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Windthrow after group and dispersed tree retention in eastern Canada

Sébastien Lavoie [a], Jean-Claude Ruel [a,*], Yves Bergeron [b], Brian D. Harvey [c]

[a] Département des sciences du bois et de la forêt, Université Laval, 2405 Rue de la Terrasse, Sainte-Foy, Québec, Canada G1V 0A6
[b] Université du Québec en Abitibi-Témiscamingue, Chaire en Aménagement forestier durable, 445 boulevard de l'Université, Rouyn-Noranda, Québec, Canada J9X 5E4
[c] Département des sciences appliquées, Université du Québec en Abitibi-Témiscamingue, 445 boulevard de l'Université, Rouyn-Noranda, Québec, Canada J9X 5E4

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ABSTRACT

Although green tree retention has been proposed as a means of maintaining forest structural attributes normally found after stand replacing disturbances, there is some concern about mortality and fall rates of residual trees. Very little data regarding windthrow after green tree retention is currently available for Quebec, eastern Canada. The present study documents windthrow after two types of retention in two regions with contrasting biophysical characteristics, and identifies the most influential factors. A retrospective survey of windthrow was conducted in dispersed and group retention cuts conducted two to five years previously. In the Abitibi region of northwestern Quebec, the level of windthrow was comparable between group and dispersed tree retention. In the North Shore region of eastern Quebec, windthrow was higher in dispersed retention and levels of mortality were generally lower than in the Abitibi region. Windthrow probability was best explained by edaphic variables, tree species, slenderness ratio, tree height or diameter at breast height, and sapling abundance. Variables such as regional mean wind speed, retained group area, simple fetch and topographical exposure (Topex to distance) were not included in the best models. Although mortality rates increased after green tree retention, it remains to be established whether this increase has detrimental effects in terms of maintaining biodiversity.

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1. Introduction

Natural disturbance-based forest management has become a dominant paradigm of forest management, particularly in jurisdictions where large tracts of natural or semi-natural forest still exist. In much of the boreal forest, stand replacing disturbances, such as fire, act as the driving force shaping ecosystem structure and composition. Even though such disturbances introduce major changes to forest ecosystems, they leave structural legacies that provide crucial post-disturbance habitats important for biodiversity maintenance and regeneration processes (Rosenvald and Lõhmus, 2008). Variable retention has been proposed as means of maintaining similar structural legacies in managed forests (Beese et al., 2003; Franklin et al., 1997). This approach consists in maintaining live and dead trees, downed dead wood or other elements that are considered valuable (Beese et al., 2003; Mitchell and Beese, 2002). Living trees can be retained as dispersed trees or as variably-sized groups. Dispersed retention has the advantage of maintaining some structure over entire logged areas whereas group retention simplifies the access to the logged areas for additional treatments and may provide better habitat quality of some species.

Green tree retention can lead to higher levels of windthrow compared to untouched forests (Bebber et al., 2005; Bladon et al., 2008; Busby et al., 2006; Hautala and Vanha-Majamaa, 2006; Rosenvald et al., 2008; Scott and Mitchell, 2005). While gradual tree mortality can provide a desired input of dead wood over the medium term, high levels of mortality in the years immediately following cutting may impact species that depend on standing live trees and compromise the longer term recruitment of dead wood (Thorpe and Thomas, 2007).

Windthrow after green tree retention has been the focus of a limited number of studies in several forest regions. Hautala and Vanha-Majamaa (2006) reported windthrow losses varying between 15% and 48% of the number of stems for Norway spruce (*Picea abies* (L.) *Karst.*) after three years, depending on soil type. In coastal British Columbia, Scott and Mitchell (2005) observed 16% windthrow of residual stems between the first and sixth year after harvesting. Mortality in excess of 30% of the number of trees has been reported in Estonia and Oregon (Busby et al., 2006; Rosenvald et al., 2008). In Alberta, Bladon et al. (2008) observed windthrow mortality levels 2.5–4 times higher in green tree retention than in intact natural stands. For white pine (*Pinus strobus* L.) in central Ontario, Bebber et al. (2005) reported almost 25% of leave trees were windthrown after dispersed retention.

* Corresponding author. Tel.: +1 418 656 2131x7665; fax: +1 418 656 5262.
E-mail addresses: sebastien.lavoie.11@ulaval.ca (S. Lavoie), Jean-claude.ruel@sbf.ulaval.ca (J.-C. Ruel), yves.bergeron@uqat.ca (Y. Bergeron), brian.harvey@uqat.ca (B.D. Harvey).

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The use of green tree retention in eastern Canada is relatively recent and no information on windthrow after green tree retention in the eastern Canadian boreal forests is currently available. The type and level of retention could influence the amount of windthrow after green tree retention. Moreover, the boreal forest of eastern Canada is not homogenous and differences in soils, wind exposure or initial stand structure could lead to strong regional differences within this zone. The aims of this study are to document and quantify windthrow losses after group and dispersed tree retention in two regions of Quebec, Canada and to identify the main sources of variation. These regions differ markedly in natural disturbance dynamics, topography and soils. The earliest green tree retention cuts in Quebec are present in both regions.

1.1. Methods

The study was conducted in operational retention cuts of Québec, Canada. Regulations for group retention cuts in this province require that small groups amounting to 5% of the area are maintained (Leblanc, 2004). For dispersed retention, 25 merchantable stems per hectare have to remain after harvesting (Leblanc, 2005).

1.2. Study area

The first region (Abitibi) is located in northwestern Québec (Fig. 1). In this region, the fire cycle, previously around 150 years, is now at 325 years (Bergeron et al., 2001). A first sector was sampled in this region, north of La Sarre (Tembec: 49°41'N to 49°37'N; 78°62'W to 78°52'W), in the western part of the black spruce–feathermoss bioclimatic domain (black spruce: *Picea mariana* (Mill.) B.S.P.) (Robitaille and Saucier, 1998). Group and dispersed tree retention harvests were applied in winter 2007–2008. A second sector was also sampled in this region. This sector is located in the Lake Duparquet Research and Teaching Forest (FERLD: 48°47'N to 48°44'N; −79°44'W to −79°40'W), in the western section of the balsam fir-white birch bioclimatic domain (balsam fir: *Abies balsamea* (L.) Mill; white birch: *Betula papyrifera* Marsh.) (Robitaille and Saucier, 1998). Group retention in this sector was applied in winter 2001–2002 whereas dispersed retention was applied in winters 2004–2005 and 2005–2006. Pre-harvest stands were mostly aspen (*Populus tremuloides* Michx.) dominated, with black spruce and jack pine (*Pinus banksiana* Lamb.) as companion species. Only 5% of the stands were pure softwoods. Cutover size varied between 8 and 94 ha. Average retained group size was 495 m^2 and an average of 60 stems/ha was remained in dispersed retention cuts.

In both sectors, topography is relatively gentle (Simard et al., 2008). Surface soil is derived mainly from glacio-lacustrine clays (85% of the sample) or glacial till (FERLD only, 7% of the sample) (Simard et al., 2008). Paludification can lead to the formation of deep organic soils (8% of the sample in the more northerly sector where Tembec cuts occurred) (Simard et al., 2007). Mesic sites dominate (74% of the sample), followed by hydric sites (18%) and xeric sites (8%). Mean annual temperature is 0.75 °C and mean annual precipitation is about 900 mm (Environnement Canada, 2009). The harvested sectors were even-aged and originated from fires that occurred around 1920 (Bergeron et al., 2004).

The second region (North Shore) is located in the Laurentian Hills, in the eastern section of the black spruce-feathermoss bioclimatic domain. Fire cycles are relatively long, from 270 to over 500 years (Bouchard et al., 2008). The topography of this region is more complex, with higher elevation and deep valleys (De Grandpré et al., 2008). Rock outcrops are very frequent and till is the main surface deposit. Deep tills (>1 m) comprise 57% of the sample whereas shallow tills and organic soils form 41% and 2% of the sample, respectively. Mesic sites dominate (61% of the sample), followed by hydric sites (35%) and xeric sites (4%). Mean annual temperatures range from −1.0 to 2.5 °C and annual precipitation ranges from 900 to 1300 mm (Environnement Canada, 2009). The mainly irregular structure of stands (Boucher et al., 2003) is a consequence of long fire cycles and predominance of older forests. Spruce budworm (*Choristoneura fumiferana* Clem.) outbreaks, windthrow and gap dynamics are the primary drivers of stand dynamics (Bouchard et al., 2008; De Grandpré et al., 2000). Three sectors were sampled in this region. In the first sector (Arbec: 51°43'N to 51°25'N; 68°12'W to 68°29'W) retention cuts were done in winter 2006–2007. In the second sector (Abitibi-Bowater: 50°25'N to 50°12'N; 68°83'W to 68°73'W), cuts were applied in winter 2007–2008. Finally cuts in the third sector (Boisaco: 50°24'N to 49°72'N; 69°99'W to 69°74'W) took place in winters 2005–2006 and 2006–2007. Pre-harvest stands were all softwood dominated. Seventy-one percent of them were dominated by black spruce. Cutover size varied between 13 and 102 ha. Average retained group size was 367 m^2 and an average of 93 stems/ha was remained in dispersed retention cuts.

In the Abitibi region, a total of 18 and 10 retention groups were sampled in the Tembec and the FERLD sectors, respectively. Two cutovers of dispersed retention were studied for each sector in Abitibi. For group retention in the North Shore region, 18, 30, and 24 groups

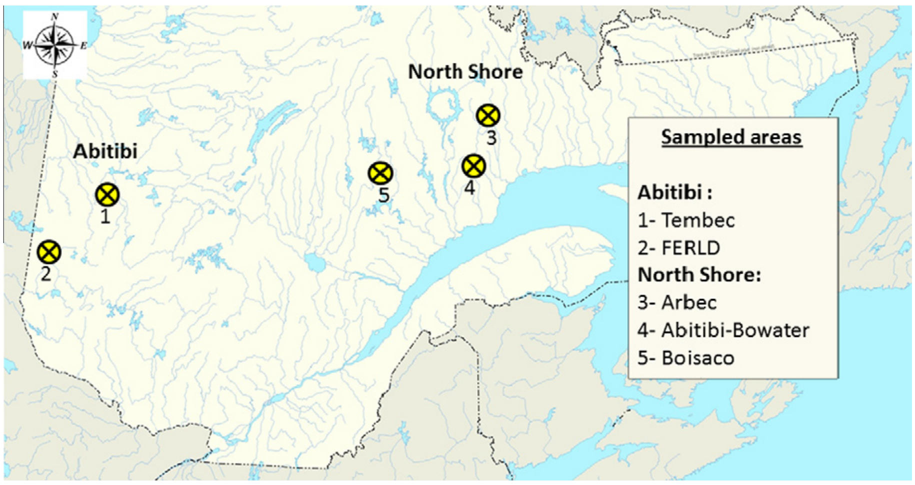

Fig. 1. Location of the study sites.

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were sampled for the Arbec, Abitibi-Bowater and Boisaco sectors, respectively. For dispersed retention in this region, three sectors were sampled in the Arbec sector and four in the Boisaco sector. No dispersed retention was sampled in the Abitibi-Bowater sector. The total area sampled amounts to 6.1 and 17.5 ha, for the Abitibi and North Shore regions, respectively

1.3. Sampling

Field sampling took place in summer 2009. Selected logged areas had to be at least two years old so that the time-effect of cutting on windthrow could be detected. Maximum cut age was five years in order to distinguish post-cut windthrow from previous mortality. This time interval corresponds with our focus on short-term mortality. Dead trees that still bore needles or whose bark was firmly attached to the trunk were considered as post-harvest mortality (Bladon et al., 2008; Fleming and Crossfield, 1983). For tree groups left in 2001–2002 at the FERLD, monitoring conducted by research forest personnel in 2004 was used in order to respect our five year post-harvest time limit.

Sampling methods differed with the type of tree retention. For dispersed retention, the edge of the logged area closest to the access road was divided into 10 sections of equal length. In each of these, a sampling starting point was randomly chosen. From each of these points, a transect was traced to the opposite edge of the cut. Transects were parallel to each other and each was subdivided into 1000 m^2 plots (10×100 m). For group retention, groups of trees were randomly chosen over the whole area using aerial photographs. The longest and the smallest axes of each group of trees were measured to determine group area.

In each group or plot, living and dead merchantable stems were numbered by species. Uprooted, broken, as well as severely leaning living trees (angle greater than 30° relative to the vertical) were classified as windthrown. Diameter at breast height (dbh) was measured on every tree. Height of every tree (standing and windthrown) was measured for dispersed retention only. From height and dbh measurements, the slenderness ratio (height/dbh) was calculated for dispersed retention.

Total area of each cutover as well as stand type, age, height and stand density were derived from forest maps. For the North Shore region, the basal area of saplings was measured in a 40 m^2 circle located at the center of the group. Saplings were not measured in the Abitibi region because they were absent or rare in most cases. Soil information was derived from soils pit dug at the center of each group or plot. Slope was also measured for each sampled plot or group. Mean annual windspeed at 30 m height was obtained from the Canadian wind atlas (http://www.atlaseolien.ca/fr/maps.php), based on a 5 km by 5 km grid.

Since topography can play a major role in explaining windthrow (Ruel, 1995; Ruel et al., 2002), the topex-to-distance (thereafter referred to as topex) was selected, based on a close relationship with local wind speed (Ruel et al., 1997). This index is calculated as the sum of horizontal angles to the main topographic feature within a 500 m distance, for each of the eight main cardinal directions (Miller et al., 1987). Topex was extracted from a database of the Quebec Ministry of Natural Resources and Wildlife, based on a routine developed by Ruel et al. (2002). The effect of cutover size and shape was quantified using the simple fetch of Scott and Mitchell (2005). This index is calculated as the sum of distance to stand edge (up to 300 m) for the eight main cardinal directions. This index was generated using Arc Map™ 9.3 from ESRI®.

To establish a reference for natural mortality in uncut stands, 400 m^2 plots were sampled in natural stands of similar composition, close to the logged areas. The same variables as for retention areas were measured. Four of these reference plots were sampled in Abitibi and 11 in the North Shore region.

1.4. Statistical analysis

The probability of windthrow was modeled at the tree level with logistic regression (Aitchison and Silvey, 1957). Regressions were calculated with the lme4 package of R software (R Development Core Team, 2009). Equations use the logistic link function and probabilities are then calculated (Jongman et al., 1995):

$$\text{logit } p = \ln \frac{p}{1-p} = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \cdots, \quad (1)$$

$$p = \frac{\exp(\beta_0 + \beta_1 X_1 + \beta_2 X_2 + \cdots)}{1 + \exp(\beta_0 + \beta_1 X_1 + \beta_2 X_2 + \cdots)}, \quad (2)$$

where β_0 is the intercept, β_i are parameters to be estimated, and X_i are the sampled variables. Parameters were estimated with the maximum likelihood method (Aitchison and Silvey, 1957).

Windthrow modeling was done using mixed models. Group or plot, harvest area, and sector were considered as random effects. Model comparison was made using corrected Akaike information criterion (AIC_c) (Burnham and Anderson, 2004).

To use AIC_c, *a priori* models were established based on known information of factors influencing windthrow. AIC_c ranks models, considering the number of parameters included (Mazerolle, 2006). However, it does not guarantee that the best model has been found.

Retention types have been treated separately since sampling techniques and measured variables differed. Both regions also had to be treated separately because too many differences existed in stand structure, topography, species composition and soil type. These differences caused a failure of the maximum likelihood function to converge when both regions were modeled together. This situation can occur when the response level always have the same value for a given combination of predictor variables. Such a case is more frequent with dummy variables such as those describing stand attributes (Altman et al., 2004).

Astrup et al. (2008) contend that parsimony is critical in developing forest models. Those with a limited number of predictors are preferable (Valinger and Fridman, 1997) and, ideally, a limited number of models should be compared (Anderson and Burnham, 2002). Hence, around ten models were compared for each region and retention type. A first group of models was built from individual tree characteristics. A second group was created using soil variables and a third used variables related to wind exposure. Another group was built from pre-harvest stand characteristics. The creation of a full model using stand characteristics was not possible because of a convergence problem of the maximum likelihood function. Consequently, a series of models using individual variables was tested. More complex models using tree characteristics, such as species, dbh, height, slenderness ratio, and soil characteristics were also built. This approach attempts to consider variables included in mechanistic models, such as ForestGales (Gardiner and Quine, 2000). In addition, for the North Shore region, basal area of saplings was added since the presence of an understory can reduce the wind load applied on larger stems (Gardiner et al., 2005).

Models considered only major species present. For group retention in the Abitibi region, spruces, jack pine and trembling aspen were included whereas for dispersed retention spruces, trembling aspen and willows (*Salix* spp) were included. Black spruce and white spruce (*Picea glauca* (Moench) Voss) were grouped together on the basis of closely related resistance to uprooting for the same tree size (Achim et al., 2005; Élie and Ruel, 2005) and to ensure an adequate sample size. For the North Shore region, only black spruce and balsam fir were included for both group and dispersed retention.

Prior to building models autocorrelation between variables was checked for each condition to be modeled. Dbh and height were considered to be autocorrelated since the associated R^2 values

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were greater than 0.5. For this reason, they were not used together in any models. Simple interactions were tested and included when significant. The parameter for overdispersion ($\hat{c}$) was also checked (Mazerolle, 2006) but no problem was detected.

2. Results

Stands in Abitibi had even-aged structures since they originated from ca 1920 fires. For group retention, a total of 1423 stems were sampled. Spruces were by far the main species sampled ($n = 802$), followed by jack pine ($n = 414$) and trembling aspen ($n = 152$). Windthrow accounted for 41.1% of the stems and spruce was most affected (for details, see Appendix). For dispersed retention in Abitibi, fewer trees were sampled ($n = 346$). Of these, 126 (36%) were windthrown. Spruces ($n = 109$), trembling aspen ($n = 102$) and willows ($n = 64$) were the main species retained and suffered losses of 41.3, 13.7 and 42.2%, respectively. In addition, although their sample sizes were small, jack pine and white birch were seriously affected by windthrow (50.0% and 40.9%, respectively).

In the North Shore region, stands had an irregular (uneven-aged) structure. They were classified as >120 years, old uneven-aged, or young uneven-aged on forest maps. A total of 2134 trees (1782 black spruce, 329 balsam fir) were measured in group retention. In dispersed retention, 1745 trees (593 spruce, 1106 fir) were sampled. Windthrow losses for the North Shore region were lower than in the Abitibi region. Windthrow levels were 17.3% and 28.0% for group and dispersed retention, respectively. Of the two main species sampled, balsam fir had higher rates of windthrow: 27.1% and 30.6% for group and dispersed retention, respectively.

Table 1
Comparison of models for group retention in Abitibi.

Models			Log likelihood	K	AIC_c	Δ_i	w_{ic}
Tree characteristics	1	Null	−869.4	4	1746.9	452.5	0
	2	Dbh + species + dbh: species	−645.2	9	1308.5	14.1	0.01
Soil characteristics	3	Drainage + deposit + humus + slope	−845.4	10	1710.9	416.4	0
Wind exposure characteristics	4	Topex + log (simple fetch) + mean annual windspeed + group area	−867.9	8	1744.4	449.5	0
Stand characteristics	5	Harvest year	−869.4	5	1748.9	453.9	0
	6	Sapling basal area + trees basal area	−868.0	7	1748.2	453.9	0
	7	Stand age	−869.4	5	1748.9	453.9	0
	8	Stand density	−869.4	6	1750.9	455.9	0
	9	Stand height	−869.1	5	1748.3	453.3	0
	10	Stand type	−853.5	10	1727.3	432.3	0
Tree and soil characteristics	**11**	**Dbh + species + dbh: species + deposit + drainage**	**−634.1**	**13**	**1294.5**	**0**	**0.99**

Table 2
Comparison of models for dispersed retention in Abitibi.

Models			Log likelihood	K	AIC_c	Δ_i	w_{ic}
Tree characteristics	1	Null	−151.3	4	310.9	41.1	0
	2	Height + species + h/d + height: species + h/d: species	−128.0	12	281.4	11.6	0.003
	3	Dbh + species + h/d + h/d: species	−133.6	10	288.1	18.3	1e−04
Soil characteristics	4	Drainage + deposit + humus + slope	−150.2	8	317	47.2	0
Wind exposure characteristics	5	Topex + log (simple fetch) + mean annual windspeed	−144.9	7	304.2	34.5	0
Stand characteristics	6	Harvest year	−149.8	6	312.9	42.3	0
	7	Trees basal area	−151.3	5	312.9	43.2	0
	8	Stand age	−151.2	5	312.7	42.9	0
	9	Stand density	−148.9	7	312.3	42.6	0
	10	Stand height	−149.4	6	311.1	41.4	0
	11	Stand type	−148.9	9	316.6	46.8	0
Tree and soil characteristics	**12**	**Height + species + h/d + height: species + h/d: species + drainage + deposit**	**−119.9**	**14**	**269.7**	**0**	**0.98**
	13	Dbh + species + h/d + dbh: species + h/d: species + drainage + deposit	−124.1	14	278.1	8.3	0.01

Table 3
Comparison of models for group retention in North-Shore.

Models			Log likelihood	K	AIC_c	Δ_i	w_{ic}
Tree characteristics	1	Null	−859.6	4	1727.3	127.0	0
	2	Dbh + species + dbh: species	−834.4	7	1682.9	82.7	0
Soil characteristics	3	Drainage + deposit + humus + slope	−814.6	11	1651.4	51.2	0
Wind exposure characteristics	4	Topex + log (simple fetch) + mean annual windspeed + group area	−851.3	8	1718.8	118.5	0
Stand characteristics	5	Harvest year	−856.2	7	1726.4	126.2	0
	6	Sapling basal area + tree basal area + pct fir	−848.4	8	1712.8	112.6	0
	7	Stand age	−850.6	8	1717.2	117.0	0
	8	Stand density	−859.6	6	1731.3	131.0	0
	9	Stand height	−852.9	5	1715.8	115.6	0
	10	Stand type	−815.0	8	1645.9	45.7	0
Tree and soil characteristics	**11**	**Dbh + species + dbh: species + deposit + drainage + sapling basal area**	**−788.0**	**12**	**1600.2**	**0**	**1**

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Table 4
Comparison of models for dispersed retention in North-Shore.

Models			Log likelihood	K	AIC_c	Δ_i	w_{ic}
Tree characteristics	1	Null	−933.4	4	1874.8	147.9	0
	2	Height + species + h/d + height: h/d + height: species	−856.6	9	1731.3	4.3	0.10
	3	Dbh + species + h/d	−866.8	7	1747.7	20.8	3e−05
Soil characteristics	4	Drainage + deposit + humus + slope + drainage: humus	−921.6	9	1861.2	134.2	0
Wind exposition characteristics	5	Topex + log (simple fetch) + mean annual windspeed	−929.5	7	1873.1	146.1	0
Stand characteristics	6	Harvest year	−929.5	6	1871.1	144.1	0
	7	Sapling basal area + tree basal area + pct fir	−931.8	7	1877.7	150.7	0
	8	Stand age	−892.8	6	1797.6	70.6	0
	9	Stand density	−930.0	6	1872.2	145.3	0
	10	Stand height	−930.6	5	1871.2	144.2	0
	11	Stand name	−931.4	7	1876.8	149.8	0
Tree and soil characteristics	**12**	**Height + height + h/d + height: h/d + height: species + deposit + drainage + sapling basal area**	**−851.4**	**12**	**1727.0**	**0**	**0.89**
	13	Dbh + species + h/d + deposit + drainage + sapling basal area	−910.7	9	1839.5	112.5	0

Table 5
Best fit models parameters for each case of retention, Abitibi region.

Group retention		Dispersed retention	
Variable	Parameter (standard error)	Variable	Parameter (standard error)
Intercept	−0.95 (0.91)	Intercept	−1.13 (1.32)
Dbh (cm)	−0.02 (0.02)	Height (m)	0.15 (0.07)
Populus tremuloides	0.13 (1.25)	*Populus tremuloides*	0.46 (3.24)
Pinus banksiana	1.13 (0.70)	*Salix* sp.	4.14 (2.26)
Mesic sites	3.19 (0.92)	Height/dbh	−0.78 (0.94)
Xeric sites	3.52 (1.43)	Tills	3.24 (0.96)
Tills	0.06 (1.17)	Xeric sites	−0.86 (0.51)
Dbh: *Populus tremuloides*	−0.16 (0.05)	Height: *Populus tremuloides*	−0.37 (0.12)
Dbh: *Pinus banksiana*	−0.24 (0.04)	Height: *Salix* sp.	−0.59 (0.21)
		Height/dbh: *Populus tremuloides*	4.69 (2.34)
		Height/dbh: *Salix* sp.	3.79 (1.75)

Topography in Abitibi is relatively flat and topex values ranged from −2.59° to 4.89 whereas the more pronounced topography of the North Shore region was reflected by topex values ranging between −52.96° and 39.00°. Patch size of group retention was also slightly different between regions; mean area was 495 m^2 in Abitibi compared to 367 m^2 in the North Shore region.

The impact of retention type was not consistent between regions. In the Abitibi region, windthrow was higher in groups whereas the opposite was found in the North Shore region. Windthrow in reference plots in intact nearby stands was much lower than in any retention treatment, reaching only 4% in the Abitibi region and 2% in the North Shore region.

Tables 1–4 present the ranking of logistic regression models, based on AIC_c. The best model for group retention in Abitibi was model 11 (Table 1; AIC_c = 1294.5; w_{ic} = 0.99) whereas model 12 was the best for dispersed retention in the same region (Table 2; AIC_c = 269.7; w_{ic} = 0.98). The best model for group retention in the North Shore region was model 11 (Table 3; AIC_c = 1600.2; w_{ic} = 1). Finally, for dispersed retention in that region, model 12 clearly outperformed the others (Table 4; AIC_c = 1727; w_{ic} = 0.89). The best models in each case included both variables related to trees and soils. No other model was strong enough to be considered.

The probability of windthrow in group retention in the Abitibi region was a function of species, diameter, soil deposit and drainage (Table 5). The effect of diameter varied with species (Fig. 2). A clear decrease in the probability of windthrow with diameter could be seen for trembling aspen and jack pine where the probability of

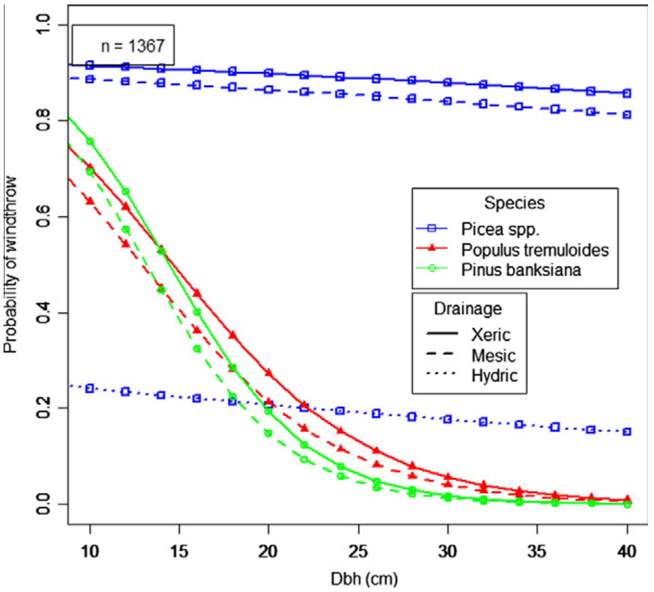

Fig. 2. Probability of windthrow for Abitibi group retention in clay deposits.

windthrow was very low for trees with dbh larger than 30 cm. Spruces were more vulnerable than jack pine or trembling aspen

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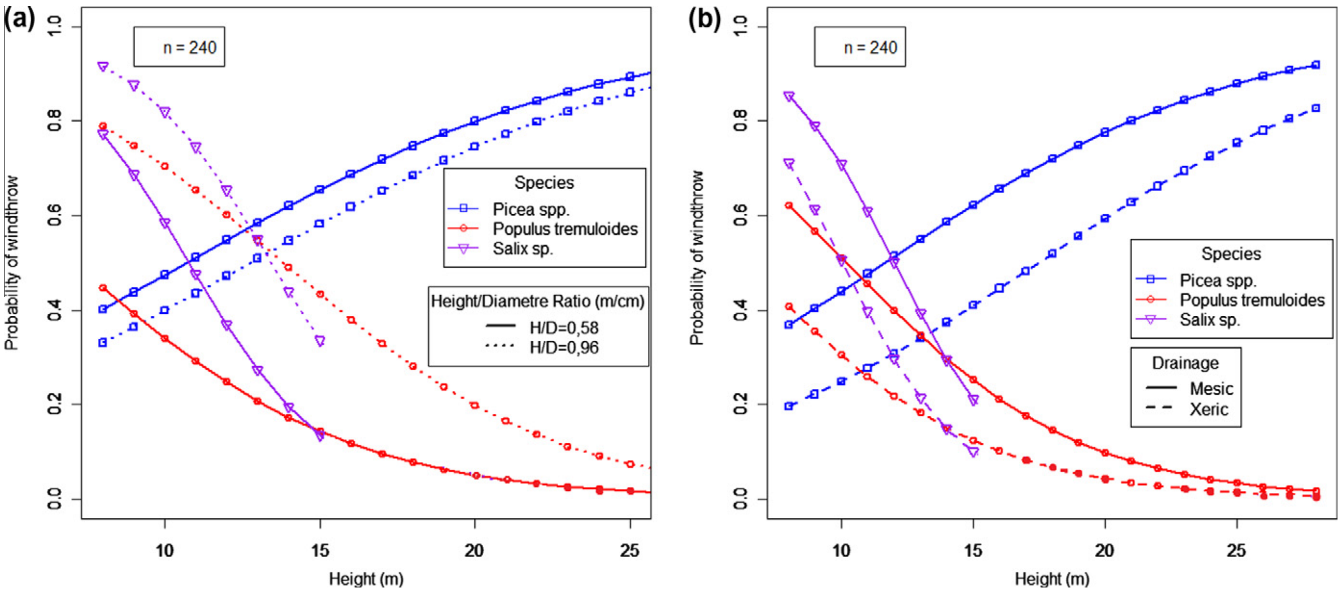

Fig. 3. Probability of windthrow for Abitibi dispersed retention (a) on a mesic clay deposit (b) on clay deposits with a h/d ratio of 0.76 m/cm.

Table 6
Best fit models parameters for each case of retention, North Shore region.

Group retention		Dispersed retention	
Variable	Parameter (standard error)	Variable	Parameter (standard error)
Intercept	−2.60 (0.85)	Intercept	−6.31 (1.26)
Dbh (cm)	0.07 (0.02)	Height (m)	0.50 (0.12)
Abies balsamea	−0.48 (0.57)	*Abies balsamea*	0.42 (0.56)
Mesic sites	−0.03 (0.75)	Height/dbh	5.70 (1.46)
Xeric sites	0.08 (0.78)	Shallow tills	−0.05 (0.23)
Shallow tills	−0.12 (0.32)	Xeric sites	−0.51 (0.19)
Organic soils	−0.19 (0.97)	Sapling basal area (m^2/ha)	−26.25 (12.19)
Sapling basal area (m^2/ha)	−48.52 (15.53)	Height: *Abies balsamea*	0.02 (0.05)
Dbh: *Abies balsamea*	0.08 (0.04)	Height: height/dbh	−0.50 (0.13)

on xeric and mesic sites. The probability of windthrow on mesic and xeric sites was quite similar for all species. However, the probability of windthrow for black spruce on hydric sites was much lower than on mesic or xeric sites. Surface deposit also appeared in the selected model but there was no difference between tills and clays with respect to the risk of windthrow.

For dispersed retention in Abitibi, the probability of windthrow was influenced by height, species, slenderness ratio, drainage and deposit (Table 5). The effect of height varied with species (Fig. 3). For willows and trembling aspen, windthrow risk decreased with height, in contrast with spruces where risk increased with height. Beyond 14 m height, spruces became the most vulnerable. The effect of slenderness ratio was also not consistent between species. For willows and aspen, the probability of windthrow increased with the slenderness ratio whereas it decreased for spruces. Moreover, in general, trees on mesic sites appeared to be more vulnerable than those on xeric sites (Fig. 3).

For group retention in the North Shore region, the probability of windthrow varied with dbh, species, surface deposit, drainage and sapling basal area (Table 6). The effect of dbh differed between species (Fig. 4). This increase was steeper for balsam fir, especially beyond 14 cm dbh. Overall, balsam fir was more vulnerable than black spruce. Surface deposit and drainage, although included in the model, had negligible influences on windthrow risk. Sapling basal area, (not measured in Abitibi), had a significant effect on windthrow risk; in effect, a high abundance of saplings had a beneficial influence on stand stability.

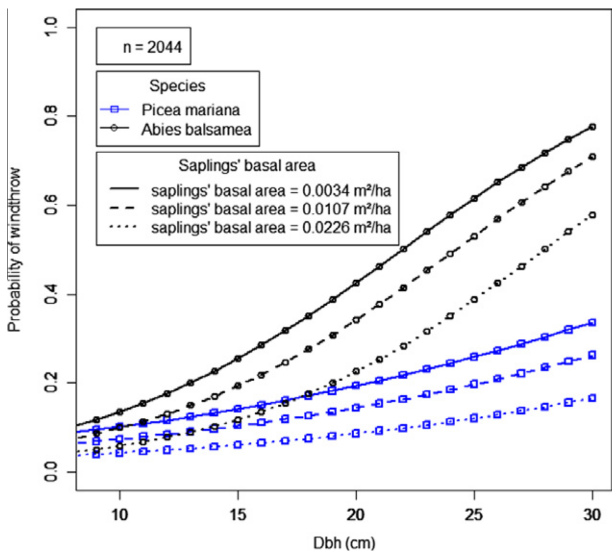

Fig. 4. Probability of windthrow for North-Shore group retention in mesic shallow till.

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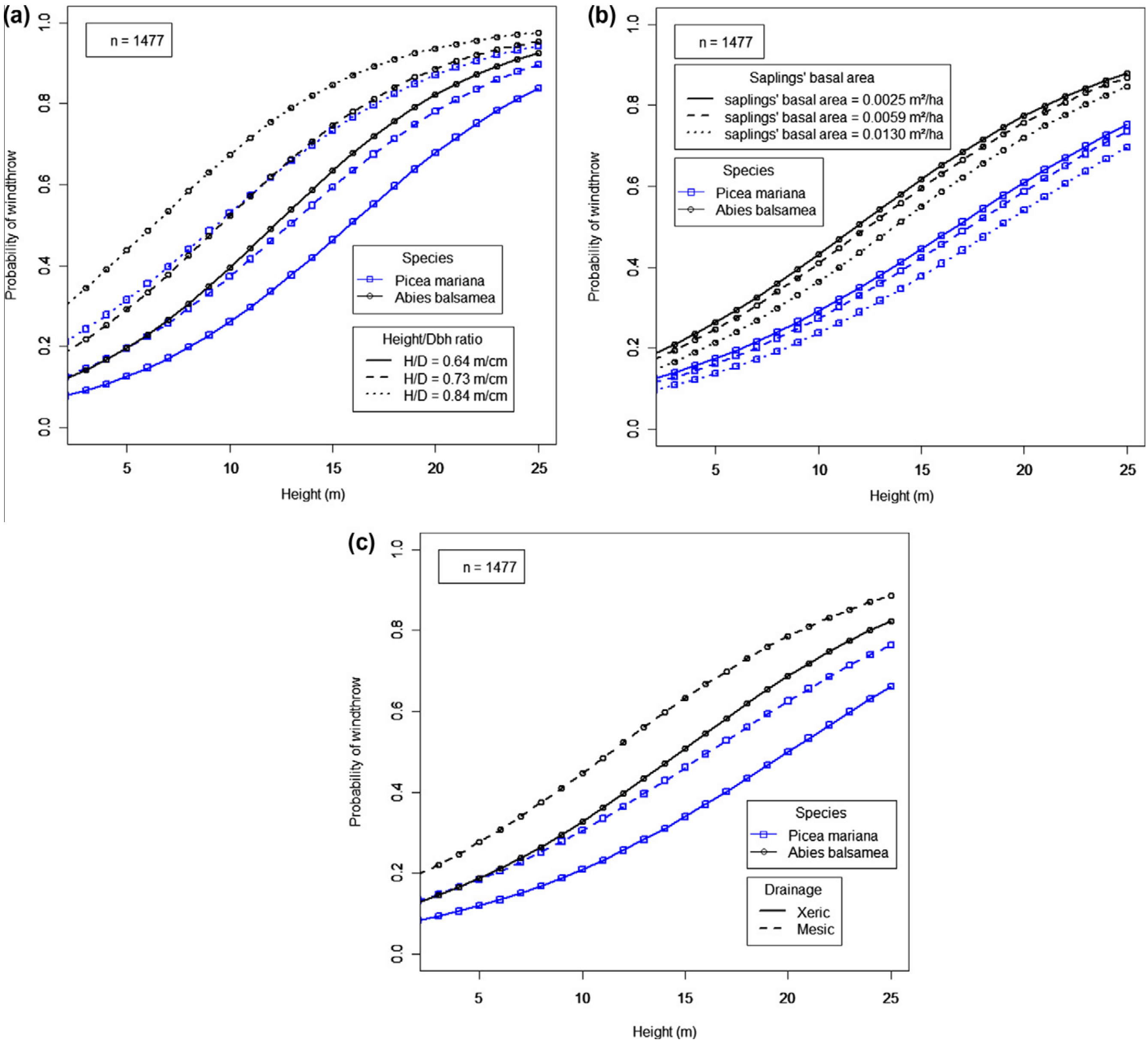

Fig. 5. Probability of windthrow for North-Shore group retention (a) in deep till with a sapling basal area of 0.0059 m^2/ha and a h/d ration of 0.73 m/cm (b) in mesic deep till with a h/d ratio of 0.73 m/cm (c) in a mesic deep till with a sapling basal area of 0.0059 m^2/ha.

The probability of windthrow in dispersed retention of the North Shore region was influenced by height, species, slenderness ratio, surface deposit, drainage and basal area of saplings (Table 6). There was a general increase in windthrow probability with height, although the effect differed slightly between balsam fir and black spruce (Fig. 5). As was the case for group retention, balsam fir was more vulnerable than black spruce for similar heights. The risk of windthrow increased with height and slenderness ratio (Fig. 5). Although the presence of saplings was not as important as for group retention, saplings did have a beneficial effect on stability in dispersed retention (Fig. 5). Trees on mesic sites were more vulnerable than those on xeric sites.

3. Discussion

Originally developed in the Pacific Northwest in the nineties (Franklin et al., 1997), variable retention harvesting has only recently been introduced in Quebec. This explains why, to our knowledge, no published information on windthrow in retention cuts is available for this part of the boreal forest. Extrapolation from other regions is challenging, since driving variables, such as wind climate, soils, and species (Ruel, 1995), can vary a lot. Even within a given region, species relative resistance to uprooting can vary with soil type (Élie and Ruel, 2005). In heavy partial harvesting experiments that removed around 85–90% of basal area in the eastern Canadian boreal forest and largely concentrated in the North Shore region, windthrow attained 22% of basal area, 10 years after harvesting (Riopel et al., 2010). This level of mortality falls about mid-way between those associated with dispersed and group retention in the North Shore region, although the time elapsed since cutting is much shorter in our study.

For all regions and retention types, the level of windthrow was higher than that observed in uncut plots. This is consistent with all other studies of windthrow following variable retention harvesting (Bebber et al., 2005; Hautala and Vanha-Majamaa, 2006; Rosenvald et al., 2008; Steventon, 2011). It thus seems clear that retention cuts increase the risk of windthrow compared to stands situated within an intact forest matrix. Since green tree retention is intended to

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emulate residual forest structures left after stand-replacing disturbances, a more appropriate comparison could probably be made with windthrow levels occurring after such disturbances. However, such information has rarely been collected in similar ecosystems. Angers et al. (2011) monitored windthrow after fire in the Abitibi region. Our level of mortality was higher than that observed in their study, even though it was likely increased by salvage harvesting that was conducted close to their sample plots. Whether this difference is meaningful or not could differ with the species to be maintained, the amount of residual canopy or the general characteristics of the landscape. According to Rosenvald and Lõhmus (2008), guidelines for green tree retention should take post-cut mortality into consideration either by increasing retention levels to compensate for anticipated mortality or selecting topographic and stand conditions that provide highest probability of low windthrow levels.

In the Abitibi region, windthrow risk varied for different species. Spruces appeared to be more vulnerable than jack pine, trembling aspen and willows. Hardwoods in general are considered to be more resistant to windthrow than conifers, especially after leaf fall (Savill, 1983) and aspen has a widespread root system with sinker roots that provide good anchorage (Perala, 1990). Jack pine generally develops tap roots that improve anchorage (Béland et al., 1999; Rudolph and Laidly, 1990) and has been shown to be more resistant to uprooting than black spruce on soils offering a good rooting potential (Élie and Ruel, 2005).

Results from the North Shore region confirm previous findings (Pham et al., 2004; Ruel, 2000) that balsam fir is more vulnerable to windthrow than black spruce. Both species have superficial rooting systems (Frank, 1990), and similar resistance to uprooting for similar stem sizes in the absence of decay (Achim et al., 2005; Élie and Ruel, 2005). Greater windthrow vulnerability of fir is probably linked to the species' higher incidence of decay. Indeed, balsam fir has been reported as the eastern Canadian species most susceptible to butt and root rots, particularly after age 50 years (Whitney, 1989).

Tree size (diameter or height) has been a significant variable in all models retained. For the North Shore region, larger trees were more vulnerable to windthrow. This concurs with previous results from partial or retention cutting experiments (Rich et al., 2007; Riopel et al., 2010; Schmidt et al., 2010; Steventon, 2011; Thorpe et al., 2008). Diameter can often be seen as a surrogate for height and higher trees are exposed to stronger winds and offer a longer leverage to the wind action (Canham et al., 2001; Ruel, 1995). For dispersed retention in this region, the effect of the slenderness ratio was also consistent with previous studies related to partial cutting where higher ratios were associated with higher vulnerability (Cremer et al., 1982; Gardiner et al., 1997; Jull, 2001; Schmidt et al., 2010).

Some of our results for group retention in Abitibi are contrary to the existing literature, notably that probability of windthrow decreased with increasing diameter. Rosenvald et al. (2008) found an increase in mortality of *Populus tremula* after retention cuts and stated that the effect of dbh on mortality could be species dependent. However, in our case, the effect of dbh was similar for all species in dispersed retention but not always consistent between retention types within the same region. In these even-aged stands, small diameter trees were likely poor vigor, suppressed or intermediate trees. In addition, the relationship between height and diameter may have been weaker since height should be relatively constant. Thus an increase in diameter could have meant a decrease of the slenderness ratio for trees within the same stand. Since height was not measured for every tree, height and slenderness ratio could not be tested. In addition, trees on hydric soils were smaller (DBH: 11.99(±2.65) cm) than on xeric or mesic soils (DBH: 14.13(±3.57) cm). This could have also influenced the fact that spruce in group retention of Abitibi appeared more resistant on hydric soils which tends to contradict the literature.

The decrease in vulnerability with height for trembling aspen and willows in dispersed retention of Abitibi is also in contradiction with the literature and difficult to explain but could be linked to the small sample size used for building the models (n = 240). If this behavior was confirmed by additional data, it could be worth retaining large trembling aspen trees, as suggested by Rosenvald et al. (2008) for *Populus tremula*.

Results from the Abitibi region did not demonstrate a noticeable effect of surface deposit. However, this is not surprising, given the similar soil texture of Cochrane tills and glacio-lacustrine clays.

In the North Shore region, stands with more saplings were less vulnerable, a finding that concurs with observations by Riopel et al. (2010). In a wind tunnel study, Gardiner et al. (2005) have shown that small stems can reduce the wind load applied on dominant trees and saplings could likely play a similar role. Small trees could also help to dissipate part of the wind energy by interactions during swaying (Gardiner et al., 2005). An abundance of saplings reflects a previously well-developed stand structure associated with gap dynamics (McCarthy, 2001). In addition to the direct effect saplings can have on wind speed, trees in irregular stands have grown in more open conditions, conducive to the development of lower slenderness ratios and better stability (Mason, 2002; Ruel, 1995).

In the North Shore region, trees in dispersed retention on mesic soils appeared to be more vulnerable to windthrow than those on xeric soils. Although several studies have compared windthrow on wet and mesic soils (Basham, 1991; Whitney, 1989; Whitney et al., 2002), very little information is available for dryer soils but it may be assumed that xeric growing conditions would favor deeper rooting on these soils.

Models that integrate variables describing wind exposure (topex, mean wind speed and simple fetch) were not selected by the model comparison procedure even though these are often identified as important variables for explaining windthrow (Ruel et al., 2002; Scott and Mitchell, 2005). This can probably be explained by the small range of values included in this study. The topography in the Abitibi region is rather flat, which was reflected in the values of the topex that remained close to zero. In the North Shore region, the range of topex values was much wider but probably did not reflect the complete variation of topography within the region. More likely, exposure extremes were probably avoided when treatments were applied since Ruel et al. (2002) observed a much wider range in another section of the Laurentian Hills. Mean wind speed varied between 3.9 and 4.3 m/s in the Abitibi region and between 4.0 and 6.2 in the North Shore region. Because the wind atlas provides wind data on a 5 by 5 km grid, important variations in wind speed can take place between grid points (Ruel et al., 1998). However, topex to distance should capture these variations. In addition to the relatively small variations in mean windspeed, this variable may not capture differences in strong winds. Simple fetch varied between 816 and 2124 m in the Abitibi region and between 1069 and 2400 m in the North Shore region. These ranges are much narrower than that studied by Larouche (2005) (52–3490 m) who found a significant effect of simple fetch on windthrow at clearcut edges. Patch size was also not included in selected models, which is contrary to results reported by Steventon (2011). However, in his study, retention patches reached much larger sizes (up to 6 ha) and a patch area of at least 1 ha (>20 times greater than mean retention patch size in this study) was suggested as a way to minimize windthrow.

Although statistical comparisons could not be made between the two types of retention, it is interesting to note the amount of windthrow associated with both treatments. For the Abitibi region, differences were minor and losses were high after both retention treatments. Small differences in species composition could probably explain such differences. For the North Shore, mortality was

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about 10% higher for dispersed retention. Maguire et al. (2007) also found higher levels of mortality in dispersed retention. Dispersed retention provides little protection against wind for isolated stems whereas, within retained groups, only edge trees are fully exposed to wind. Differences in species composition could also play a role since balsam fir was more abundant in dispersed retention.

Although results could not be compared statistically between the two regions, it is interesting to note that losses were generally higher in the Abitibi region. Many variables, such as topography, dimension of retained groups, soil types and stand structure, differed between the two regions. Since topex and dimension of retained groups were not retained in the best models, it is likely that they did not play a major role in explaining regional differences. Major soil differences could however play a role. Soils in Abitibi were mostly glacio-lacustrine clays whereas tills were dominant in the North Shore region. However, we did not observe a greater vulnerability on clay soils relative to tills in the Abitibi region. Clay soils are often wet soils that do not favor rooting (Busby, 1965; Moore, 1977; Ruel, 1995; Schaetzl et al., 1989) but our clay soils were generally considered as mesic. In addition, we could not demonstrate a higher susceptibility on wet soils in our study. Since black spruce was found in group retention on tills of both regions, it is interesting to look at model estimates for trees of similar size. For a tree of 15 cm DBH on a mesic till, the probability of windthrow would reach 88% in Abitibi, in comparison with 18% in the North Shore, suggesting that other factors were playing a major role. However, tills in the two regions differed in terms of soil texture (clays for one of the sectors in the Abitibi region and sandy loams for the North Shore region) so that the comparison is not perfect.

The difference in fire cycles between the two regions had a significant effect on the stand structures sampled. In the Abitibi region, stands originated from fires that occurred around 1920. They were thus characterized, in part, by an even-aged structure and mature mixes of shade-intolerant species with little understory. The combination of stand structure and the fact that breakup was beginning in these stands at the time of harvesting would make residual trees particularly vulnerable to windthrow. In the North Shore region, stands were generally over 120 year old and presented an irregular structure with gaps and an abundant sapling layer. Since the abundance of saplings was seen to play an important role in residual tree stability, differences in stand structure between the two regions appear to explain part of the differences in windthrow.

4. Conclusion

This study provides original data on windthrow after green tree retention in two contrasted regions of Eastern Canada. Results show increased mortality in comparison with uncut natural stands. However, such an increase in mortality should also be expected for residual trees left after a stand replacing disturbance. Whether this increase is critical or not in terms of maintaining biodiversity remains to be established. This could depend on the ecosystem studied and the level of retention.

Results for the Abitibi region show high levels of windthrow after a short period of time (not more than 5 years). Differences between group and dispersed retention were relatively minor. Windthrow modeling for this region showed several trends that are contrary to the existing literature. It is likely that the relatively small sample size for some species and some confounded effects played a role and that these models should be used with caution.

For the North Shore region, windthrow levels were lower, especially for group retention and modeling results were more in accordance with the literature. Among the variables retained in the models for explaining windthrow were height, dbh, slenderness ratio, species, surface deposit, drainage, and sapling abundance. Modeling results from the North Shore region, as well as regional differences, suggest that original stand structure played a significant role in determining the incidence of post-cut windthrow.

This study also provides decision support tools for identifying the stands most vulnerable to windthrow. If the intent is to minimize windthrow, selecting stands with well-developed structure and favoring group over dispersed retention are good strategies. For the eastern Canadian boreal, black spruce is less vulnerable to windthrow than balsam fir so should be preferentially selected for retention. However, the level of windthrow that can be accommodated without compromising the objectives of the treatment remains an open question and depends on the objectives themselves. Moreover, objectives other than the operational goal of minimizing short-term windthrow could require the use of dispersed retention or the use of retention in more regular-structured stands. The present study provides data to better anticipate treatment effects in these conditions and to possibly adjust the level of retention to be prescribed.

Although this study involved an important field effort, the number of cutovers sampled was limited and reflects the relative novelty of this treatment in Quebec. Additional studies would be desirable, especially in Abitibi. As with any new practice, the application of variable retention treatments is rapidly evolving, and as such, windthrow losses may be expected to become better controlled in the future.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.foreco.2011.12.018.

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