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Forest structural attributes after windthrow and consequences of salvage logging

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ABSTRACT

In the eastern boreal forest of Quebec (Canada) windthrow is a major natural disturbance, given the long fire cycle interval. Understanding windthrow is essential to ecosystem-based forest management. Dead wood, live trees, and pit-and-mound microtopography are major post-windthrow attributes with known ecological importance. So far, these structural post-windthrow attributes have not been described for this ecosystem. In addition, ecological consequences of salvage logging after windthrow remain unknown, with no specific salvage standard being applied to maintain such attributes and biological legacies. In this study, comparisons were made between salvaged and unsalvaged windthrow to identify which post-windthrow attributes were more greatly affected by harvest operations and to clarify management options. Downed coarse woody debris (downed CWD), snags, live trees, and pits and mounds were characterized. We showed that downed CWD and snags diminished after salvage operations, with a more uniform distribution among decay classes. Pit and mound density was reduced after salvage logging compared to unsalvaged windthrow, with pits being smaller in the salvaged plots. From an ecosystem management perspective, retention patches with dead wood and standing living trees should be kept in salvaged cut-blocks. To minimize salvage operation effects on microtopography, machinery trails should be reduced to a minimum. Also, a certain proportion of windthrow should be exempted from logging operations.

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1. Introduction

Ecosystem management, also known as natural disturbance-based management, has become the standard for forest management, not only in many regions of Canada, but also throughout the world (Mitchell and Beese, 2002; Fischer et al., 2006). Understanding natural disturbance is essential for establishing silvicultural treatments that reduce the disparity between natural and managed ecosystems (Gauthier et al., 2009). In the eastern boreal forest of Quebec (Canada) ecosystems can be affected by several episodes of windthrow, both partial and stand-replacing, which make windthrow one of the most important types of natural disturbance in the region (Ruel et al., 2010; Waldron et al., 2012). The higher occurrence of windthrow in the eastern compared to the western part of the boreal forest can be explained by the longer fire cycle in the former (Bouchard et al., 2008). While most studies of natural disturbance in the boreal forest of Quebec have focused on wildfire or insect outbreaks, an increased understanding of windthrow is essential for improving our understanding of forest dynamics, particularly in forests with long forest fire intervals, such the northeastern boreal forest of Canada (Bouchard et al., 2009; Waldron et al., 2012).

Windthrow creates attributes and biological legacies within ecosystems, including dead wood, pit-and-mound microtopography, and seedbed diversity (Beatty and Stone, 1986; Schaetzl et al., 1989; Ulanova, 2000). The ecological importance of snags and woody debris on the ground has been demonstrated repeatedly (Siitonen, 2001; Jonsson et al., 2005). Moreover, organisms that use dead wood are associated with either one or with many specific categories of dead wood and, thus, decay class and size both play a role from an ecological standpoint (Caza, 1993; Siitonen, 2001; Jonsson et al., 2005). Pit-and-mound microtopography refers to the slight surface elevations and depressions that are formed by tree uprooting. Uprooting mixes the soil, increasing nutrient element availability (Beatty and Stone, 1986; Ulanova, 2000). In certain forest ecosystems, these features can cover a relatively high proportion of the forest floor (Ulanova, 2000). This particular form of microtopographic disturbance also exposes or creates a variety of seedbeds, thereby promoting the germination and the growth of different plant species (Peterson and Campbell, 1993; McCarthy, 2001). From the perspective of natural-based management, these post-windthrow attributes should be described.

In many parts of the world, salvage operations are undertaken following episodes of natural disturbance. Effects of salvage log-

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ging following fire have recently received considerable attention from the scientific community (Nappi et al., 2004, 2011; Greene et al., 2006). Yet, the ecological consequences of salvage logging after windthrow largely remain unknown in Quebec and there is no specific standard by which this technique can be implemented to maintain post-windthrow attributes. This lack of information should induce caution in our current management practices. Indeed, by looking at studies conducted elsewhere in the world on post-windthrow salvage logging (Loeb, 1999; Greenberg, 2001; Lain et al., 2008), we can suppose that key structural attributes as dead wood, pit-and-mound microtopography and live trees would be reduced by salvage logging operations. The comparison between salvaged and unsalvaged windthrow could help highlight those post-windthrow attributes that are more affected by harvest operations (Gauthier et al., 2009) and improve management choices. Therefore, the aims of our study were: (1) to characterize post-windthrow structural attributes and (2) to compare those attributes with a post-windthrow salvage-logged area.

2. Materials and methods

2.1. Study area

The study was conducted in the eastern black spruce-feather moss subdomain of the boreal forest, which lies within the North Shore administrative region of Quebec, Canada (Fig. 1). Black spruce (*Picea mariana* (Mill.) B.S.P.) and balsam fir (*Abies balsamea* (L.) Mill.) are the dominant tree species, but white birch (*Betula papyrifera* Marsh.) and trembling aspen (*Populus tremuloides* Michx.) can also be found. The area is characterized by a fire low occurrence because of its humid and cold climate. Fire cycles on the North Shore range from 270 to over 500 years (Cyr et al., 2007; Bouchard et al., 2008). Mean annual precipitation is about 1300 mm and mean annual temperature ranges between −2.5 °C and 0 °C. The main surface deposit is till, with rock outcrops frequently occurring at the tops of steep slopes (Robitaille and Saucier, 1998).

Given the long fire cycles in the region (Bouchard et al., 2008), windthrow and outbreaks of spruce budworm (*Choristoneura fumiferana* (Clemens)) are the main natural disturbances in this ecosystem, which is affected by gap dynamics (Pham et al., 2004; De Grandpré et al., 2009; Bouchard and Pothier, 2010). Stands with irregular age structure represent a large proportion of black spruce forests of the area (Boucher et al., 2003). In recent years, the region has been severely affected by several windthrow episodes. In 2003, a partial windthrow occurred followed by a major windthrow event in 2006. In total, over 88,000 ha were affected by windthrow (both partial and severe) during this period. A major salvage plan followed the 2006 windthrow. The ensuing salvage operations affected more than 20,000 ha (Ruel et al., 2010).

2.2. Data collection

Salvage logging plans of the main forest products company, Resolute Forest Products, were used to select salvaged sites. Salvaged sites had been harvested in the summer of 2007 or 2008. Not only are downed trees harvested during logging but standing trees are also removed, which helps to compensate for higher logging costs (Roy, 2008) and the reduced wood quality of dead trees (Ruel et al., 2010). Harvesting was conducted with a single-grip harvester and a forwarder. These harvest machines had inflatable tires with chains to increase traction (Jean-François Gauthier, personal communication). Cut-blocks having similar stand characteristics and a wide range of windthrow severities were chosen. High resolution aerial photographs that were provided by the Quebec Ministry of Natural Resources and Wildlife (MRNFQ), together with field estimation, were used to determine windthrow severity of each plot. These severities were distributed over four classes, which were: (1) 0–24%, (2) 25–49%, (3) 50–74%, and (4) 75% or greater stand mortality. Six cut-blocks of approximately 25 ha were sampled. In each cut-block, sampling followed a systematic approach, with a plot (0.04 ha) established every 100 m. Unsalvaged plots were grouped in 12 blocks, based on initial stand and soil characteristics, determined with the forest inventory maps provided by MRNFQ. The number of blocks was higher in unsalvaged treatment as it was not possible to use a systematic approach because of safety and accessibility issues. A relatively wide range of windthrow severities were represented in sampled plots, but, because of safety

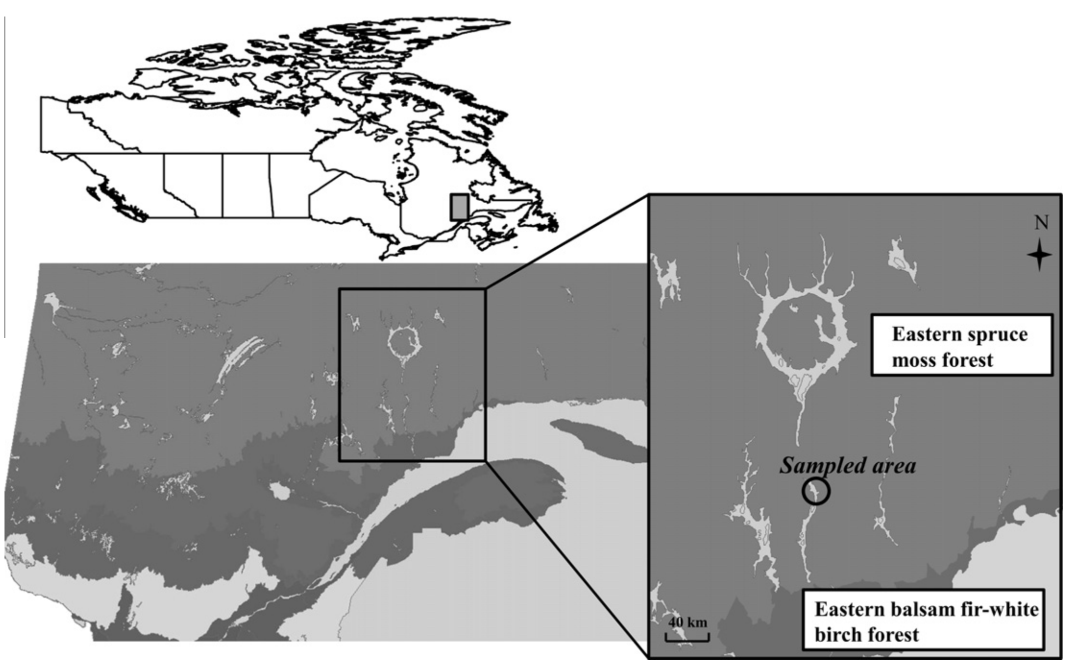

Fig. 1. Study area.

issues, most of the plots were restricted to severity classes 1–3 (Table 1).

Data were collected during the summers of 2008 and 2010. A total of 137 plots of 11.28 m radius (0.04 ha) were sampled. Ninety-four of these plots were located in salvage logging and 43 were located in unlogged windthrown areas (Table 1). After salvage logging, downed coarse woody debris (downed CWD) and standing dead and living trees were measured on all 94 plots during summer, 2008. Microtopographic features were measured only on 49 plots in 2010, because the other 45 plots had been scarified between 2008 and 2010. All measurements in the unlogged plots were taken during summer, 2010.

Downed CWD were quantified and described in the main plot of 400 m^2. We measured all downed material in plots with diameters greater than 9 cm and lengths greater than 30 cm. Downed material volume was estimated with the Smalian's formula by taking both end diameters plus the length. Even if some authors have suggested other formula for wood estimation (Fraver et al., 2007), we chose Smalian's because it is the one used by MRNFQ and forest industries of Quebec (Lemieux, 2011). When debris on the ground was a whole tree, butt (large end) diameter was taken and length measured until the diameter had decreased to 9 cm. When the log was crossing plot edge, only the portion within the plot was measured. Downed CWD were also classified according to Hunter's decay stages (Hunter, 1990; Hunter and Schmiegelow, 2011). This deadwood classification system or similar systems are widely used in Quebec and elsewhere (Jenkins et al., 2004; Webster and Jenkins, 2005; Cimon-Morin et al., 2010; Barrette et al., 2012). It can be directly used in the field to characterize degradation of trees in the forest. There are two Hunter classifications, one for standing trees and one for trees on the ground. There are five classes for downed wood material. Class 1 is characterized by an intact bark and wood texture with the presence of twigs. Class 2 also as an intact bark but there is no more twigs on the log. Classes 3 and 4 have lost almost all the bark and are sagging near the ground or totally on the ground. Finally, class 5 refers to a soft and powdery wood texture partially covered by bryophytes (Hunter and Schmiegelow, 2011).

Standing dead trees (snags) and standing live trees were also quantified and described in the main plot of 400 m^2. Living trees and snags with a diameter at breast height (DBH) that was >9 cm, were measured. Snags were trees with no green foliage. DBH of both living and dead standing trees were measured and regrouped into classes. Sixteen initial DBH classes have been collapsed to five for subsequent analyses. Snags and living trees were also classified according to Hunter's decay stages (Hunter, 1990). There are seven classes for standing trees. Classes 1 and 2 represent standing living trees and classes 3–7 are for dead trees. Class 3 represents a dead but intact tree, class 4 is trees which are losing bark, class 5 is clean of bark, class 6 is broken and decomposed and class 7 is highly decomposed (Hunter and Schmiegelow, 2011).

Three transects of 19.54 m were placed to form a triangle within each 400 m^2 plot; one of the corners was oriented northward from the plot center. Further, dimensions were measured for tree-fall pits and mounds that were crossed by each transect. Maximal lengths and widths were measured for pits and mounds, as well as pits depth and mounds height. Positions of pits, mounds, and forest floor along each transect were measured to obtain the area occupied by microtopography.

Table 1
Number of salvaged and unsalvaged plots in each windthrow severity class.

Severity class	Unsalvaged plots	Salvaged plots
1	15	5
2	12	48
3	14	25
4	2	16
Total	43	94

2.3. Statistical analysis

Downed CWD volume, snag density, live tree density and pit-and-mound microtopography dimensions and cover proportion were analyzed. ANOVAs ($p = 0.05$) using type III sums of squares were conducted for each of these response variables. Because of a significant Levene's test, we also included a different variance term for each treatment with the varIdent function of R to meet assumptions of homogeneity of variance (Pinheiro and Bates, 2000). Responses variables were $\log_{10}$ or square-root transformed when necessary. In all the analysis, we used mixed-models to account for the nested structure of the data. For the unsalvaged windthrow treatment, plots with similar stand and site characteristics were regrouped into blocks, as were plots within the same cut-block for the salvage logging treatment. Blocks and plots were used as random factors. The analyses were conducted using the packages nlme, car and multicomp in the R statistical environment (R-Development Core Team, 2011). For downed CWD volume and snag density, the influence of treatments, windthrow severity, decay classes and their interactions were tested. For live tree density, the influence of treatments, windthrow severity, and DBH classes and their interactions were tested. With contingency-tables and χ^2 tests, we compared CWD volume, snags density and live trees density distributions between treatments. The effect of treatments on pit-and-mound microtopography area, sizes and proportions were tested. Microtopography characterization did not include windthrow severity classes because only one value of forest floor, pits and mounds was present for severity class 4; thus, it was not possible to calculate a standard error for this treatment.

3. Results

3.1. Downed coarse woody debris

When comparing the mean volumes and lengths of downed CWD, significant interactions were noted between treatments and severity, and between treatments and decay classes (Table 2). Distribution of downed CWD among decay classes was different between treatments ($\chi^2 = 1118.9$, $df = 4$, $p < 0.001$). The total mean volumes of downed CWD were 80.3 m^3/ha and 43.8 m^3/ha for unsalvaged and salvaged windthrow respectively. Partial χ^2 tests done between treatments showed a significant difference for all the decay classes ($p < 0.001$). Mean downed CWD volume is greater in the unsalvaged treatment than in the salvaged plots for decay classes 2–5. In the unsalvaged plots, decay classes 2 and 5 had the greatest volume of downed CWD (26.47 ± 5.52 m^3/ha and 20.08 ± 3.34 m^3/ha) and in the salvaged plots, decay classes 2 and 3 had the greatest volume of downed CWD (14.63 ± 1.52 m^3/ha and 16.33 ± 1.36 m^3/ha) (Fig. 2).

Distribution of downed CWD among windthrow severity classes was different between treatments ($\chi^2 = 471.1$, $df = 3$, $p < 0.001$). Partial χ^2 tests conducted between treatments showed a significant difference for windthrow severity classes 2–4 ($p < 0.001$). There was no difference between treatments for severity class 1 ($\chi^2 = 1.7$, $df = 1$, $p = 0.18$). Volume of downed CWD for severity classes 2, 3 and 4 in the windthrow treatment was higher than in the salvage logging treatment. The increase of downed CWD volume with the severity is more pronounced for the unsalvaged treatment than for the salvaged one (Fig. 3).

Table 2
ANOVA summary of linear mixed-effect models for the effects of treatments, decay classes, and windthrow severity on CWD volume and length and snag density. Significant effects ($p < 0.05$) are indicated in bold.

Source	Num. *df*	CWD volume		CWD length		Snag density	
		F	*p*	*F*	*p*	*F*	*p*
Treatments	1	13.03	**0.0026**	50.09	**<0.0001**	85.68	**<0.0001**
Decay classes	4	58.03	**<0.0001**	8.06	**<0.0001**	12.27	**<0.0001**
Severity	3	8.63	**<0.0001**	4.99	**0.0027**	0.12	0.9494
Treat∗Decay	4	7.11	**<0.0001**	18.74	**<0.0001**	11.95	**<0.0001**
Treat∗Severity	3	5.36	**0.0017**	5.16	**0.0022**	0.49	0.6910

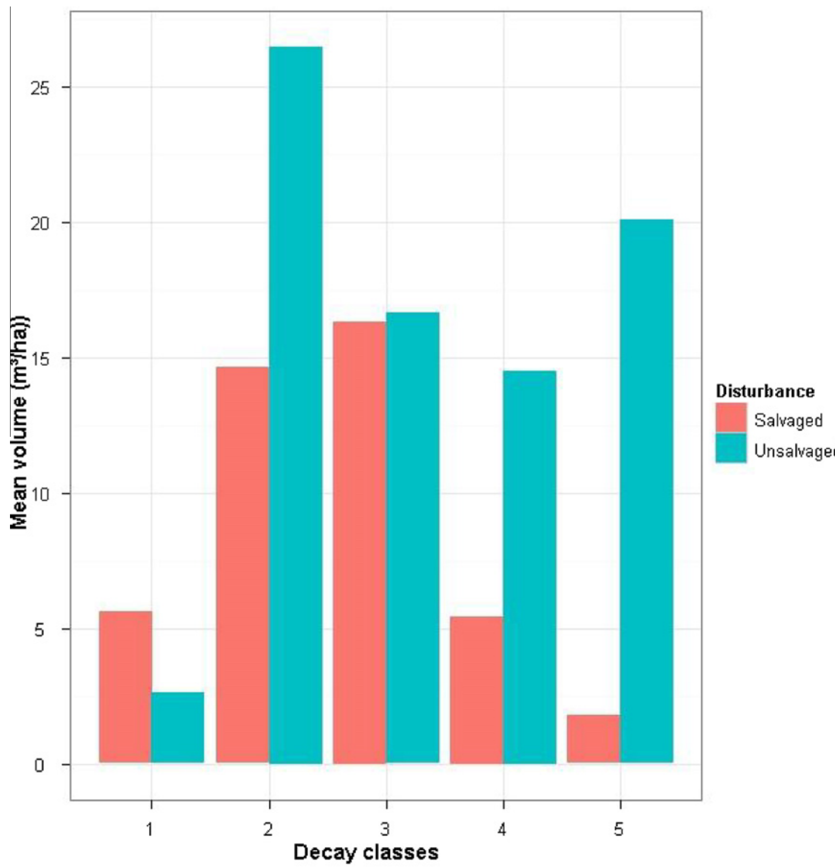

Fig. 2. Mean volume of downed CWD (m^3/ha) after windthrow were salvaged or left unsalvaged, as a function of decay class.

3.2. Snags

Mean snag density differed between treatments, but this difference depended on decay classes (treatments∗decay classes interaction). Windthrow severity had no significant effect (Table 2). Distribution of snag density among decay classes was different between treatments ($\chi^2 = 62.4$, $df = 4$, $p < 0.001$). The total mean densities of snags were 312.5 stems/ha and 41 stems/ha for unsalvaged and salvaged windthrow respectively. Partial χ^2 tests done between treatments showed a significant difference for decay classes 3, 5 and 6 ($p < 0.001$). There was no difference between treatments for decay class 4 ($\chi^2 = 3.8$, $df = 1$, $p = 0.052$) and for decay class 7 ($\chi^2 = 0.18$, $df = 1$, $p = 0.67$). Unsalvaged windthrow showed a higher density of snags than salvaged windthrow, regardless of decay class, except for classes 4 and 7, which did not differ from salvaged windthrow. For unsalvaged windthrow, decay class 5 had the highest mean density of snags (121.02 ± 23.09 stems/ha; Fig. 4) and after salvage logging, decay class 6 had the highest mean density of snags (13.95 ± 2.20 stems/ha).

3.3. Standing living trees

Densities of live trees varied among the plots, given significant interactions between treatments and DBH classes, and between treatments and windthrow severity (Table 3). Unsalvaged and salvaged windthrows have been presented in separate figures because of the scale difference. The mean density of living trees was lower in the salvaged treatment than after unsalvaged windthrow, regardless of DHB categories. Distribution of live trees density among DBH categories was different between treatments ($\chi^2 = 33.5$, $df = 4$, $p < 0.001$). Partial χ^2 tests conducted between treatments showed a significant difference for all the DBH classes ($p < 0.001$). For salvaged windthrows, DBH class 10 had the highest tree density (Fig. 5a), with a mean density of 9.31 ± 1.51 stems/ha. In the unsalvaged windthrow treatment, DBH class 12 had the highest tree density with a mean of 375.58 ± 41.16 stems/ha. DBH classes 20 and 24 and more had the lowest mean density (Fig. 5c).

The mean density of living trees was lower in salvaged treatment than after unsalvaged windthrow, regardless of severity

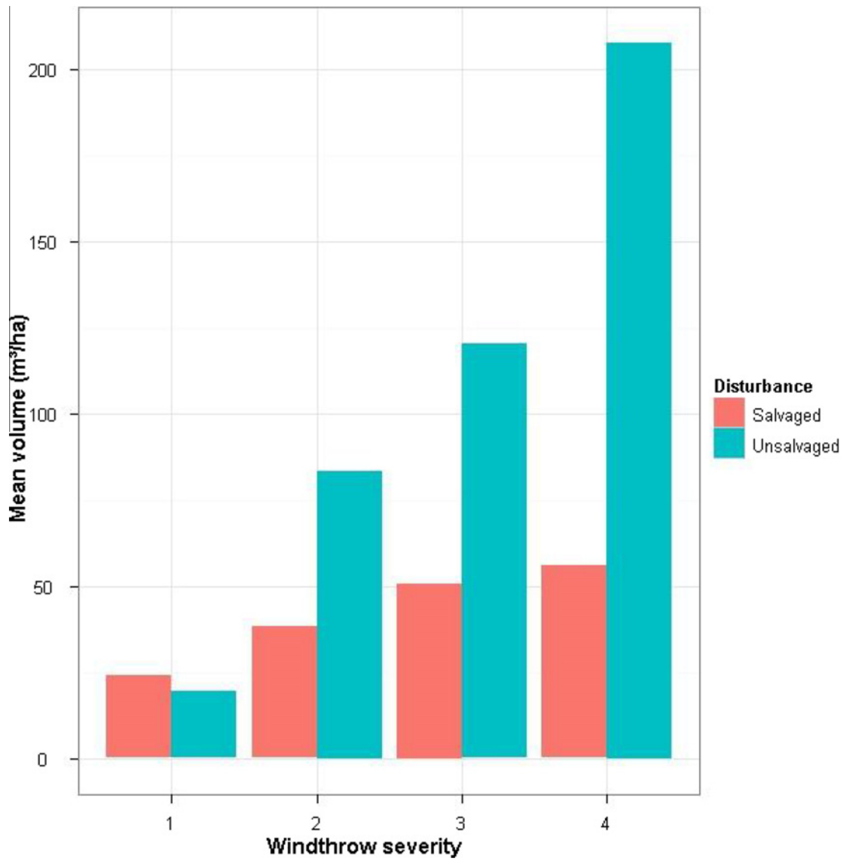

Fig. 3. Mean volume of downed CWD (m^3/ha) after windthrows were salvaged or left unsalvaged, as a function of windthrow severity.

classes. Distribution of live trees density among windthrow severity classes was different between treatments ($\chi^2 = 108.5$, $df = 3$, $p < 0.001$). Partial χ^2 tests done between treatments showed a significant difference for all the severity classes ($p < 0.001$). In salvaged windthrow, live tree density was about the same for all severity class and was always under 4 stems/ha (Fig. 5b). In unsalvaged windthrow, severity class 1 had the highest tree density, with 296.67 ± 31.44 stems/ha (Fig. 5d).

3.4. Microtopography

Pit and mound area was influenced by the type of microtopography and the interaction between the type of microtopography and the treatment. Treatments alone had no significant effects on pit and mound sizes. Considering the treatments and type of microtopography interaction, only pits in the unsalvaged treatment were significantly larger than the other microtopographic types (Table 4). Pit depth and mound height were not influenced by the treatment (Table 4).

Along the transects, nine mounds and eight pits have been encountered in the salvaged plots, and 37 mounds and 31 pits in the unsalvaged ones. In other words, 40% of the plots after unsalvaged windthrow had at least one pit and/or mound compared to 16% of the plots after salvage logging. Mean density of microtopographic features (pits and mounds) by plot were significantly different between treatments, but the type of microtopography (pits or mounds) had no influence on the mean number per plot (Table 4).

Microtopography types (pits, mounds and forest floor), treatments, and their interaction influenced significantly the proportion of transects that were covered by pits, mounds and forest floor. Severity of windthrow did not significantly affect cover proportions of microtopography and forest floor. Cover was similar for pits and mounds, regardless of treatments. The proportion of forest floor (i.e., undisturbed ground surface) was significantly greater than that of pits and mounds in both treatments. The forest floor proportion in unsalvaged windthrows was significantly greater than after salvage logging (Table 4).

4. Discussion

Characterization of post-disturbance attributes and biological legacies is important for measuring the effects of silvicultural practices on ecosystems and for implementing ecosystem forest management (Gauthier et al., 2009). Our study has described, for the first time, the post-windthrow structural attributes of an eastern Canadian boreal forest. It also qualified the alterations to post-windthrow structural attributes that are imposed by salvage logging.

4.1. Dead wood

Our results have shown that mean volumes of downed CWD of all decay classes are affected by salvage logging and that decay class 5 was the most affected by salvage logging. Highly decomposed downed CWD is used by many deadwood-dependent organisms (Siitonen, 2001; Vaillancourt et al., 2008; Jacobs and Work, 2012) and, thus, the substantial amounts of wood occupying advanced decay classes after windthrow could have a major ecological role. Scandinavian studies have shown that many deadwood-dependent organisms are on the red list (Berg et al., 1994; Jonsell et al., 1998; Jonsson et al., 2005). It lets us to think that many

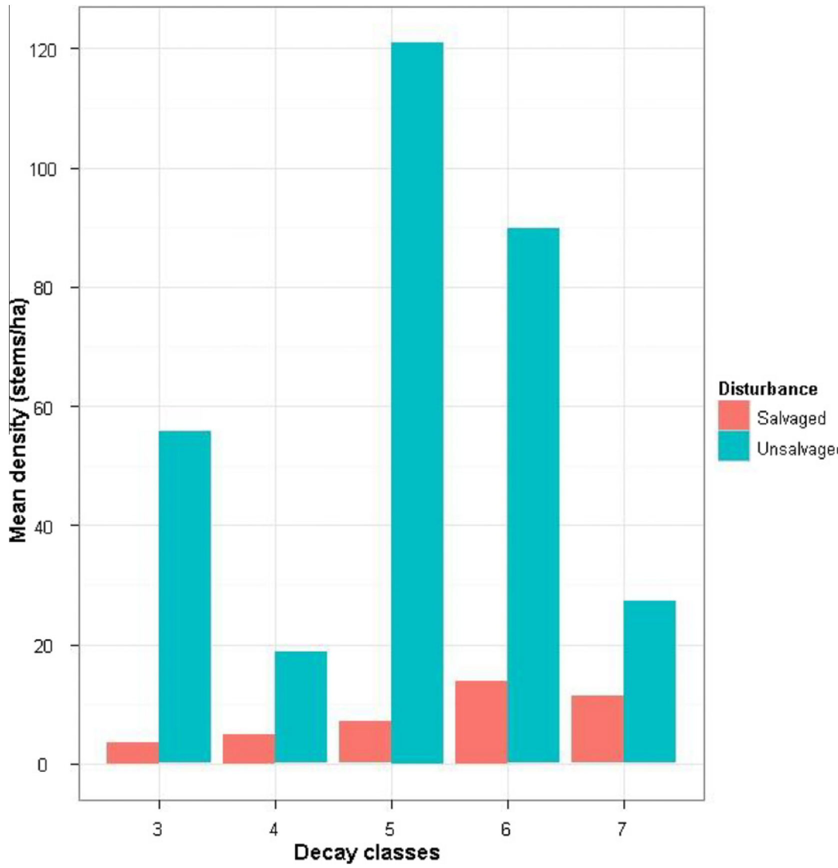

Fig. 4. Snags mean density (stems/ha) after windthrows were salvaged or left unsalvaged, as a function of decay class.

Table 3
ANOVA summary of the linear mixed-effect model of standing live tree density. Significant factors ($p < 0.05$) are indicated in bold.

Effect	Num. *df*	*F*	*p*
DBH	4	26.40	**<0.0001**
Windthrow severity	3	3.52	0.0173
Treatments	1	872.31	**<0.0001**
Treat*Windthrow severity	3	10.74	**<0.0001**
Treat*DBH	4	27.42	**<0.0001**

wood-dependent organisms may be affected by the wood volume reduction after salvage logging operations. Also, field observations have demonstrated a clear trend towards shorter downed CWD lengths after salvage logging compared to unsalvaged windthrow. Differences in lengths or diameters of downed CWD after salvage logging operations have been previously demonstrated in other forest ecosystems (Loeb, 1999; Greenberg, 2001). The absence of large dimension downed CWD after salvage logging can decrease structural heterogeneity of the ecosystem and could affect biodiversity. When salvage operations were preceded by a severe windthrow, no more downed CWD was left in the cut-block relative to that found in salvage logging following partial windthrow. This result showed that a substantial quantity of downed CWD is harvested in salvaged operations, even when a high proportion of trees are on the ground, increasing operational difficulties.

The distribution of downed CWD volume among decay classes in unsalvaged windthrow could be explained by the imposition of two recent windthrow episodes, one in 2003 followed by one in 2006. Within the same area, Ruel et al. (2010) found that downed trees, which has been dead for four years, were already highly degraded. Our sampling was done seven years after the first windthrow event and four years after the second. Thus, trees that had been affected by these events were already degraded, which could explain the low proportion of downed CWD in decay class 1 and the large quantity of degraded downed CWD. The large volume of downed CWD in decay class 2 may be explained by high individual tree mortality following the 2006 windthrow event. In our plots, mean total downed CWD volume was about 80 m^3/ha, which is similar to estimates made for old-growth boreal forest found in eastern Canada (Sturtevant et al., 1997; Cimon-Morin et al., 2010). When considering only windthrows where >50% of the basal area has been affected, mean volume was between 120 and 200 m^3/ha. Our results showed that downed CWD volume in unsalvaged windthrow increases with disturbance severity, which was not a surprising.

In terms of decay stages, snags after salvage logging were not as diversified as after unsalvaged windthrow. In fact, all decay classes after salvaged operations had about the same density of snags, which was not the case for the unsalvaged treatment. In other words, salvage logging results not only in a density reduction of standing deadwood but also a decline in its structural complexity. This reduction could have adverse ecosystem-level effects, since dead wood of different decay classes is an important resource for many different fauna and fungi. The low occurrence of old snags after salvage logging and the lack of recruitment of dead trees can negatively affect bird populations, for example, as snags of later decay stages can support large secondary cavity-nesting species (Vaillancourt et al., 2008).

In unsalvaged windthrows, the distribution of snags among the different decay classes was heterogeneous, which reflects not only the two windthrow episodes but also the continued affects of individual mortality. Aakala et al. (2008) showed that, in the eastern

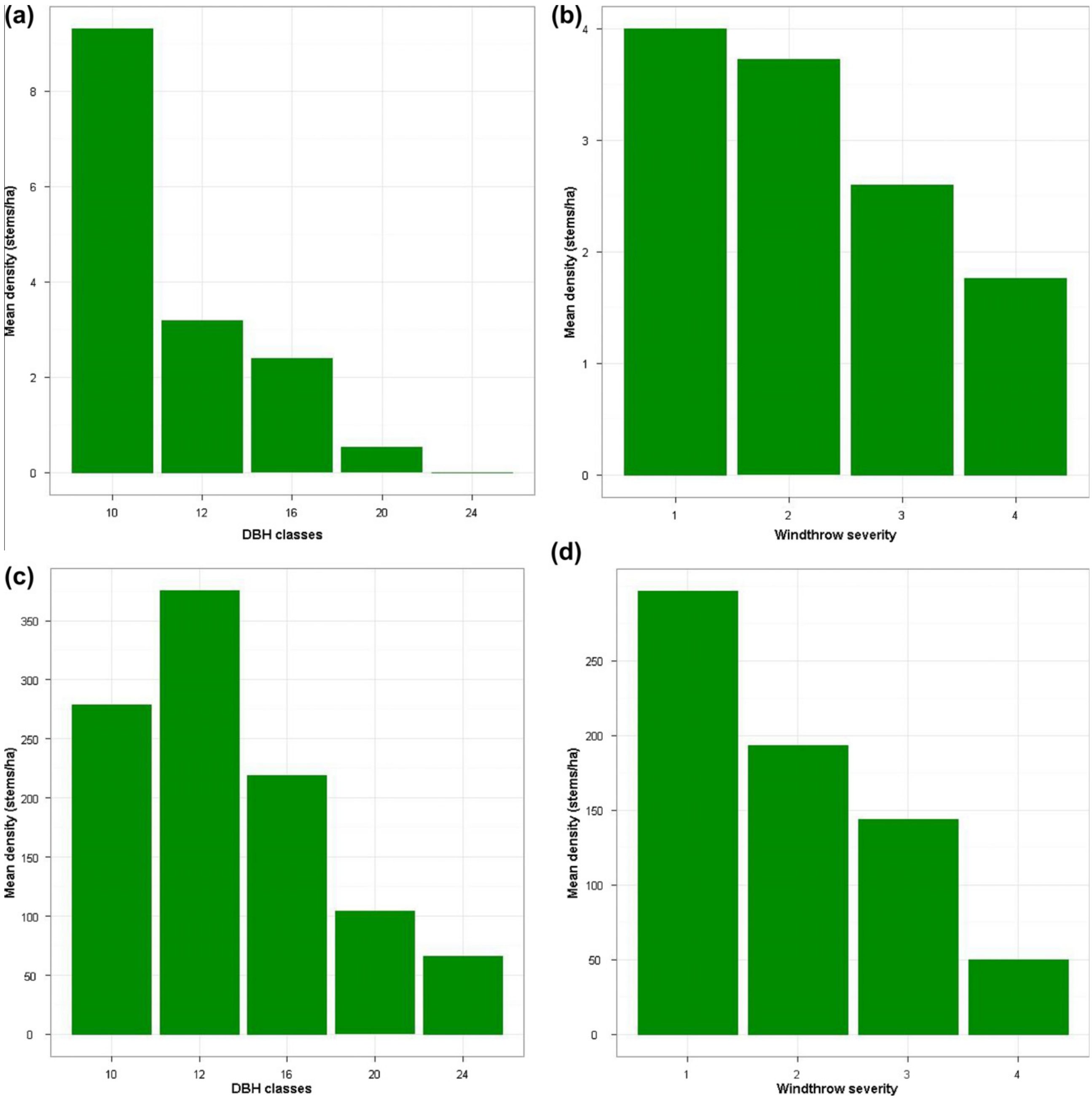

Fig. 5. Mean density of live trees (stems/ha) after windthrows were salvaged (a, b) or left unsalvaged (c, d), as a function of DBH class and windthrow severity.

Table 4
Mean values (±SE) for size of microtopographic features, number of microtopographic features, and mean proportion (%) of transects covered by pits, mounds and forest floor by plot. Within row, values superscripted by the same letter do not differ significantly at $p = 0.05$.

	Salvaged			Unsalvaged		
	Mounds	Pits	Forest floor	Mounds	Pits	Forest floor
Area (m^2)	1.04^{b} (0.21)	1.31^{b} (0.48)	–	1061^{b} (0.18)	2.70^{a} (0.48)	–
Height or depth (m)	0.79^{a} (0.13)	0.23^{b} (0.04)	–	0.97^{a} (0.08)	0.24^{b} (0.03)	–
Number	0.15^{b} (0.04)		–	0.58^{a} (0.12)		–
Proportion of transects	0.27^{c} (0.11)	0.18^{c} (0.09)	99.55^{a} (0.19)	1.13^{c} (0.32)	0.98^{c} (0.36)	97.93^{b} (0.57)

boreal forest of Quebec, the transition of snags between the earliest and the intermediate decay classes occurred rapidly; further, the residence time for snags in intermediate classes was longer than the initial stage. In five years, a substantial proportion of snags in the initial decay classes, therefore, would become and remain intermediate (Aakala et al., 2008). In our study, the high density of snags in intermediate decay class could be the consequence of the 2003 windthrow episode, while the high density of decay class 3 individuals could be the result of the 2006 windthrow. These results accord with those of Aakala et al. (2008), since our data were collected in 2010, i.e., 7 and 4 years after the recent windthrow episodes. Disturbance severity had no effect on snag density. This result could be explained by a higher mortality rate being incurred by uprooting than by stem breakage in black spruce stands (Fleming and Crossfields, 1983; Cimon-Morin et al., 2010). The mean density of snags after windthrow was 62.5 stems/ha in our study site, with a maximum density in the intermediate decay class. The preponderance of snags in intermediate decay classes has been observed in natural forest in the region (Desponts et al., 2004; Vaillancourt et al., 2008). To our knowledge, no studies regarding post-windthrow snag density have been published for old-growth boreal forest. In our study area, Pham et al. (2004) reported a mean snag density of 155 (±54.2) stems/ha in old-growth black spruce forest, while Aakala et al. (2008) found a similar snag density of 131.9 (±45.4) stems/ha. Neither of these studies, though, was conducted on a site that had recently been affected by windthrow. Our results showed that snag density after windthrow is lower than that densities encountered by other authors in the same area when the forest remained unaffected by recent windthrow (Pham et al., 2004; Aakala et al., 2008). This difference could be attributed to the breakage of snags by falling trees in windthrow-affected areas.

4.2. Living trees

Our results have shown that live tree density was strongly reduced by salvage logging and that large trees were largely absent after salvage operations. Trees that were left on cut-blocks were smaller, probably because they are less profitable for industry. Even if most living trees that remain in the ecosystems following windthrow are in the smallest DBH categories, there are still a certain number of large individuals. The definition of ''large tree'' can differ among ecosystems, but in our study area, trees greater than 20 cm DBH were generally considered as such (Vaillancourt et al., 2008). Large living trees in an ecosystem are essential to the fauna (Nilsson et al., 2002) and, if they fall, they would provide considerable downed CWD input. Live trees also contribute to the variation in light and humidity conditions experienced by understory vegetation and seedbeds. The density of living trees is a post-windthrow structural characteristic, but their spatial distribution must also be considered from the viewpoint of ecosystem management. Our study did not evaluate the spatial distribution of trees but an ongoing study in the same area will provide some information regarding this aspect (Waldron et al. unpublished).

4.3. Microtopography

Pit-and-mound microtopography is characteristic of post-windthrow environments (Ulanova, 2000; McCarthy, 2001) and its ecological importance is well-known. Pit-and-mound microtopography increases seedbed heterogeneity (Jonsson and Esseen, 1990; Ulanova, 2000) and thus, can increase plant species richness in comparison to unaffected forest floor (Peterson and Campbell, 1993). This microtopography also promotes tree seedlings establishment (Kuuluvainen and Juntunen, 1998; Sebkova et al., 2012). Thus, in forest ecosystem management, knowledge about post-windthrow pit-and-mound microtopography is essential.

Pit and mound sizes are associated with tree and root system dimensions (Peterson et al., 1990; Clinton and Baker, 2000). In our study, most trees were black spruces and balsam firs that originated from old stands (>70 years old). The size similarities among the uprooted trees found in the two treatments can explain the absence of differences in mound area and height and pits depth. Smaller area covered by pits after salvage logging compared to after unsalvaged windthrow may be explained by the fact that, after tree harvesting, stumps are partially or completely returned to their pre-windthrow position (Doyon and Bouffard, 2008). The number of pits and mounds, together with the proportion of transects that were covered by pits and mounds, was higher after windthrow than after salvage logging, probably for the same reason. Also, logging trails were frequent and penetrated deeply into the soil of harvested sites, which can contribute to reducing the number and cover of pits and mounds after salvage operations. In contrast, pit and mound cover in unsalvaged plots was lower than results that have been obtained in other studies. For example, Peterson et al. (1990) found that 11% of their study area in Pennsylvania was covered by pits and mounds, while Cooper-Ellis et al. (1999) obtained an 8.3% cover in Massachusetts. Harrington and Bluhm (2001) calculated 4.4% for pit and mound area in their study site in Georgia. Pit and mound cover has appeared to be quite variable among forest ecosystems, so comparisons between studies are difficult to make.

4.4. Management implications

Our study showed some differences in structural attributes between unsalvaged and salvaged windthrows. As ecosystem management should be adaptative (Gauthier et al., 2009), studies can and should be used to improve silvicultural and management practices. Even if substantial quantities of windthrow are not salvaged, salvage logging is often concentrated locally, which reduces post-disturbance attributes at a local and regional scale. However, salvaged logging can be performed with caution to minimize negative impacts on the ecosystem (Foster and Orwig, 2006). To preserve a certain quantity of living trees, downed CWD and snags, retention practices should be established in salvage logging operations, with retention patches identified prior to harvest operations. To maintain post-windthrow structural condition, larger patches should be prioritized. These patches should maintain within the salvaged area not only downed CWD and snags of different decay classes, but also living trees with a wide variation in size (Jönsson et al., 2007). Some trees will likely fall in the years following variable retention, particularly larger trees (Jönsson et al., 2007; Lavoie et al., 2012). In black spruce stands with an irregular structure, this proportion remains relatively small (Lavoie et al., 2012) and windthrow that occurs in retention patches could allow deadwood recruitment in salvaged logging blocks.

Thresholds of downed CWD volume or snag density that need to be maintained on cut-blocks are difficult to establish because of the lack of studies regarding relationships between the fauna and dead wood quantity and characteristics in our study area. Variable retention standards that have been used in the area could be applied for now. However, monitoring studies of attributes and the use of retention patches by fauna should be done and thresholds subsequently changed, where required. In addition, to minimize the impact of salvage operations on microtopography, machinery trails should be reduced to their minimum. At the management unit scale, a certain amount of windthrow should be preserved. Thresholds that have been previously suggested in Quebec for salvage logging following fire could be used (Nappi et al., 2011), in a manner that respects the precautionary principle. For instance, 30% of the windthrow that occurred in the last five years could be kept intact, with a certain quantity located beside intact forest. Also, portions of forest that were unaffected by windthrow should be kept intact along the disturbance perimeter, because of their potential ecological roles. Again, silvicultural and management strategies must be implemented in an adaptative fashion, with a monitoring plan in place (Drapeau et al., 2009; Gauthier et al., 2009).

5. Conclusion

This study has described for the first time the post-windthrow structural attributes of irregular-structured boreal forest in eastern Quebec. To apply ecosystem management, an understanding of natural disturbances is essential, and this study has increased knowledge regarding the effects of windthrow. Our results have shown reduction and a more uniform distribution of post-windthrow structural attributes due to salvage logging. Downed CWD, snags, living trees and, to a lesser extent, pit-and-mound microtopography, were all affected by salvage logging operations. This study provided a decision support tool for identifying which post-windthrow attributes are more affected by salvage operations and highlighted the improvement that should be done in post-windthrow forest management to reduce the gap between natural disturbances and managed forests. Other key post-windthrow attributes, as tree regeneration, vegetation biodiversity and seedbed description should also be considered in salvage logging operations. Spatial organization of windthrow should also be studied to choose the sites where salvage logging operation should occur.

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