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A landscape-level tool for assessing natural regeneration density of *Picea mariana* and *Pinus banksiana* following fire and salvage logging

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ABSTRACT

We present a landscape-level operational natural regeneration assessment tool, created by linking a validated forest regeneration model with forest inventory maps. Using basal areas obtained from temporary plots and seedbed distributions from field data, seedling densities are simulated for pure *Picea mariana* (Mill.) and *Pinus banksiana* (Lamb.) stands under burned intact, and traditional first winter and delayed salvage scenarios. These stands are grouped by age-class, pre-fire stand cover, surficial deposit, and drainage. Following this classification scheme, simulated seedling densities are transferred onto forest inventory maps for the Lake Matagami lowland ecological region (6a) in the boreal forest of Quebec (used as a case study to illustrate the potential of the tool). The final output of the model is an estimate of seedlings/m^2 following moderate to severe fire, and fire followed by 100% salvage in pure *P. mariana* and *P. banksiana* stands, however it is also capable of simulating partial salvage. Results are expressed as seedlings/ha, and illustrate that in our study area, only 35% of intact *P. mariana* and 6% of intact *P. banksiana* stands need to be planted following fire; however under the traditional 100% first winter salvage scenario, 100% of *P. mariana* and 74% of *P. banksiana* stands necessitate planting. If salvage logging is delayed until the second, third, or fourth winter following fire, planting will be required in 98%, 88%, and 66% of *P. mariana*, and 65%, 56%, and 56% of *P. banksiana* stands respectively. This type of tool allows managers and foresters to quickly assess reforestation needs following fire and salvage at the landscape level, and can be used to better plan the timing and location of salvage operations and subsequent silvicultural treatment application. In addition to being able to schedule operations faster, foresters will also be able to quickly identify regions where natural regeneration could be inadequate or excessive. Potential cost estimates of future interventions such as planting, aerial seeding, and pre-commercial thinning could be made. Foresters can also assess the current vulnerability of management units to fire and can identify regions at particular risk.

1. Introduction

Picea mariana (Mill.) BSP (black spruce) and *Pinus banksiana* (Lamb.) (jack pine) are two common conifer tree species in the boreal forest of North America (Charron and Greene, 2002; De Groot et al., 2004), and also the significant target species of the forest industry. These two species are well adapted to fire; *P. banksiana* is serotinous and *P. mariana* is semi-serotinous. Both possess aerial seedbanks that effectively distribute seeds following this disturbance (Gauthier et al., 1993; Enright et al., 1998; Greene et al., 1999). *P. banksiana* releases approximately 90% of its seeds in the first year following fire whereas *P. mariana* takes up to five years to release the same amount (Greene et al., 2013). Furthermore, smoldering combustion of the organic layer creates seedbeds that are more receptive (Miyanishi, 2001; Miyanishi and Johnson, 2002) to the seeds of these two species.

Fire is the dominant disturbance in the boreal forest of North America, burning approximately 2 million ha of forest in Canada annually between 1959 and 1997 (Stocks et al., 2002). 97% of total area burned is caused by large (>200 ha), high intensity, stand-replacing fires (Amiro et al., 2001; Flannigan et al., 2001; Stocks et al., 2002). The regional and continental-scale fire regime in the

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boreal forest is predominantly controlled by climate (Carcaillet et al., 2001). Fire frequency (Amiro et al., 2001; Flannigan et al., 2001, 2009; Bergeron et al., 2004, 2010; Soja et al., 2007; Wotton et al., 2010), fire season length (Wotton and Flannigan, 1993; Boulanger et al., 2013), and area burned (Flannigan et al., 2005) are expected to increase throughout the boreal forest as the climate warms in the future (IPCC, 2013).

Stand stocking can sometimes be inadequate following fire. Stands can be young when burned and therefore the trees are not mature enough to produce adequate seed (Viglas et al., 2013; Zasada, 1971; Viereck and Johnston, 1990; Gauthier et al., 1993), or pre-fire densities of mature trees can be low; conversely some fires can be so severe that they kill the seeds in the aerial seedbank (Pinno et al., 2013) regardless of pre-fire stand density or age. However, in terms of species composition there are usually no dramatic post-fire changes (Greene and Johnson, 1999; Ilisson and Chen, 2009; Boucher et al., 2014). Natural regeneration density, and by extension stand stocking, can be reduced even further by salvage operations due to the removal of the aerial seedbank (Greene et al., 2006, 2013; Splawinski et al., 2014), and seed mortality within remaining slash piles. It has an especially deleterious effect on *P. mariana* due to its extended abscission (seed release) schedule (Splawinski et al., 2014).

Salvage is typically applied within the first winter following fire to minimize degradation of burnt boles due to wood-boring insects, stain fungi, wood-decay fungi, and checking (cracking/splitting of wood due to wind, freezing, irregular drying, etc.) (St-Germain and Greene, 2009), due to easier accessibility to stands that are located far from an established road network, and less damage to the ground due to freezing and snow accumulation (O'Mahony et al., 2000). These stands can be regenerated artificially through planting (Saint-Germain and Greene, 2009) or, if access is restricted, through aerial seeding, however treatment application is costly. Conversely, when regeneration densities are excessive, pre-commercial thinning can be employed to control stand density, thereby maximizing growth and yield of residual trees and minimizing rotation period (Vassov and Baker, 1988).

Due to limited understanding of post-salvage regeneration processes, managers and foresters currently lack the tools needed to assess reforestation needs following fire and salvage at the landscape level. We know of only two other spatial models that examine regeneration following fire in the boreal forest. The first, Valeria et al. (2011) is conceptual in nature, does not include salvage logging, and model projections are expressed as: good, average, low natural regeneration or not predicted. It qualitatively estimates the capacity of the expected regeneration in boreal forests following fires based on the scientific literature and research conducted by the Natural Sciences and Engineering Research Council of Canada (NSERC) – Université du Québec en Abitibi-Témiscamingue (UQAT) – Université du Québecà Montréal (UQAM) Industrial Chair in Sustainable Forest Management (SFM Chair) and the Canadian Forest Service (CFS) over fifteen years. The second, the Canadian fire effects model (CanFIRE) is based primarily on fire behavior, and simulates stand-level physical and ecological effects of fire (de Groot, 2012). It also does not include a salvage-logging component, and natural regeneration is calculated as a function of pre-fire stand density and age, and fire severity, and not seed production. Conversely, existing empirical stand-level models and tools (ex. Splawinski et al., 2014, 2015) although important, are limited to the local scale, and thus prove impractical when the goal is to quickly assess regeneration needs over large burned areas. Indeed, the stand-level tool presented by Splawinski et al. (2015) requires some fieldwork in order to estimate regeneration density.

The development of landscape-level assessment tools that can be used remotely to project regeneration density following fire and salvage is undoubtedly necessary. Such tools will improve the planning and execution of salvage operations both spatially and temporally, and subsequent silvicultural treatment application through the identification of regions and individual stands where natural regeneration density will be inadequate or excessive. Foresters can then estimate the potential cost of future interventions such as mechanical site preparation, planting, aerial seeding, and pre-commercial thinning. They can also assess the vulnerability of management units to the current fire regime, and can identify regions at particular risk.

The primary goal of this study is to illustrate how a tool that can project natural regeneration density can be used to define whether artificial regeneration will be required following a fire, and to help in assessing optimal salvage location and timing in order to increase regeneration success. This will be accomplished by using the Lake Matagami lowland ecological region (6a) in the boreal forest of Quebec as a case study. The objectives are therefore fourfold: (1) to outline how provincial forest datasets can be used to determine the parameters required by Splawinski et al. (2014) natural regeneration model, (2) to use the model along with Quebec forest inventory maps and temporary plots to simulate natural regeneration densities in pure *P. mariana* and *P. banksiana* stands following moderate to severe fire and salvage logging (traditional first winter salvage and delayed salvage) at the landscape-level (here we use Lake Matagami lowland ecological region (6a) in the boreal forest of Quebec as a case study); (3) develop recommendations on silvicultural practices based on simulated seedling densities; and (4) transfer simulated results onto GIS forest inventory maps.

2. Methodology

2.1. Case study area

The Quebec Ministry of Forests, Wildlife and Parks (formerly the Ministry of Natural Resources) employs a hierarchical ecological classification system that sub-divides territories based on ecological factors from the continental to local scale. The Lake Matagami lowland ecological region (6a) is located in the western part of the managed continuous boreal forest of Quebec, and represents part of the western portion of the spruce–moss bioclimatic domain (Fig. 1) (Blouin and Berger, 2005; Bergeron et al., 1998). It covers an area of 48,231 km^2, and is dominated by *P. mariana*, *P. banksiana*, and balsam fir (*Abies balsamea* (L.) Mill.) forest, interlaced with bogs, rivers, and lakes (Blouin and Berger, 2005; Bergeron et al., 1998). The regional climate of the western portion of the spruce–moss bioclimatic domain is sub-polar sub-humid continental, with approximately 825 mm of total precipitation per year; the growing season lasts four to five months and average annual temperature ranges from 0 to −2.5 °C (Blouin and Berger, 2005; Bergeron et al., 1998). The landscape of ecological region 6a is dominated by glaciolacustrine clays and sands, Cochrane tills, glacial tills, and organic surficial deposits; underlain by crystalline rock of granitic and volcanic origin (Blouin and Berger, 2005; Bergeron et al., 1998). Our study area makes up 43,572 km^2 or 90% of ecological region (6a). Forest inventory data in the remaining area was not available for simulation.

2.2. Simulation model

The regeneration model developed by Splawinski et al. (2014) simulates natural regeneration densities (seedlings/m^2) of *P. mariana* and *P. banksiana*, following fire and salvage at the stand level in the first 6 years following fire, here defined as the establishment phase. Data from provincial forest inventories and the field will be used to define model parameters. Simulated seedling

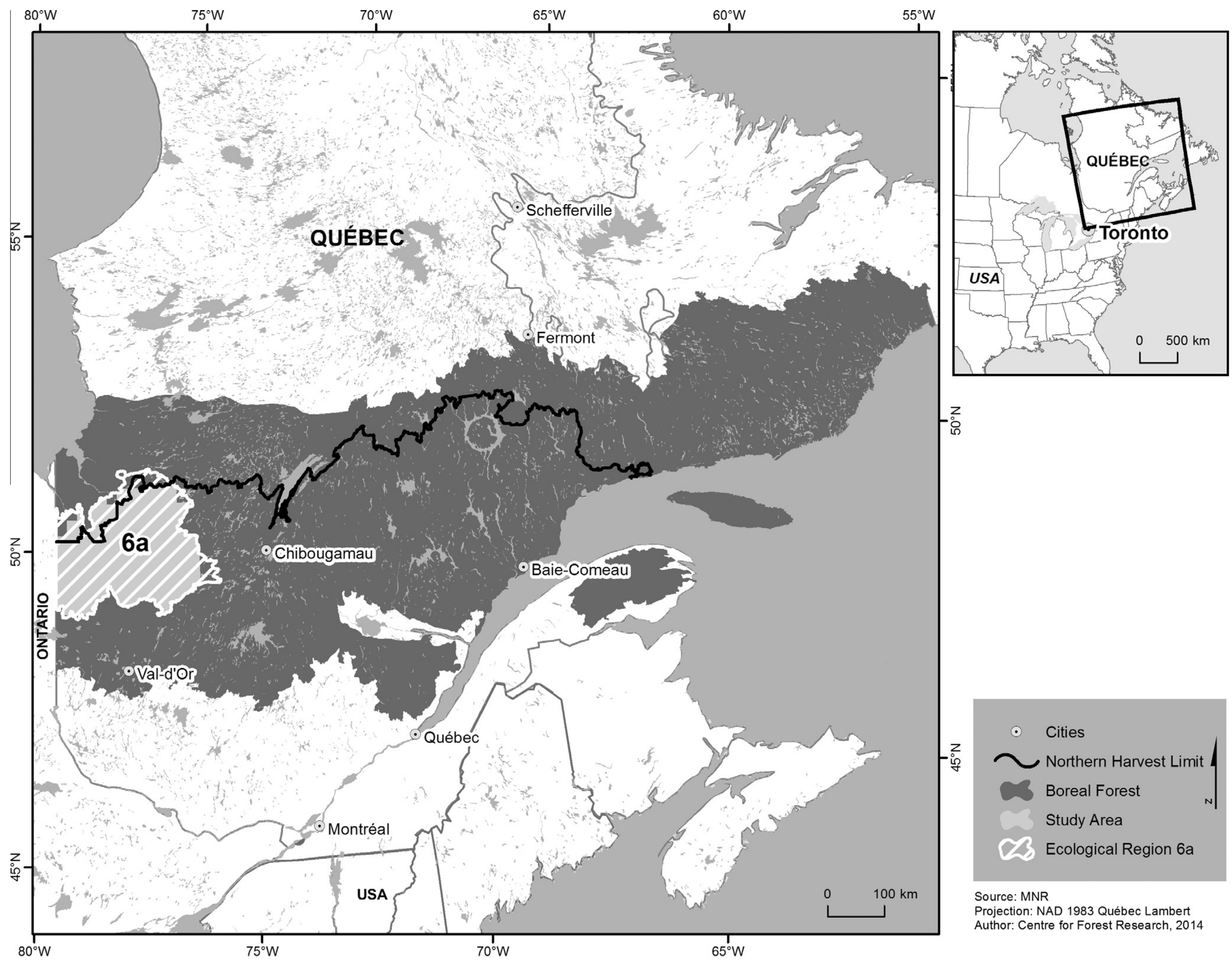

Fig. 1. Map of Quebec illustrating Lake Matagami lowland ecological region (6a) with study area, the boreal forest, and the northern harvest limit.

densitites will then be attributed a silvicultural recommendation, with results being transposed onto GIS-based forest polygons thereby allowing regeneration assessment at the landscape level (Fig. 2). The regeneration model does not simulate stand variability in seedling density, but rather a single density value, therefore stocking distribution cannot be inferred. Silvicultural recommendations are thus based on this single simulated density.

The model has been parameterized based on: (1) initial seed availability as a function of pre-fire tree basal area (m^2/m^2), survival of seeds through the passing of the flaming front, and salvage proportion in terms of basal area removed; (2) seed abscission as a function of time; (3) seedling survivorship as a function of seed mass, the proportion of good, poor, and lethal seedbeds, and granivory; and (4) seedling and seed mortality as a function of salvage operations (Fig. 2). It is most sensitive to the parameters of seeds survival through the passing of the flaming front, and granivory. The regeneration model was successfully validated in Splawinski et al. (2014) by comparing simulated vs. observed seedling densities obtained from 3 wildfires under burned intact and 100% salvaged scenarios. For the purpose of this study both the simulated seedling density and the tree basal area parameter will be converted to seedlings/ha and m^2/ha respectively, as this represents the standard used by foresters in the field.

For model simulations, two parameters are required: (1) pre-fire basal area to calculate initial seed availability, and (2) seedbed proportions to calculate seedling germination and survival. Stand-level data can be extracted from available provincial and national forest inventories, and permanent/temporary sample plots (ex. Gillis et al., 2005; Ministère des Ressources naturelles, 2007, 2009). The remaining parameter values (seed abscission schedules, seed mass, granivory rate, seedling and seed mortality and seed removal as a function of salvage operations, and seasonal availability of seeds for germination) were taken from Splawinski et al. (2014).

2.3. Field data

For our study area, data sourced from both forest inventory maps and temporary and permanent plots (Lord and Faucher, 2003; Ministère des Ressources naturelles, 2007, 2009; Pelletier et al., 2007) was used to estimate the pre-fire basal area and seedbed proportion parameters, thus allowing simulation of the natural regeneration density following both fire and salvage, and the creation of a landscape tool for Lake Matagami lowland ecological region (6a) (Fig. 2).

Temporary plots are sampled on-site across the boreal forest of Quebec during each inventory period, which is carried out roughly every decade. Each plot is 400 m^2 (circular, sampled using an 11.28 m radius). It provides stand-level details on species composition, DBH, age, stem density, height, surficial deposit, and drainage (i.e. moisture regime) (Ministère des Ressources naturelles, 2007). The 3rd inventory period was completed in

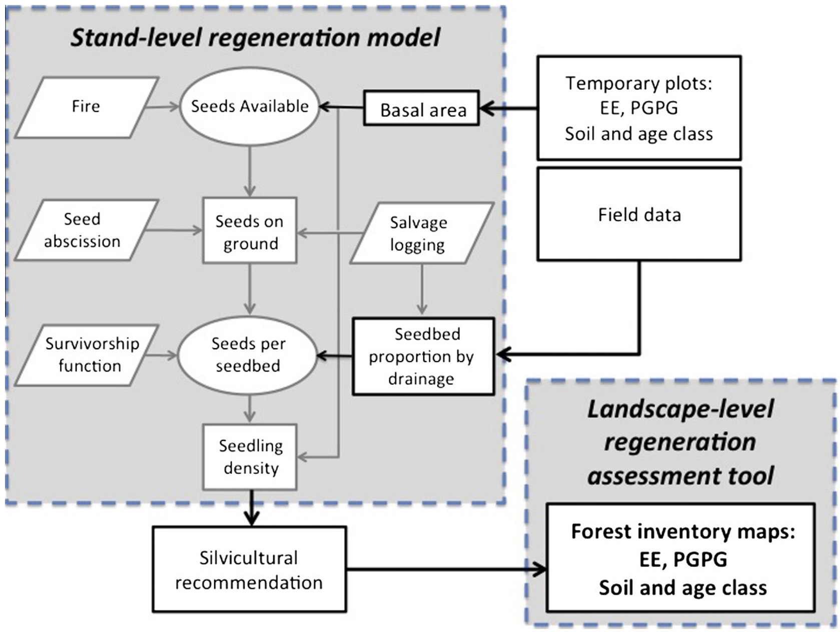

Fig. 2. Simplified structure of the modeling effort for both pure *Picea mariana* (EE) and *Pinus banksiana* (PGPG), showing driving parameters (parallelograms), processes (ellipses), stocks (squares), and external data inputs.

2002 (Ministère des Ressources naturelles, 2008). The 4th inventory is currently underway, however it has not yet been completed, thus the data is as yet unavailable.

In order to obtain estimates of the basal area/ha for stands of a given composition, we used the 3rd inventory temporary plot data (pure *P. banksiana* (PGPG) ($n = 191$) and pure *P. mariana* (EE) ($n = 2423$) stands); we subsequently refer to these species with these codes. In order to be classified as a pure stand, each species must make up at least 75% of the stand basal area. The surficial deposits and drainage observed in these 2623 temporary plots were grouped into soil classes (Table 1), using the same classification scheme as Valeria et al. (2011). This was done as surficial deposit and drainage will have an effect on both forest composition (Gauthier et al., 1996), stand productivity, and tree growth (Bonan and Shugart, 1989; Rudolph and Laidly, 1990; Beland and Bergeron, 1996; Harper et al., 2002). Deposit and drainage codes were developed by the Quebec Ministry of Forests, Wildlife and Parks. Surficial deposits are classified based on texture, composition, origin, and morphology; drainage is classed based on both the horizontal and lateral drainage potential, ranging from excessive to very poor (Ministère des Ressources naturelles, 2009).

2.4. Stand basal area

The mean and lower and upper 95% confidence interval for observed basal areas was obtained for each soil class, further subdivided by stand cover class (Table 2). These cover classes were developed by the Quebec Ministry of Forests, Wildlife and Parks, and represent the % of forest crown cover in a stand (Ministère des Ressources Naturelles, 2009). Cover class A represents >80%, B represents 60–80%, C represents 40–60%, and D represents 25–40%. Some cover classes were labeled N/A (not available) as their sample size in the temporary plot dataset was too small to determine basal area. A total of 2623 temporary plots were used to determine basal area/ha, 2431 for *P. mariana* and 192 for *P. banksiana*. All stems of *P. mariana* and *P. banksiana* sampled within these plots were included in the analysis. Details on temporary plot sample size can also be found in Table 2.

Stand age classes were grouped as follows: For *P. mariana* all stands with an age class of $\geqslant$50 years were grouped into the mature category, whereas for *P. banksiana* we used $\geqslant$30 years (Table 1). This follows the argument of (Zasada, 1971; Viereck and Johnston, 1990; Gauthier et al., 1993; Viglas et al., 2013) which indicates that below these respective ages adequate seed production will not be achieved by either species regardless of the stand basal area. Stands identified as young uneven-age (JIN) or old-growth (>120 year age class) (VIN) were excluded from the analysis, as the former does not achieve adequate growth/volume to warrant harvesting or salvage (Ministère des Ressources naturelles, 2003), whereas seed production is expected to decline in the latter due to old age, senescence, and gap formation (Van Bogaert et al., 2015). ANOVA was used to compare the mean basal area/ha by soil class for both *P. mariana* and *P. banksiana* to make sure the grouping scheme was done correctly and would prove useful; levels of significant factors were compared using Tukey's HSD post hoc method (Table 2). A α value of 0.05 was used for all statistical analyses.

2.5. Seedbeds and drainage types

Expected seedbed proportions following fire were determined for 3 drainage types: xeric, mesic, and hydric (Table 3) by combining available data from 2 sources: The first being the seedbed proportions used in the original model; details on methodology can be found in Splawinski et al. (2014). This included (1) the 2005 Lebel-sur-Quevillon (Quebec) wildfire, and (2) the 1997 Val Paradis (Quebec) wildfire. The second source comes from unpublished fieldwork conducted in the 2010 La Tuque wildfire (Quebec), following the same methodology used at the Lebel-sur-Quevillon wildfire. Seedbeds were divided into classes: good (mineral soil, living mosses); poor i.e., high porosity (residual duff, thick layer of leaves, dead mosses, lichens); and lethal (exposed rocks, firm logs, charred logs, and standing puddles) Splawinski et al. (2014).

The drainage types were then associated with each soil class (Table 4) thereby providing an expected seedbed proportion. Sand and shallow rock were considered xeric; till, Cochrane till, and mesic clay were considered mesic; and lastly, hydric clay and organic were considered hydric. Since seedbed data was unavailable for the organic soil class, for the purpose of the simulations we used the hydric seedbed proportions.

Table 1
Grouping of deposits and drainage into soil class, and age class for *Picea mariana* and *Pinus banksiana*, Lake Matagami lowland ecological region (6a).

Species	Soil class	Sample size (*n*)	Age class	Deposit code	Drainage code
Picea mariana	Sand	225	⩾50 years	2A, 2AE, 2AK, 2B, 2BD, 2BE, 3AC, 3AE, 3AN, 4GS, 4GSM, 4GSY, 4P, 5S, 5SM, 5SY, 6S, 6SM, 6SY, 9, 9S, 9A	N/A
	Shallow rock	36	⩾50 years	R, R1, R1A, R4A, R4GA, R4GS, R5S, R6S, R7T, R1AA, M1AA, M1A, 1AR, 8A	N/A
	Till	1575	⩾50 years	1A, 1AD, 1AM, 1AY, 1AB, 1BD, 1BF, 1BG, 1BI, 1BIM, 1BIY, 1BP, 1BT, 1Y	N/A
	Cochrane till	92	⩾50 years	1AA, 1AAM, 1AAY, 1AAR	N/A
	Mesic clay	28	⩾50 years	4A, 4GA, 4GAM, 4GAR, 4GAY, 5A, 5AM, 5AY	00, 10, 13, 16, 20, 21, 30, 31, 33
	Hydric clay	209	⩾50 years	4A, 4GA, 4GAM, 4GAR, 4GAY	40, 41, 43, 50, 51, 53, 54, 60, 61, 62, 63
	Organic	266	⩾50 years	7, 7E, 7T, 7TM, 7TY	N/A
Pinus banksiana	Sand	118	⩾30 years	2A, 2AE, 2AK, 2B, 2BD, 2BE, 3AC, 3AE, 3AN, 4GS, 4GSM, 4GSY, 4P, 5S, 5SM, 5SY, 6S, 6SM, 6SY, 9, 9S, 9A	N/A
	Till	74	⩾30 years	1A, 1AD, 1AM, 1AY, 1AB, 1BD, 1BF, 1BG, 1BI, 1BIM, 1BIY, 1BP, 1BT, 1Y	N/A

2.6. Simulation scenarios

Three scenarios of natural regeneration density of *P. mariana* and *P. banksiana* following moderate to severe fire were simulated: (1) in burned intact stands, (2) in stands subjected to 100% salvage logging in the first winter following fire (traditional method), and (3) in stands subjected to 100% salvage logging in the 2nd through 4th winter following fire (delayed salvage). Here we define moderate and high severity fire as resulting in 100% stand mortality (not necessarily a function of intensity). We simulate a June fire date, since the majority of fires in the boreal forest burn in late spring (Stocks et al., 2002).

Salvage operations are typically carried out the first winter following fire. For the purpose of the simulations, we will employ a December salvage date for all salvage scenarios, representing the median month as seen in the Lebel-sur-Quevillon fire from Splawinski et al. (2014). Simulations were carried out using the mean basal area, and for the upper and lower 95% confidence interval value (Table 2), thereby providing seedlings/ha for each combination.

2.7. Transfer of results onto forest inventory maps

Quebec forest inventory maps provide details on forest classes subdivided by: species composition, stand cover, age class, height, forest layers, stand origin (i.e. fire, partial burn, clearcut, insect outbreak, plantation), slope class, surficial deposit, and drainage (Lord and Faucher, 2003). Maps are generated through the interpretation of aerial photos at a scale of 1:15,000 (Lord and Faucher, 2003).

Resulting seedling densities obtained for both species under all five scenarios were transferred onto existing forest inventory maps of the 3rd decadal inventory using GIS (here ESRI® Arc GIS 10.1). First, forest types (EE or PGPG), cover, age, and soil class were identified for the Lake Matagami lowland ecological region (6a) of the boreal forest of western Quebec following the classification scheme found in Tables 1 and 2. The Splawinski et al. (2014) regeneration model was then used to simulate seedling densities based on each basal area found in Table 2, including the mean basal area, as well as the lower and upper 95% confidence limit values. The resulting seedling densities were then classed under a silvicultural recommendation (color coded) and attributed to each representative forest polygon. The following 3 silvicultural recommendations (based on the resulting seedling density) were used: (1) Planting required: fully re-stocked stands are generally obtained with a seedling density ⩾10,000/ha (⩾1 seedling/m^2) (Greene et al., 2002), therefore any stands that exhibit seedling densities below this threshold need to be planted in the future. This silvicultural recommendation was further broken down into 2 categories: (a) planting required regardless of salvage, representing stands that did not achieve adequate stand stocking due either to inadequate pre-fire basal area or stand cover; and (b) planting required due to salvage, representing stands that exhibited adequate stand stocking under the burned intact scenario but subsequently needed to be planted under any of the 100% salvage scenarios. (2) Adequate natural regeneration and potential pre-commercial thinning: Any stands that exhibit seedling densities at or above this threshold will not require planting as stand stocking should be considered as sufficient; conversely, since these stands exhibit a seedling density ⩾10,000/ha, they may need to be thinned at some point in time in order to increase growth rates in residual trees (Morris et al., 1994). (3) Planting required due to stand immaturity: *P. mariana* stands younger than 50 years and *P. banksiana* stands younger than 30 years were not simulated (Table 1), as they will not produce adequate seed stock given their young age (Zasada, 1971; Viereck and Johnston, 1990; Gauthier et al., 1993;

Table 2
Sample size, basal area m^2/ha (mean, lower and upper 95% confidence limit-CL) for *Picea mariana* (EE) and *Pinus banksiana* (PGPG) by soil and cover class. Levels of significant factors were compared using Tukey HSD.

Species	Soil class	Cover class	Sample size (n)	% By cover class	Basal area/ha			Significance Tukey HSD
					Mean	Upper 95% CL	Lower 95% CL	
EE	Sand	A	6	2.7	38.6	41.5	35.7	a
		B	27	12.0	29.9	31.7	28.1	b
		C	96	42.7	21.3	22.7	20.0	c
		D	96	42.7	14.7	15.7	13.7	d
EE	Shallow rock	A	1	2.8	N/A	N/A	N/A	N/A
		B	3	8.3	N/A	N/A	N/A	N/A
		C	11	30.6	17.7	21.9	13.4	a
		D	21	58.3	12.3	14.5	10.0	a
EE	Till	A	25	1.6	37.5	39.9	35.2	a
		B	238	15.1	30.2	31.1	29.3	b
		C	655	41.6	21.2	21.6	20.7	c
		D	657	41.7	14.2	14.6	13.8	d
EE	Cochrane till	A	1	1.1	N/A	N/A	N/A	N/A
		B	16	17.4	32.2	35.4	29.1	a
		C	52	56.5	21.8	23.2	20.4	b
		D	23	25.0	14.7	17.4	11.9	c
EE	Mesic clay	A	0	0	N/A	N/A	N/A	N/A
		B	13	46.4	32.8	35.7	29.8	a
		C	11	39.3	24.5	29.5	19.5	b
		D	4	14.3	19.2	27.5	10.9	b
EE	Hydric clay	A	2	1.0	N/A	N/A	N/A	N/A
		B	30	14.4	30.8	33.1	28.5	a
		C	90	43.1	22.1	23.3	20.8	b
		D	87	41.6	15.8	17.0	14.6	c
EE	Organic	A	1	0.4	N/A	N/A	N/A	N/A
		B	15	5.6	28.8	33.1	24.6	a
		C	109	41.0	20.8	21.9	19.6	b
		D	141	53.0	13.7	14.5	12.8	c
PGPG	Sand	A	0	0	N/A	N/A	N/A	N/A
		B	11	9.3	18.5	21.3	15.6	a
		C	62	52.5	13.5	14.6	12.4	b
		D	45	38.1	8.9	10.0	7.7	c
PGPG	Till	A	1	1.4	N/A	N/A	N/A	N/A
		B	10	13.5	18.0	20.6	15.5	a
		C	37	50.0	14.4	16.0	12.8	a
		D	26	35.1	8.2	9.5	6.9	b

Table 3
Seedbed proportions by drainage types.

Drainage	Seedbed proportion (%)			
	Mineral soil	Moss	High porosity	Lethal
Xeric	9	1	85	5
Mesic	1	2	87	10
Hydric	0	9	81	10

Table 4
Species and the seedbed proportions associated with each soil class.

Species	Soil class	Associated seedbed proportions
Picea mariana	Sand	Xeric
	Shallow rock	Xeric
	Till	Mesic
	Cochrane till	Mesic
	Mesic clay	Mesic
	Hydric clay	Hydric
	Organic	Hydric
Pinus banksiana	Sand	Xeric
	Till	Mesic

Viglas et al., 2013). These stands were automatically considered as having inadequate natural regeneration and thus will require planting in order to achieve adequate stand stocking.

3. Results

3.1. Simulations

Simulated natural seedling regeneration densities/ha for burned intact and 100% salvaged forest (Table 5) varied by soil and cover class. As expected, the higher the cover class, the higher the seedling density for both species. Under all simulated scenarios, *P. mariana* exhibited the greatest seedling densities on the hydric clay soil class, followed by organic, sand, shallow rock, mesic clay, cochrane till, with lowest seedling densities on till. For *P. banksiana* the greatest seedling densities were observed on the sand soil class, followed by till, under all scenarios.

3.2. Burned intact scenario

For *P. mariana* under the burned intact scenario, adequate natural regeneration densities (full stand stocking) on the sand soil class are obtained for cover classes A through C, with planting necessary in cover class D. For shallow rock, data are unavailable for cover class A and B, however based on the simulated densities in the subsequent classes it is safe to assume that adequate natural regeneration densities will be obtained. For cover class C planting may be necessary as the upper CL density exceeds the minimum threshold for planting whereas the lower CL does not; planting will

Table 5
Simulated seedlings/ha (lower to upper 95% confidence limit-CL) for burned intact (BI), and 100% salvaged forest in the first (S1), second (S2), third (S3), and fourth (S4) winter following fire, in pure *Picea mariana* (EE) and *Pinus banksiana* (PGPG) stands by soil and cover class.

Species	Soil class	Cover class	Seedling density/ha (lower CL–upper CL)				
			BI	S1	S2	S3	S4
EE	Sand	A	18,100–20,900	3500–4000	7100–8200	9500–11,000	11,100–12,800
		B	14,400–16,200	2800–3100	5700–6400	7600–8500	8800–9900
		C	10,400–11,800	2000–2300	4100–4600	5500–6200	6400–7200
		D	7300–8300	1400–1600	2900–3300	3800–4400	4500–5100
EE	Shallow rock	A	N/A	N/A	N/A	N/A	N/A
		B	N/A	N/A	N/A	N/A	N/A
		C	7100–11,400	1400–2200	2800–4500	3700–6000	4400–7000
		D	5400–7700	1000–1500	2100–3000	2800–4000	3300–4700
EE	Till	A	11,200–12,600	2100–2400	4400–4900	5900–6600	6800–7700
		B	9400–9900	1800–1900	3700–3900	4900–5200	5700–6100
		C	6700–7000	1300–1300	2700–2800	3500–3700	4100–4300
		D	4600–4800	900–900	1800–1900	2400–2500	2800–3000
EE	Cochrane till	A	N/A	N/A	N/A	N/A	N/A
		B	9300–11,200	1800–2200	3700–4400	4900–5900	5700–6900
		C	6600–7500	1300–1400	2600–3000	3500–3900	4100–4600
		D	4000–5700	800–1100	1600–2200	2100–3000	2400–3500
EE	Mesic clay	A	N/A	N/A	N/A	N/A	N/A
		B	9500–11,300	1800–2200	3700–4400	5000–5900	5800–6900
		C	6400–9400	1200–1800	2500–3700	3300–5000	3900–5800
		D	3700–8800	700–1700	1400–3500	1900–4600	2200–5400
EE	Hydric clay	A	N/A	N/A	N/A	N/A	N/A
		B	21,100–24,400	4100–4700	8300–9600	11,100–12,800	12,900–14,900
		C	15,700–17,500	3000–3400	6200–6900	8200–9200	9600–10,700
		D	11,200–12,900	2100–2500	4400–5100	5900–6800	6900–7900
EE	Organic	A	N/A	N/A	N/A	N/A	N/A
		B	18,400–24,400	3500–4700	7200–9600	9700–12,800	11,200–14,900
		C	14,800–16,500	2800–3200	5800–6500	7800–8600	9100–10,100
		D	9900–11,100	1900–2100	3900–4400	5200–5800	6000–6800
PGPG	Sand	A	N/A	N/A	N/A	N/A	N/A
		B	15,500–20,300	8500–11,100	10,700–14,000	10,800–14,200	10,800–14,200
		C	12,700–14,600	7000–8000	8800–10,100	8900–10,200	8900–10,200
		D	8400–10,600	4600–5800	5800–7300	5900–7400	5900–7400
PGPG	Till	A	N/A	N/A	N/A	N/A	N/A
		B	10,100–12,900	5500–7100	7000–8900	7100–9000	7100–9000
		C	8600–10,400	4700–5700	5900–7200	6000–7300	6000–7300
		D	5000–6600	2800–3600	3500–4600	3500–4600	3500–4700

be necessary for cover class D. On the till soil class, adequate natural regeneration densities are obtained for cover class A, with a potential need for planting in cover class B (the upper CL density exceeds the minimum threshold for planting however the lower CL does not); planting will be necessary in cover class C and D. For both Cochrane till and mesic clay, data are unavailable for cover class A, however we can assume that adequate natural regeneration densities will be achieved; as for cover class B, adequate densities are obtained. Conversely, for cover class C and D planting will be necessary on both soil classes. For both hydric clay and organic soil classes, adequate natural regeneration densities are obtained under all cover classes.

For *P. banksiana* under the burned intact scenario, adequate natural regeneration densities on the sand soil class are obtained for cover class A through C, with a potential need for planting in cover class D as the upper CL density exceeds the minimum threshold for planting whereas the lower CL does not. On the till soil class adequate natural regeneration densities are obtained for cover class A and B, with a potential need for planting in cover class C (the upper CL density exceeds the minimum threshold for planting however the lower CL does not), and a definite need for planting in cover class D.

3.3. Traditional salvage logging

Turning to the first winter salvage scenario, planting will be necessary for *P. mariana* on all soil and cover classes. For *P. banksiana*, adequate natural regeneration densities on sand are obtained for cover class A, with a potential need for planting in cover class B (the upper CL density exceeds the minimum threshold for planting however the lower CL does not). Planting will be necessary on both the C and D cover classes. On the till soil class planting will be necessary in all cover classes. Although data on cover class A are unavailable for both deposits, we can safely assume expected densities based on simulated densities in the subsequent classes i.e., on sand, adequate natural regeneration should be obtained in cover class A, given how high the densities are in cover class B; conversely on till given the low density in cover class B, planting will most likely be necessary in cover class A.

3.4. Delayed salvage logging

If salvage is delayed until the second winter following fire, *P. mariana* will obtain adequate natural regeneration densities on both the hydric clay and organic soil class for cover class A (data are unavailable for this class, however based on the simulated densities in the subsequent classes it is safe to assume that adequate natural regeneration densities will be obtained). All other soil and cover classes will require planting.

If salvage is delayed until the third winter, adequate natural regeneration densities are obtained on sand for cover class A, and on the hydric clay and organic soil class for cover class A and B. Planting may be required on shallow rock for cover class A since the upper CL density exceeds the minimum threshold for planting however the lower CL does not (this is inferred from subsequent class densities as data is unavailable for this cover class). All other soil and cover classes will require planting.

If salvage is delayed until the fourth winter, adequate natural regeneration densities are obtained on sand for cover class A, on the hydric clay for cover class A, B, and C, and on the organic soil class for cover class A, and B. Planting may be required on shallow rock for cover class A, and on the organic soil class for cover class C, since the upper CL densities exceed the minimum threshold for planting however the lower CL densities do not.

Turning to *P. banksiana*, if salvage is delayed until the second winter following fire, adequate natural regeneration densities are obtained on sand for cover class A and B (data is unavailable for cover class A, however based on the simulated density in cover class B, it is safe to assume that adequate natural regeneration densities will be obtained). Planting may be required on sand for cover

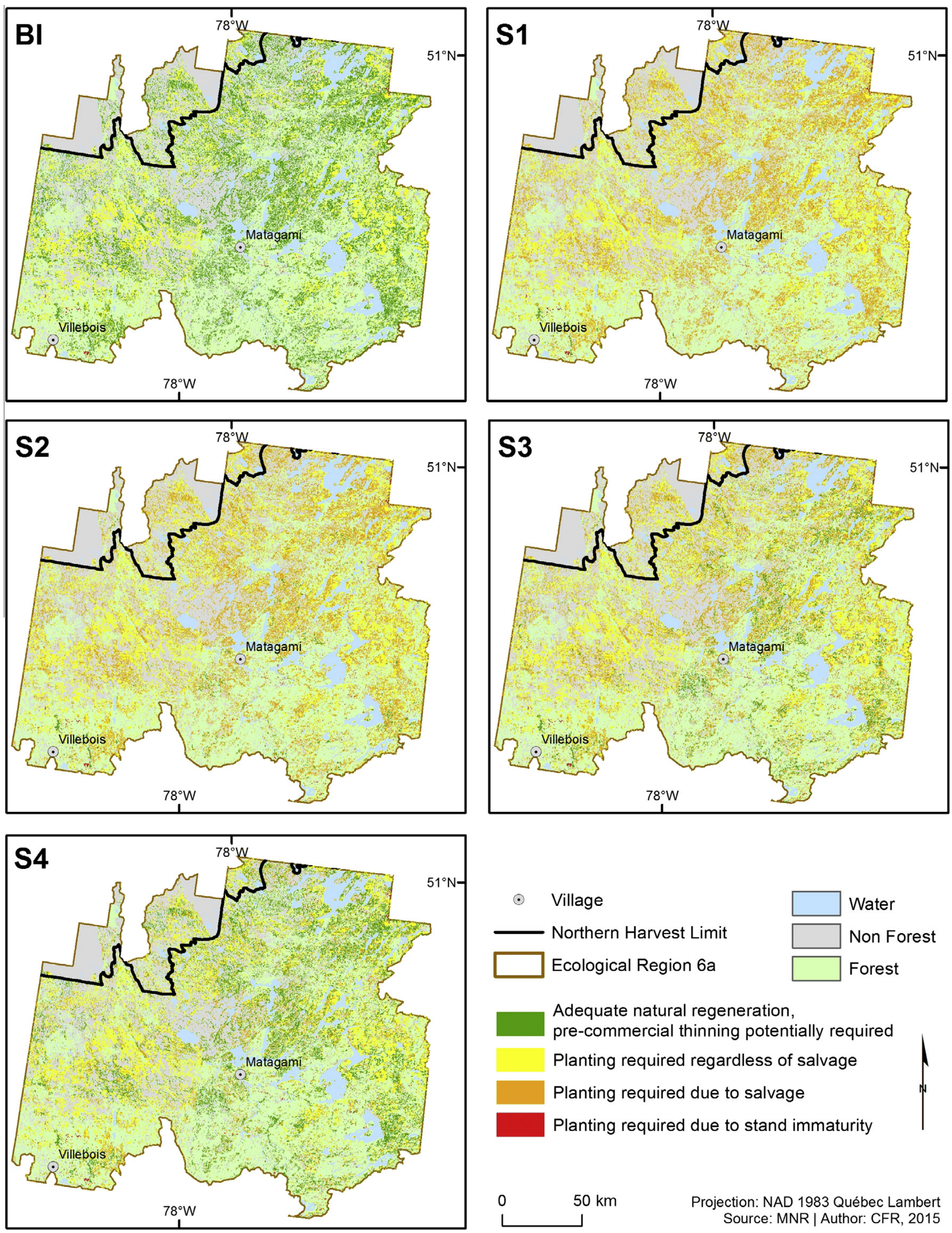

Fig. 3. Spatial assessment tool with silvicultural recommendations for all pure *Picea mariana* and *Pinus banksiana* stands used in the burned intact (BI) and 100% salvaged scenarios in the first (S1), second (S2), third (S3), and fourth (S4) winter following fire, for ecological region (6a). Light green represents all other forest types not simulated; these include pure coniferous stands excluding *Picea mariana* and *Pinus banksiana*, as well as mixed coniferous and deciduous, and pure deciduous stands.

class C and on till for cover class A, since the upper CL density exceeds the minimum threshold for planting however the lower CL and mean do not (due to a lack of data for cover class A on till, is inferred from subsequent class densities). All other soil and cover classes will require planting. There is no discernable change if salvage is delayed until the third or fourth year following fire.

Table 6
Percent of stands (%) by scenario for each silvicultural recommendation by species.

Species	Silvicultural recommendation	Percent (%) of simulated stands by scenario				
		BI	S1	S2	S3	S4
EE	Adequate natural regeneration	64.74	0	1.72	11.6	34.38
	Planting required	35.26	100	98.28	88.4	65.62
	Planting required due to stand immaturity (excluded)	2.3	2.3	2.3	2.3	2.3
PGPG	Adequate natural regeneration	93.83	26.22	34.98	44.08	44.08
	Planting required	6.17	73.78	65.02	55.92	55.92
	Planting required due to stand immaturity (excluded)	0.08	0.08	0.08	0.08	0.08

* Total number of stands simulated by species: *Picea mariana* (EE) $n = 81{,}543$, *Pinus banksiana* (PGPG) $n = 1233$.

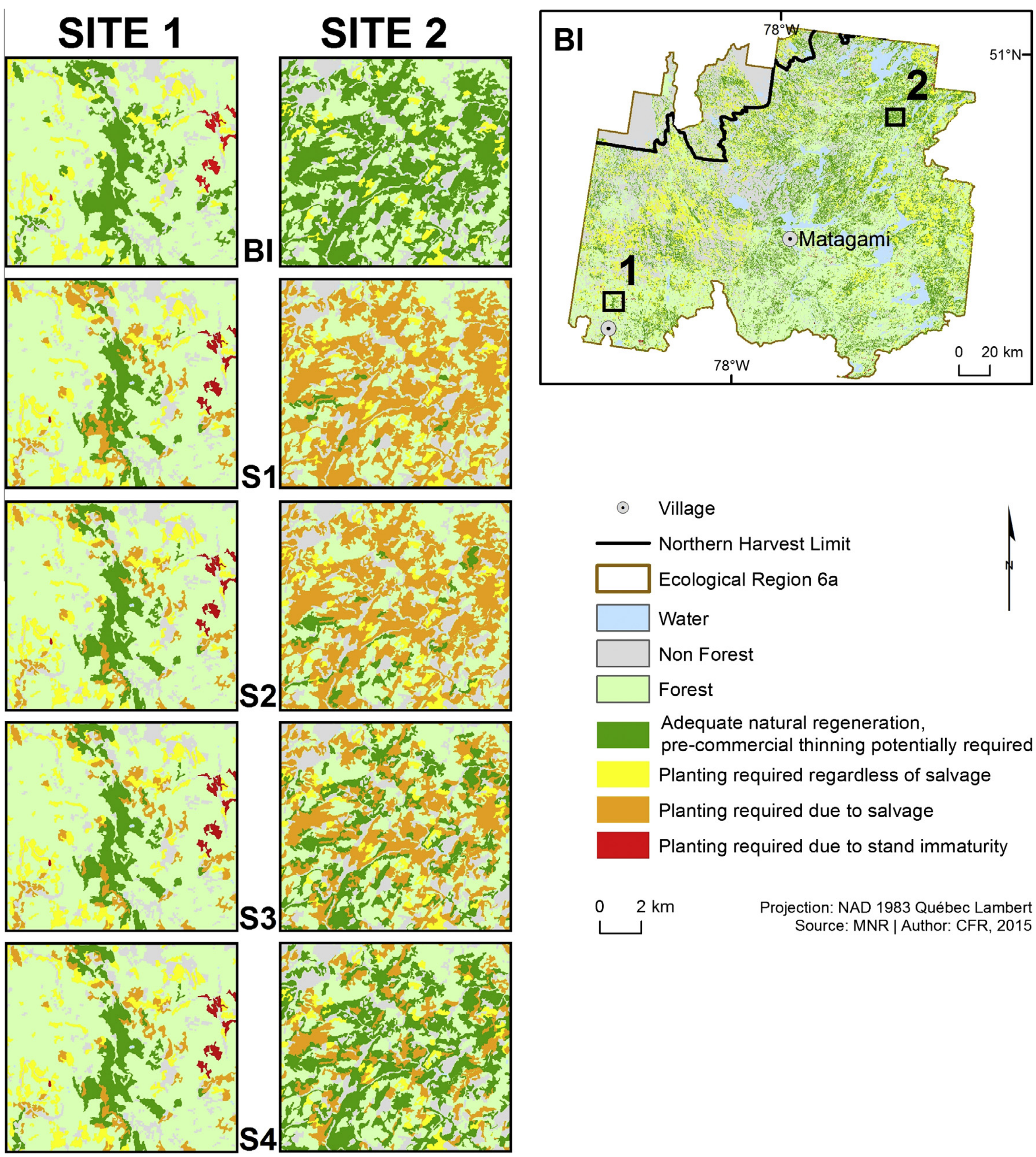

Fig. 4. Greater detail for two regions within ecological region (6a) illustrating silvicultural recommendations in the burned intact (BI) and 100% salvaged scenarios in the first (S1), second (S2), third (S3), and fourth (S4) winter following fire. Site 1 is dominated by *Pinus banksiana* stands (PGPG), whereas site 2 is dominated by *Picea mariana* stands (EE). Light green represents all other forest types not simulated; these include pure coniferous stands excluding *Picea mariana* and *Pinus banksiana*, as well as mixed coniferous and deciduous, and pure deciduous stands.

3.5. Landscape-level forest regeneration maps

The simulated pure *P. mariana* and *P. banksiana* stands make up 26% of the study area in the Lake Matagami ecological region (6a). Non-simulated areas represent other forest types, including both pure and mixed coniferous stands (8%), mixed coniferous–deciduous stands (7%), and deciduous stands (3%). Recent ($\leqslant$30 year age class) clear-cuts and CPRS without an associated forest type (due to their young age) represent 10% of other forest area, with other disturbances (including recent fires) representing 5%. Water represents 8%, and non-forest (mostly bogs) 33%.

Results were applied to 81,543 pure *P. mariana* and 1233 *P. banksiana* stands, representing an area of 12,102 km^2 and 210 km^2, respectively, in our study area, for a total of 82,776 stands (Fig. 3). For *P. mariana* under the burned intact scenario (excluding immature stands which make up 2% of the total), planting is required in 35% of stands whereas adequate natural regeneration densities are obtained in 65% of stands. For *P. banksiana* (also excluding immature stands which make up 0.08% of the total), planting is required in 6% of stands whereas adequate natural regeneration densities are obtained in 94% of stands (Table 6).

For *P. mariana* under the traditional first winter salvaged scenario (excluding immature stands), planting is required in 100% of stands. For *P. banksiana* (also excluding immature stands), planting is required in 74% of stands whereas adequate natural regeneration densities are obtained in 26% of stands (Table 6).

Under the delayed salvaged scenario (excluding immature stands) in the second, third and fourth year following fire, *P. mariana* will require planting in 98%, 88%, and 66% of stands respectively. For *P. banksiana* (also excluding immature stands), planting will be required in 65%, 56%, and 56% of stands respectively (Table 6).

Fig. 3 illustrates that three areas within ecological region (6a) are vulnerable to fire (i.e. less likely to regenerate naturally): middle-west, middle-east, and north-east, as they contain the greatest density of stands requiring planting. Virtually all stands are vulnerable to fire under the traditional first winter salvaged scenario, except for the densest pre-fire *P. banksiana* stands found on sand (Table 5), and will require planting (Figs. 3 and 4 and 100% first winter salvage scenario, yellow, orange, and red). Delaying salvage reduces the vulnerability of stands to regenerate adequately following fire (Figs. 3 and 4 and 100% second through fourth winter salvage scenarios, yellow, orange, and red).

4. Discussion

4.1. Ecological and economic considerations

Both *P. mariana* and *P. banksiana* are well adapted to fire. As expected, post-fire regeneration densities are greatest in stands that exhibit higher pre-fire tree densities; conversely, it illustrates how salvage operations have a negative impact on the regeneration potential of *P. mariana*, and to a lesser extent, *P. banksiana*, due to the removal of the aerial seedbank. Salvage operations that de-limb boles and leave the cone-bearing branches near the stump (cut-to-length system) rather than in slash piles by skidding to landings (tree-length system) typically still create piles of branches. While the cones on these branches will eventually open, there will be little lateral movement of the winged seeds so close to the ground due to very low wind speeds (Greene and Johnson, 1996), and even lower among the branches; i.e., the dispersal potential is essentially eliminated. Further, many of the seeds will fall on fresh wood (a lethal substrate) or germinate on an otherwise unsuitable microsite in the dense shade created by the tangle of branches. Thus, the seeds in these cones can add little to stand stocking.

In *P. mariana* stands, delaying salvage logging until the third or fourth winter following fire will improve natural regeneration densities, thereby reducing the need for planting, especially in the densest cover classes. This is well illustrated in both Figs. 3 and 4 (Site 2), where the most marked changes occur with the greatest delay in salvage logging. Such a delay could translate into significant savings for those responsible for reforestation, when one considers the relatively high cost of planting (approximately CAD 800 $/ha) (St-Germain and Greene, 2009).

Conversely for *P. banksiana*, due to its rapid abscission schedule (Greene et al., 2013; Splawinski et al., 2014), a delay in salvage of more than two years does little to improve natural regeneration densities. Indeed, salvage operations carried out in the first or second winter following fire in the densest cover classes may reduce the need for intensive pre-commercially thinning in the future, by decreasing stem densities to more manageable levels. This decrease can be attributed to the mortality of advanced regeneration as a result of the salvage operation itself.

Delaying salvage logging in order to maximize natural regeneration will reduce the need to plant, however any savings incurred with such a delay may be offset by the reduced value of wood due to various degradation agents (St-Germain and Greene, 2009). Damaged wood may no longer be suitable for sawlogs, but for example, may instead be used for bioenergy production (Munroe, 2014) or pulp (St-Germain and Greene, 2009). Due to the absence of studies exploring this comparison, we hesitate to recommend a delay based purely on economic reasons. From a biodiversity conservation perspective however, such a delay would greatly benefit pyrophilous avian and arthropod species such as certain woodpeckers and saproxylic beetles, which are reliant on, and abundant in recently burned stands (Lindenmayer and Noss, 2006; Russell et al., 2007; Saab et al., 2009; St-Germain and Greene, 2009; Cobb et al., 2011).

4.2. Silvicultural recommendations

Here we prescribe planting when natural regeneration densities are <10,000 seedlings/ha, under both burned intact and 100% salvaged scenarios. This does not mean that stands under the burned intact scenario are not regenerating properly following fire; indeed in terms of species composition and final stem density there are no dramatic post-fire changes (Greene and Johnson, 1999; Ilisson and Chen, 2009; Boucher et al., 2014), therefore we would not expect a stand with low density prior to fire to regenerate at high density following fire. The planting recommendation is based on suitable density for future harvesting; that is, producing the maximum volume/ha that can be harvested under the next rotation period.

In Quebec, pre-commercial thinning is generally applied to stands with a density of at least 4000 stems/ha (Ministère des Ressources naturelles, 1999). Here we use a density $\geqslant$10,000 seedlings/ha ($\geqslant$1 seedling/m^2) as a threshold for both the thinning and planting recommendations. As these stands are regenerated naturally, seedlings will establish in greater density on more favorable seedbeds, which occupy a relatively smaller proportion of total available seedbeds in a stand. This means that stem density will not be homogenously distributed across the stand. Since a natural regeneration density of 10,000 seedling/ha should yield full stand stocking (Greene et al., 2002), it would not make sense to prescribe thinning in stands with 4000 seedlings/ha (0.4 seedlings/m^2).

Natural regeneration densities are constrained by the availability of suitable seedbeds, which are generally lower in eastern Canada than the west following fire due to thicker post-fire organic layer depths (Greene et al., 2006, 2007; Splawinski et al., 2014). In xeric stands, the thin organic layer is reduced significantly by smoldering combustion to expose mineral soil, however few mosses survive; in mesic stands, the thicker organic layers yield far less mineral soil but slightly more mosses; in hydric stands mineral soil is not exposed however many more mosses survive (Table 3). If the good seedbeds (mineral soil and moss) are grouped, xeric and hydric stands exhibit similar proportions, whereas in mesic stands the proportion is significantly lower (Table 3). Sphagnum, which is typically found in hydric and organic sites, is considered a very good seedbed for germination due to its water holding capacity; conversely, it is considered a poor growing medium as it has a poor nutrient status (Lavoie et al., 2007). Our approach examines establishment density and not subsequent growth. We point out however, that paludified mesic and hydric stands containing large amounts of sphagnum may need to be mechanically scarified and planted even when natural regeneration densities are adequate in order to expose the underlying surficial deposit, thereby improving growth rates (Lafleur et al., 2011; Henneb et al., 2015). Delaying salvage in order to maximize natural regeneration may therefore not be suitable in such stands. Conversely in sphagnum bogs where the organic layer is deeper than the effective range of the scarifier, scarification and subsequent planting may not improve growth rates, therefore a reliance on the natural regeneration, if adequate, may be considered. In our opinion, the latter sites should be avoided altogether due to their unproductive nature; they will maybe never achieve volume that would warrant harvesting in the future.

4.3. Application of the landscape-level assessment tool

Using GIS software, the maps generated under both burned intact and salvage scenarios can be used as an operational assessment tool (Fig. 3) to better plan salvage operations in both time and space, as well as subsequent silvicultural treatment application by identifying stands and regions where natural regeneration density will be expected inadequate or potentially excessive. Foresters and managers could use this tool to determine salvage sequence, estimate potential costs of mechanical site preparation, planting, and aerial seeding in stands that are inaccessible for summer planting, as well as pre-commercial thinning in their management units. Assessment of the vulnerability of management units to fire and the identification of regions at particular risk is also possible, especially if the goal is to maintain or create productive stands that yield maximum potential volume for future harvesting. From an economic perspective, stands that require planting due to either low pre-fire stand density or immaturity are most vulnerable to fire under the burned intact scenario (Fig. 3, burned intact scenario, yellow and red).

4.4. Forest management considerations

P. mariana and *P. banksiana* stands that have not yet reached reproductive maturity (i.e. optimal seed production potential), are most vulnerable to fire, as they do not possess adequate seed stock to successfully regenerate following this type of disturbance (Greene et al., 1999). Current boreal forest management practices attempt to mimic stand-replacing wildfires by employing even-aged management (Bergeron et al., 2002, 2001); indeed it is the dominant management method in Canada (CCFM, 2009). Clearcutting is most widely used under this management scheme (Keenan and Kimmins, 1993; Masek et al., 2011), representing approximately 90% of forest harvested (CCFM, 2009; Masek et al., 2011). Like fire, even-aged management reduces the proportion of older stands as it re-sets succession (Weber and Flannigan, 1997; Bergeron et al., 2001, 2002; Flannigan et al., 2001; McRae et al., 2001). With a typical rotation period of ~100 years, and a fire return interval of ~150 years in eastern Canada (Bergeron et al., 2002), the proportion of younger age-classes will undoubtedly increase in the future. This may be accelerated by the expected increases in fire frequency and area burned (Amiro et al., 2001; Flannigan et al., 2001, 2005, 2009; Bergeron et al., 2004, 2010; Soja et al., 2007; Wotton et al., 2010). With an increase in younger age classes, the susceptibility of management units to poor natural regeneration following fire may increase, requiring more widespread and costly reforestation interventions at the landscape level.

4.5. Model limitations

The Splawinski et al. (2014) model projects stem density per m^2, and not stand stocking. Therefore, although we can make recommendations for stand management practices such as planting and pre-commercial thinning, we cannot elaborate as to how intensive (spatially) these practices may have to be i.e. if a stand will require fill vs. widespread planting, or what % will need to be thinned. This is also why, as previously mentioned, we suggest pre-commercial thinning when *P. mariana* or *P. banksiana* densities $\geqslant$10,000 seedlings/ha.

Only pure coniferous stands are simulated under our current approach, and the model is incapable of including a hardwood component; thus species such as aspen or birch were not included.

4.6. Case study limitations

Aerial photos used to determine forest type, structure, and age are interpreted at a scale of 1:15,000, with a minimum polygon size of 4–8 ha (Ministère des Ressources naturelles, 2009). Given this relatively coarse resolution, discrepancies between stands identified through this method and data collected on-site through temporary plots was sometimes observed (e.g., a forest type identified through aerial photography as pure *P. mariana* with a cover class A was actually labeled *P. mariana* mixed with some *P. banksiana*, with a cover class B using the temporary plot observation). This is an issue of scale, since temporary plots are 400 m^2 in size. The potential future application of LiDAR imagery to perform forest inventory may help resolve this issue, by delineating stands and estimating stand characteristics more accurately (i.e. not relying on a minimum size and aerial photo interpretation, as with the current method) (Hyyppä et al., 2008; Van Leeuwen and Nieuwenhuis, 2010; Olofsson and Holmgren, 2014). Within-stand variability in natural regeneration might then be easier to integrate into the landscape regeneration assessment tool.

The surficial deposits obtained from forest inventory maps are interpreted at the landscape level. Local variations in topography, slope, aspect, and organic layer depth may result in differing drainage patterns. For example even though the regional surficial deposit may be classified as sand, organic matter accumulation at the base of a slope may result in sphagnum bog development, yielding poor drainage at the stand level. This problem illustrates the challenges associated with linking top-down and bottom-up controls on, in this case, forest type, stand cover class, and drainage patterns.

The classification scheme used for grouping deposits, drainage, as well as other attributes such as age (Tables 1 and 2) was necessary given the limited sample size of temporary plots for each forest type (Table 2). Perhaps a better grouping scheme would be possible if a substantially greater sample size for each forest type was available. Indeed, although the model can also be applied to mixed coniferous or coniferous–deciduous stands containing *P. mariana* or *P. banksiana*, the lack of a sufficient temporary plot sample size prevented the pursuit of this objective.

We also associated a particular drainage (in this case for seedbed distribution) to a particular surficial deposit. In Table 4 for example, we associated xeric seedbeds to a deposit of sand, whereas we associated mesic seedbeds to a deposit of till. In reality a deposit of sand can also be mesic whereas a deposit of till can be xeric. In addition, we had to associate the hydric seedbed proportions to the organic deposits due to a lack of seedbed data. In reality we expect organic deposits to have a greater proportion of living sphagnum following fire, therefore we are most likely under-representing simulated results for this deposit.

The species classification scheme used in the 3rd inventory considers a stand as being pure *P. mariana* or *P. banksiana* if these species make up at least 75% of the basal area (Pelletier et al., 2007). This means that hardwoods (<25%) can in fact be present even in stands that are considered ''pure'' coniferous. This has important implications, as aspen and birch also readily regenerate following fire, albeit by basal sprouting and root suckering as opposed to seed from aerial seedbanks (Greene et al., 1999). Indeed, their relative densities can increase in salvaged stands due to more favorable growing conditions (Greene et al., 2006; Boucher et al., 2014).

4.7. Future research

The Splawinski et al. (2014) regeneration model is parameterized to examine moderate to high severity fires, i.e., where 100% stand mortality occurs. Potential future research could focus on updating the model to simulate low and extreme severity fire. Low severity fires have been shown to negatively affect conifer recruitment due to less removal of the organic layer, thereby creating fewer suitable substrates for germination (Veilleux-Nolin and Payette, 2012); and also may have the potential to reduce the availability of seeds as fewer serotinous cones would be opened by the passage of the flaming front (Lecomte et al., 2006). Conversely, extreme fire severity within the canopy can limit the reproductive capacity of serotinous tree species (Pinno et al., 2013; Lecomte et al., 2006), by greatly reducing or eliminating the aerial seedbank (Pinno et al., 2013).

The approach taken thus far has focused on timber supply and natural regeneration, however as previously mentioned, many other species depend on burned landscape to fulfill part or all of their life cycles. Habitat suitability models (HSI) have been developed for select cavity-nesting birds in burned landscapes (Russell et al., 2007; Saab et al., 2015), and are based on identifying suitable avian habitat in burned forest, something that is highly species-dependent (Saab et al., 2009; Munroe, 2014). Currently, Saab et al. (2015) are developing a GIS-based decision support tool using HSI's aimed at guiding salvage logging operations in order to conserve habitat for species at risk. The combined use of models and GIS software presents a promising tool for planning and achieving management objectives in burned landscapes. Strategies can be developed integrating both economic and ecological objectives, thereby satisfying the needs of various stakeholders and making the practice of salvage logging more sustainable.

5. Conclusion

The proposed approach allows foresters and managers to not only determine reforestation needs for both *P. mariana* and *P. banksiana* following fire and salvage at the landscape level, but to also identify which management units are most susceptible to fire due to a lack of suitable natural regeneration following these disturbances. Costs of reforestation and the application of pre-commercial thinning could also be estimated across a large region, thereby aiding in determining future budget allocations and salvage sequence. In ecological region (6a), according to our model, only 35% of *P. mariana* and 6% of *P. banksiana* stands need to be planted following fire; however under the traditional 100% first winter salvage scenario, 100% of *P. mariana* and 74% of *P. banksiana* stands necessitate planting. If salvage logging is delayed until the second, third, or fourth winter following fire, planting will be required in 98%, 88%, and 66% of *P. mariana*, and 65%, 56%, and 56% of *P. banksiana* stands respectively. More widespread reforestation interventions may be required in the future due to an increase in younger age classes at the landscape level resulting from even-aged management practices and increases in fire frequency and area burned. Importantly, this project illustrates how the Splawinski et al. (2014) regeneration model can be linked with government and industry forest inventory maps to provide a unique spatial assessment tool. The stand-level data used in this study is typically available in other jurisdictions in one format or another, therefore although we only present results from Lake Matagami lowland ecological region (6a) in Quebec's boreal forest, with the help of our framework this tool can be created for other regions that possess the necessary data.

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