity in the study region primarily focuses on deciduous species. As a contrast in the right panel, composition is altered via the abscissa as BS, MC, H_{ALL} or MD is systematically replaced by B_{ALL}. Fire initiations could either increase or decrease with increased burning, depending on the forest type burned. The net effect of these transitions is difficult to predict without the use of more complex methods, such as the simulation models we present in the latter part of this study.

3.2. Fine-scale evaluation of statistical patterns

The positive relationship between fire initiation and area of harvested forest implies fires are more likely to start in harvested stands, however the regression analysis lacks spatial resolution to validate a fire's specific placement. To test the inference we used spatially explicit AVI for a subset of 171 voxels where an area of 1000 ha or more had been harvested, intersected with point data recorded by fire operations staff. There were 73 initiations recorded in these samples. Since fire locations were not always precise we calculated a 100 m radius buffer around each fire, then classified it as occurring in a harvested stand if its sampling radius overlapped with an AVI polygon classified as H_{ALL}. Using this criterion, 28 of the 73 fires (38%) were recorded as initiating within the H_{ALL} cover type. If initiations were a spatially random process we would expect nine fire initiations in the harvested cover type given the total area of H_{ALL} in the selected region. These proportions were significantly different using an exact binomial test ($p = 4.7 \times 10^{-8}$). A disproportionate number of fires are in fact starting in, or very near to, harvested stands. This local analysis supports our regional statistics that the probability of initiation increases as a function of the area of recently harvested stands.

3.3. Simulation

Stand-level disturbance by harvesting or burning ranged between zero and four events over 100 years in both scenarios. For those stands disturbed by two or more events, 78% of stands were disturbed twice, 19% were disturbed three times and 3% were disturbed four times. For stands that experienced more than two disturbances we divided the disturbance history into pairs. For example, if a stand was burned in year 10, harvested in year 80 and burned in year 85, this stand then had two disturbance pairs: 10:80 and 80:85. The mean ΔYEAR between pairs across 10 replicates of the harvested scenario was 26.3 for H–B and 37.4 for B–B (Fig. 3a); the mean was significantly lower for H–B than for B–B ($t = -167$, $df = 222{,}077$, $p = 2.2 \times 10^{-16}$). The mean interval in the un-harvested (fire only) scenario was 37.5 for B–B (Fig. 3a). Forest harvest occurred on MC and MD stands, whereas fires burned proportionally more BS, JP and MC than MD. As a result, the H–B events not only occurred at a reduced interval compared to B–B, but affected different forest types than the B–B events. Data from actual events recorded in the AVI show the same pattern as the Tardis simulations (Fig. 3b). For the observed H–B, the minimum value was zero, mean was 5.7 years, and maximum 19 ($n = 101$ events). For the observed B–B, the minimum value was three, mean was 16.3 years, and maximum 32 ($n = 102$ events). Though the simulated and observed ΔYEAR values show similar patterns, the relatively short disturbance intervals of H–B and B–B from the AVI records reflect an observation period that covers only 32 years and is therefore biased to smaller intervals.

The mean area (AREA) affected by H–B was 247,294 ha, corresponding to an area of 24,729 ha per decade (Fig. 3c), or roughly 2.5 landscapes per decade. Since approximately 25% of the study region was harvestable mesic forest, the harvestable area affected annually by H–B was 1%. The mean area affected by B–B was 639,165 ha in the harvested scenario, and 734,468 ha in the un-harvested scenario, corresponding to an area of six and seven

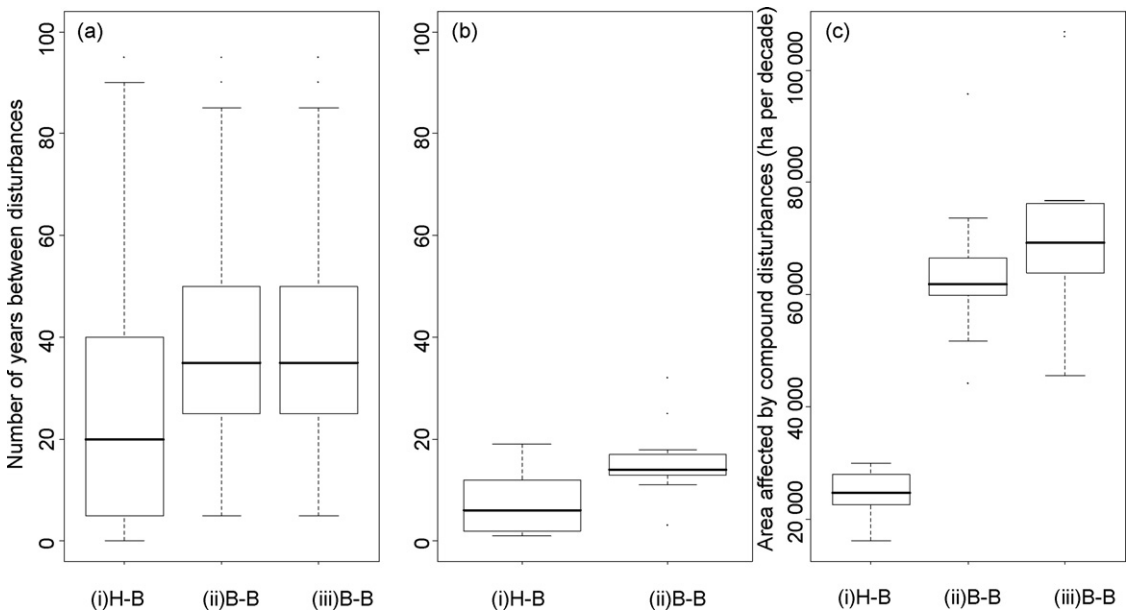

Fig. 3. The boxplot distribution of return intervals, summarized for all simulations, between two consecutive disturbances (a) for stands in harvested (i and ii) and unharvested (iii) scenarios. HARVESTED–BURNED (H–B) refers to areas where a harvesting is followed by fire, BURNED–BURNED (B–B) refers to areas burned repeatedly. Panel b shows the distribution of observed intervals recorded in the AVI. Panel c shows the area of forest affected by these compound disturbances for harvested (i and ii) and unharvested (iii) scenarios.

landscapes per decade, respectively. Thus, despite the strong, positive relationship between harvested areas and fire initiation detected in our statistical model, the area affected over the long-term by a reduced disturbance interval was relatively small.

4. Discussion

We quantified the effect of forest management on regional lightning fire initiation in the mixedwood boreal forest and found an increased probability of fire in landscapes with increasing amount of recently harvested forest while fire initiation decreased in areas recently burned. A fine-scaled evaluation confirmed that fire initiation was more likely to occur within recently harvested stands than would be expected by chance. Our model showed the increased fire initiation in harvested stands might last for as along as 30 years, and fine temporal detail of this harvest effect remains unresolved due to our relatively coarse sampling protocol although data suggested the effect was strongest in harvested aspen stands up to a decade old. In contrast, the probability of fire initiation decreased with the area of recently burned forest, and this effect also lasted for three decades. Our analyses imply that though some aspects of harvest activity may successfully emulate natural disturbance, harvesting is not a surrogate for fire from the perspective of fire itself. Furthermore, our research adds a caveat to suggestions that harvesting reduces fire hazard and can be considered a method of fire protection. This may be true in the long-term, however we show the efficacy of harvest as a prevention strategy against fire initiation is clearly questionable in the years immediately post-harvest.

The identification of structural differences in post-harvest and post-burn vegetation is necessary to understand mechanisms responsible for the differences in fire initiation we observed, although our broad age classes smooth out substantial variation in fuel characteristics. The initiation of a lightning fire requires and ignition event, and then suitable conditions to sustain a fire so that it grows large enough to be detected. Standing harvest residuals may act as conduits for lightning to strike within a harvested stand where dry and abundant fine fuels are available to burn (Kiil, 1968; Song, 2002; Baxter, 2004), although Kiil (1968) considers the fire hazard associated with fine fuels from harvesting slash as transitory, decreasing with increasing decay and disintegration.

In addition to the fine fuels and slash remaining after forest harvest, post-disturbance regeneration might also contribute to flammability. Song (2002) suggests that areas containing aspen prior to harvesting will typically regenerate through a period of lush growth from 'suckering' aspen shoots, towards maximum leaf area and canopy density in the period 10–25 years after a fire. Lush, deciduous growth is thought to reduce the potential for fire spread (Forestry Canada Fire Danger Group, 1992) however Greenway et al. (2002, in Song, 2002) suggest aspen regeneration may be inhibited in post-harvest stands as compared to post-burn stands due to competition with grasses such as *Calamagrostis canadensis* (Michx.) (Hogg and Lieffers, 1991). When *Calamagrostis* survives in the understory of mixedwood stands it may quickly dominate a harvested site and compete with aspen (Greenway and Lieffers, 1997; Greenway et al., 2002, in Song, 2002) resulting in dead *Calamagrostis* shoots and litter in combination with abundant slash material, that results in increased fire hazard (Forestry Canada Fire Danger Group, 1992) despite any regeneration of young, so-called fire-proof, deciduous trees. The aspen stand type represents most of the area harvested in the study region. Finally, Hogg and Lieffers (1991) suggest that *Calamagrostis* dominates after low severity burns or harvesting, whereas fireweed (*Epilobium angustifolum* L.) that is less flammable due to rapid disintegration of dead shoot material, is favoured after severe fire. All of these structural features suggest greater flammability on recently harvested stands than those recently burned. However, alternative hypotheses must also be investigated, for example, that the effect is caused from harvest occurring disproportionately on dry sites inherently more likely to burn than wetter ones.

Forest managers have suggested it might be necessary to burn harvested areas to maintain biodiversity in mixedwood boreal forests, and we speculate this treatment might also be prudent to reduce fire hazard. Stephens (1998) demonstrates variation in fire behaviour through a suite of harvest/biomass removal/burn treatments in Californian mixed-conifer forests, showing that post-harvest burn treatments resulted in the reduction of subsequent fire behaviour. However, the perceived risk of hold-

over or escaped fires precludes systematic burning of slash in our study region (Baxter, 2004).

Variation in the probability of fire initiation was related to the relative composition of black spruce, mature aspen and white spruce in addition to harvested and burned areas. Fire initiation increased with greater areas of black and white spruce while decreased with a greater area of aspen. Furthermore, a lightning fire cannot occur without appropriate weather conditions and our analyses showed that annualized indices of daily fire weather and lightning ($FFMC_{JWL}$) and regional weather (SSR) also explained significant variation in fire initiation in addition to forest structure. These findings are consistent with previous research in the region demonstrating biotic and abiotic regulation of fire initiation by forest cover type and weather for a time series prior to harvesting activity (Krawchuk et al., 2006).

Our simulations were orchestrated to identify the emergent regional effect of the relationship between fire initiation, harvested, and burned area that we detected in regression models. Over time the positive relationship between harvest and fire initiation manifested as an expected, shorter fire return interval after harvesting than after burning. These simulated results were supported by observed disturbance intervals recorded in the AVI, even though the observation period was relatively short. Since the majority of harvest occurs in mesic forests containing aspen and white spruce communities, these types were disproportionately affected in comparison to those incurring consecutive burns. Fire is relatively rare in aspen stands (Van Wagner, 1977; Quintilio et al., 1991) so the short disturbance interval introduced by the compound effect of harvesting followed shortly by burning is novel to this mixedwood boreal ecosystem. The simulations demonstrate how the outcome of the interaction between harvesting and fire is an accelerated disturbance that is more frequent than the natural disturbance interval (BURNED–BURNED), and thus an alteration of the fire regime that forest managers seek to emulate.

The cumulative area affected by the overlapping disturbance of harvesting and burning was approximately 1% of the merchantable forest, amounting to a local peculiarity rather than a broad scale effect. However, increases in fire and harvest activity expected with climate change and with further development could expand the area affected by compound disturbances in the future. Since our study region is a managed forest, silvicultural intervention such as planting or site preparation could mediate some of these outcomes. However, there is currently no stipulation to re-stock harvested areas that are subsequently burned.

Outcomes of the two simulation scenarios (harvested and unharvested) are fully dependent on the integration of many decision rules and parameters coded within the software. For example, Tardis does not include a relationship between forest harvest, fire escape and fire size. If these effects are also strong and positive, they could increase the area of compound disturbance generated by forest harvest. In contrast, the strength and duration of an increase in fire immediately after harvest might be balanced by reduced fire activity induced as harvested areas age to mature aspen stands that are less flammable. Here, we have only described and implemented simple changes to Tardis to isolate the effects of burn and harvest parameters on the fire initiation process. The creation of a simulation with specific goals to quantify changes in forest composition and additional fire indices (e.g., area burned) is a necessary next step in understanding the net effects of the interaction of harvesting and burning in the mixedwood boreal forest.

The frequency and type of disturbance in and around a forest stand can influence stand composition through an interaction with species' life history strategies (Weir et al., 2000; Johnstone and Chapin, 2006). Johnstone and Chapin (2006) illustrate that shorter fire return intervals (<30 years) result in decreased densities of conifer regeneration relative to expected pre-fire community dominance by conifers in the boreal forest of British Columbia and the Yukon, Canada. Weir et al. (2000) suggest that species with older ages of maturity, no serotinous cones, and no vegetative reproduction such as white spruce were reduced or locally exterminated through an increase in fire frequency propagated by homesteading in Saskatchewan's mixedwood boreal forests. The ΔYEAR index illustrated that harvested stands burned before the expected next harvest, and in the case of white spruce stands before mature, cone-bearing individuals could grow to contribute seed to the area. A decrease in white spruce recruitment through accelerated disturbance could add even more pressure to concerns of 'unmixing' the mixedwood boreal via current regeneration policies (Weir et al., 2000; Cumming, 2001a), through a reduction in white spruce populations.

Paine et al. (1998) described compound disturbance as 'disturbances that have occurred either simultaneously or in a sequence rapid enough that recovery from the single pulse has not significantly progressed'. Compound disturbances are suggested to initiate transition among alternative stable ecosystem states (Paine et al., 1998; Jasinski and Payette, 2005), but Paine et al. (1998) posit that three or more disturbance events may be required to initiate this change. In our simulation experiment, the majority of disturbed stands incurred only two disturbance events over 100 years, but three or four pulses also occurred. The amount of disturbance that a system can experience without a change in state, via variables and processes that control behaviour, is referred to as its ecological resilience (Holling, 1973). Though an alternative stable state (Holling, 1973; Scheffer et al., 2001; Beisner et al., 2003) has not yet been identified in the mixedwood boreal forest, a *Calamagrostis*-dominated community assemblage, as described earlier, could be considered a candidate. Fires that burn through recently harvested areas, and which are carried by grasses and fine fuels, may be less intense than those fires that burn abundant or larger fuels in stands with mature trees conducive to crowning. The result would be a feedback of low-intensity grass fires and subsequent grass recruitment (Hogg and Lieffers, 1991; Brooks et al., 2004) suggestive of an alternative stable state under environmental conditions also suitable for mixedwood forest growth.

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