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Ecosystem Management of Eastern Canadian Boreal Forests: Potential Impacts on Wind Damage

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Abstract: *Research Highlights:* Windthrow can interfere significantly with ecosystem management practices. In some cases, their goal could still be reached but this may prove more complex in other cases, like the partial cutting of old-growth stands. In situations where windthrow is common without any human intervention, the use of partial cutting to maintain some stand structures may lead to a feedback loop leading to additional windthrow. *Background and Objectives:* Forest ecosystem management using natural disturbances as a template has become the management paradigm in many regions. Most of the time, the focus is on fire regime and effects. However, windthrow can be common in some places or can interfere with practices implemented in an ecosystem management strategy. This paper looks at interactions between ecosystem management and windthrow. *Materials and Methods:* The paper builds on three case studies looking at various elements that could be part of ecosystem management strategies. The first one looks at the impact of green tree retention, while the second one looks at the impact of reducing the size and dispersing clearcuts, and the last one examines the impact of a range of cutting practices in irregular old-growth stands. *Results:* Green tree retention leads to increased windthrow, especially when applied within mature even-aged stands. Reducing the size of clearcuts and dispersing them over the landscape also involves substantial windthrow along edges. Partial cutting in old-growth stands can lead to relatively high mortality, but part of it is not necessarily related to wind since it occurs as standing dead trees. Differences in the amount of damage with tree size and species have been found and could be used to reduce wind damage. *Conclusions:* Approaches to minimize wind damage in ecosystem management can be designed using existing knowledge. However, using windthrow as a template to design management strategies would prove more complex.

Keywords: ecosystem management; windthrow; boreal forests; silviculture; partial cutting

1. Introduction

In many parts of the world, natural disturbances are used as a template from which to design silviculture and management strategies [1]. By understanding natural disturbance dynamics and reproducing their main characteristics through harvesting practices, ecosystem management is postulated to maintain the biodiversity and main functions of forests, acting as a coarse-filter approach [2]. Since the natural disturbance regime varies across regions, it is necessary to modulate the design of ecosystem management strategies for a given ecosystem. The boreal forest of eastern Canada has been the focus of intense research efforts to understand natural disturbance dynamics and effects and in the designing of alternative management strategies. In addition, the relative importance of different disturbance agents varies a lot in eastern Canada, providing good opportunities to look at a variety of scenarios.

Fires are more common in drier parts of western Quebec, where estimates of historical fire cycle range from 80 to 300 years [3]. In this region, even though small fires are more common, fires larger than 10,000 ha account for most of the burned area [4]. Even though fire is often taken as the disturbance type to first consider in ecosystem management, treated areas of that size are generally considered not socially acceptable and are not mimicked [5]. Regardless of fire size, some forest patches partially or completely escape fire, providing residual habitat [6]. After such disturbance, new stands develop as even-aged cohorts of fire-adapted species [7].

In wetter portions of the eastern Canadian boreal forest, fire cycles greatly exceed species longevity, reaching even 500 years in places [8]. In this context, other disturbances such as insect outbreaks or windthrow can play a major role [9]. In eastern Canada, spruce budworm (*Choristoneura fumiferana* Clem.) is the main defoliating insect and three outbreaks have been documented in the 20th century [10]. Complete windthrow is generally a sporadic event, but the mean return period for partial windthrow was estimated at 450 years, which is comparable to fire in this region. In rare site conditions, it could even be as low as 40 years [11]. Disturbed areas are generally smaller than those affected by fire and present variable levels of residual trees, depending on disturbance intensity. This variation usually leads to greater stand complexity, typical of intermediate intensity disturbances.

Designing forest management based on natural disturbance regimes requires some important adjustments both at the landscape and at the stand scale. At the landscape scale, the proportion of different age classes would have to be designed to mimic the one that would be associated with the local fire regime. Bergeron and Harvey [12] and Moussaoui et al. [13] have proposed a framework that considers three different successional stages and propose silvicultural practices adapted for each of them. The first-cohort stage corresponds to even-aged stands that have established after fire. In the second-cohort stage, the progressive death of trees that established after fire leads to the development of two-storey stands with an understorey of tolerant species. When stands escape fire for long periods, a more complex structure driven by gap dynamics develops. These stands correspond to third-cohort stands.

In a context of large-scale clearcutting, green tree retention has been proposed to retain the structure typically left after fire. Trees can be kept either as individuals or as groups, providing different ecosystem benefits [14]. Changes in cutblock size [2] have been proposed and the use of partial cutting has been suggested to represent intermediate intensity disturbances [7,15] typical of moderate intensity insect outbreaks or windthrow. Smaller clearcuts involve an increase in the length of edges and concerns have for long been expressed regarding wind damage along newly exposed edges. Similar concerns have been expressed relative to the use of partial cuts in previously unmanaged stands [16]. According to the framework of Bergeron and Harvey [12] and Moussaoui et al. [13], partial cuts should play an important role in the management of second and third cohort stand types.

This paper builds on three case studies conducted across the eastern portion of the Canadian boreal forest in order to look at how windthrow dynamics can be taken into account in designing a forest ecosystem management strategy. It aims at quantifying losses associated with some of the proposed measures of ecosystem management and looks at variables affecting these losses. More specifically, it looks at the impact of green tree retention in clearcuts, of reduced clearcut size, and of using different intensities of partial cutting in boreal stands of Quebec, eastern Canada. Even if these case studies were partially published before, this paper presents a new perspective on integrating these practices in a forest ecosystem management context, using the three cohort framework of Bergeron and Harvey [12] and Moussaoui et al. [13] as a reference.

2. Materials and Methods

2.1. Case Study 1: Green Tree Retention

The first case study is a retrospective study looking at dispersed and group retention in two regions of Quebec differing in soil types, stand structure, and climate. In western Quebec, two sectors were studied (Sector 1: 49°41′ N to 49°37′ N, 78°62′ W to 78°52′ W; Sector 2: 48°47′ N to 48°44′ N; 79°44′ W to 79°40′ W). These sectors are located in the western part of the black spruce (*Picea mariana* (Mill.) B.S.P.)–feathermoss bioclimatic domain and in the western section of the balsam fir (*Abies balsamea* (L.) Mill)–white birch (*Betula papyrifera* Marsh.) bioclimatic domain [17]. Topography is generally gentle and dominant surface deposits are glacial tills and glacio-lacustrine clays. Study stands were 90 years old and developed after fire. These stands correspond to the first-cohort stand type defined by Bergeron and Harvey [12]. The second region is located in the eastern section of the black spruce–feathermoss bioclimatic domain and included three sectors (sector 1: 51°43′ N to 51°25′ N, 68°12′ W to 68°29′ W; Sector 2: 50°25′ N to 50°12′ N, 68°83′ W to 68°73′ W; sector 3: 50°24′ N to 49°72′ N; 69°99′ W to 69°74′ W). Topography was more complex and stands had an irregular or uneven-aged structure, typical of this region with a long fire cycle [18]. These correspond to the second or third-cohort stand types defined by Bergeron and Harvey [12].

In Quebec, group retention aims at maintaining small groups of trees representing 5% of the harvested area [19], whereas dispersed retention aims at keeping 25 trees/ha [20]. Dispersed and group retention were sampled 2–5 years after cutting. This range was considered appropriate to capture the main effects of cutting on windthrow while still being able to differentiate pre- vs. post-cutting mortality. Pre-harvest stands in western Quebec were mostly aspen (*Populus tremuloides* Michx.) dominated, with black spruce, white spruce (*Picea glauca* (Moench) Voss), and jack pine (*Pinus banksiana* Lamb.) as companion species. Only 5% of the stands were pure softwoods. Pre-harvest stands in eastern Quebec were all softwood dominated and 71% of them were dominated by black spruce. Cutover size varied between 8 and 102 ha.

For group retention, a random sample of groups within a given clearcut was selected from aerial photographs, whereas for dispersed retention, transects subdivided into 0.1 ha plots were used. In each group or plot, living and recently dead trees were counted by species and their diameter at breast height (dbh) measured. Height of every tree was measured for dispersed retention only. For the eastern region, the basal area of saplings was additionally measured in a 40 m^2 circle located at the center of the group. Saplings were not measured in the western region because they were rare, which is typical of stands of the first-cohort type. Topex, an index of topographic exposure, was extracted from a database of the Quebec Ministry of Natural Resources and Wildlife, based on a routine developed by Ruel et al. [21]. The effect of cutover size and shape was quantified using the simple fetch index of Scott and Mitchell [22]. This index is calculated by summing the distance across the cutover to the nearest forest edge for the eight main compass directions. To establish a reference for natural mortality in uncut stands, fifteen 400 m^2 plots were sampled in natural stands, close to the logged areas. In western Quebec, average retained group size was 495 m^2 and an average of 60 stems/ha remained in dispersed retention cuts. In eastern Quebec, average retained group size was 367 m^2 and an average of 93 stems/ha remained in dispersed retention cuts.

The probability of windthrow was modeled at the tree level with logistic regression [23]. Regressions were calculated with the lme4 package of R software [24]. Separate analyses were conducted by region and retention type because of failure to converge when grouping them together. In western Quebec, black and white spruce were grouped together, based on their closely related resistance to uprooting for the same tree size [25,26]. Models suggested by prior knowledge of variables influencing windthrow were compared using corrected Akaike information criterion (AIC_c) [27]. The tested variables included mean annual wind speed, Topex, fetch index, slope, surface deposit, organic layer thickness, drainage, species, height, diameter, tree and sapling basal area, stand age, height/diameter ratio, and retained group size. Detailed methods and results can be consulted in Lavoie et al. [28].

2.2. Case Study 2: Small Dispersed Clearcuts

The second case study took place in central Quebec, in the eastern section of the balsam fir–white birch bioclimatic domain [17]. It consists of two contiguous sectors (47°13′ to 47°27′ N, 71°01′ to 71°08′ W), one harvested following existing regulations for public lands, the other harvested using small dispersed clearcutting. Topography is dominated by small hills and surface deposits are mainly glacial tills of varying depths. In the first sector, cutblocks could reach 150 ha and 60 m buffer strips were left between adjacent cutblocks. In the second sector, cutblocks were smaller. According to the management strategy, only 30% of the cutblocks would be larger than 30 ha and none would exceed

100 ha [29]. Recent cuts should not cover more than a third of a given landscape unit, thus not requiring the same buffer strips. In both sectors, 20 m buffer strips were left along streams. Wind damage was studied 5–6 years after cutting. Stands were largely dominated by balsam fir and would mostly correspond to the second-cohort stand type of Bergeron and Harvey [12].

Two different approaches were used to look at the effect of clearcutting dispersion effects. A field sampling was used to look at the effect of stand and site variables on the amount of windthrow. Four clusters of three, 100 m^2 plots, extending 50 m perpendicular from the forest edge, were sampled, one for each perpendicular edge orientation. Living and recently dead trees were sampled in each plot. Age and height were measured on three study trees in each cluster. Topex and the fetch index were calculated as in the previous case study [21,22].

Modeling the effect of variables on wind damage was conducted in two steps because of an excess of plots with no windthrow. Since windthrow was mostly concentrated in the first plot along the edge, only this plot was used for modeling. The first step used logistic regression to explain the presence/absence of windthrow. Prior knowledge of variables influencing wind damage was used to build models that were compared using AICc. The variables tested included: fetch index, Topex, drainage, surface deposit, slope, stand age, height and density, height/diameter ratio, and percent basal area of balsam fir. The second step explained the level of windthrow, using plots where it was present, by using linear regression with a logit transform. Variable selection used a stepwise approach. More details can be found in Larouche [30].

To get a picture of wind damage at the cutblock level, aerial photographs were used. The complete harvesting program of 1995 was analyzed in each sector, using photographs taken in 2000. A 30 m zone around each cutblock was defined and separated into homogenous units in terms of stand characteristics and edge orientation. Stand characteristics were derived from forest cover maps and the amount of damage was estimated by photo-interpretation. Volumes were derived from stand tables. Since the harvested areas and volumes differed by sector, losses were reported relative to edge length and harvested area. More details can be found in Larouche et al. [31].

2.3. Case Study 3: Partial Cutting in Old-Growth Stands

The third case study took place on four sites of northeastern Québec, in the eastern section of the black spruce–feathermoss bioclimatic domain (between 68°00′ and 69°30′ W and between 50°15′ and 51°15′ N). Stands were irregular to uneven-aged, dominated by black spruce and balsam fir, corresponding mostly to the third-cohort stand type of Bergeron and Harvey [12]. Soils were derived from glacial tills and were often thin over bedrock. Steep slopes are common in the area.

Four treatments were compared to an uncut control. The first treatment (CC) is a treatment known as CPRS (Coupe avec Protection de la Régénération et des Sols) in Québec and as CLAAG (careful logging around advance growth) in Ontario [32]. This treatment is the dominant silvicultural practice in Québec and involves the removal of all merchantable stems (dbh > 9 cm) while protecting the advance regeneration. Only a few residual trees remain. The second treatment, CP14 (Coupe avec protection des petites tiges marchandes (CPPTM) in Québec and harvesting around regeneration protection (HARP) in Ontario), is a partial cut that involves the removal of stems with dbh greater than 14 cm. It is operationally used in irregular boreal stands with an abundance of saplings and vigorous small merchantable stems that can respond positively to treatment [33]. This typically corresponds to two-storied, irregular, or uneven-aged stands of the second- or third-cohort stand type of Bergeron and Harvey [12]. The two other types of cutting (SCperm and SCtemp) are variants of the selection cutting method, differing in harvesting patterns. In SCperm, widely spaced permanent main skid trails are used at each entry. Secondary skid trails that are not permanent provide access to the whole area and enable a relatively even distribution of the harvest. With SCtemp, trails are more widely spaced and are not reused at each rotation. The first harvest will occur close to the skid trail and will cover only half of the area, leaving the inter-trail part for the next entry. This means that the harvest is more concentrated than with the previous variant. Both use a rotation length between 60 to 70 years, with a target removal of 35% of basal area. Their aim is to maintain as much as possible the attributes of the third-cohort stand type of Bergeron and Harvey [12].

Treatments were applied at two scales. At the larger scale, treatments were applied over 10–20 ha units. Three permanent plots were established in each of these. This level of treatment is further referred to as the overall treatment. Nested within these large units, smaller units (0.25 ha) were located where each of the treatments was also applied. One permanent plot was established in each of these small units. The treatment associated with these small units is referred to as the local treatment. This nested design makes it possible to look at the impact of local treatment on windthrow when the overall landscape differs.

The ForestGales model is a hybrid mechanistic model that can estimate the critical wind speed for stem breakage and overturning and the associated probabilities of occurrence, given a specific wind regime [34]. Critical wind speeds were calculated with the model, introducing a competition index to account for the position of a given tree within the canopy, as suggested by Hale et al. [35]. Basal area of trees larger than the subject tree was selected as the competition index to be introduced.

First, mixed logistic regression models were built to explain the probability of tree survival, five years after harvesting, and models were compared using AICc. In a second step, mixed logistic models were built to predict the probability of overturning relative to trees dying standing. Additional details can be found in Ruel and Gardiner [36] and Ruel et al. [37].

3. Results

3.1. Case Study 1: Green Tree Retention

Although results could not be compared statistically, wind damage was higher in the western region than in the eastern region. The impact of retention type was not consistent between regions. In the western region, the proportion of windthrown trees reached 41% and 36% for group and dispersed retention, respectively. In the eastern region, damage amounted to 17% and 28% of the number of stems for group and dispersed retention, respectively. Windthrow in reference plots in intact nearby stands was much lower than in any retention treatment, reaching only 4% in the western region and 2% in the eastern region.

Variables that were retained in all four models (two types*two regions) included species, tree size (either height or dbh), soil deposit, and drainage. Many of these variables interacted with species in western Quebec, so that their effect was difficult to generalize. The effect of diameter was not always constant across models but the probability of damage increased with tree size for spruce and fir. In the western region, saplings were not measured because of their low abundance. In the eastern region, they contributed to a reduction of the probability of damage. In this region, balsam fir was more vulnerable than black spruce (Figure 1). Topex and the fetch index were not retained in any of the models. Complete results can be found in Lavoie et al. [28].

3.2. Case Study 2: Small Dispersed Clearcuts

Windthrow was mostly found in the first 10 m around clearcuts, reaching an average of 15% of the basal area. Farther from the edge, it stabilized around 5%. Sixty-seven percent of the edge plots had at least one windthrown tree. Based on the field inventory, the probability of having some windthrow increased with height and to some extent with the fetch index, reaching around 0.9 when tall stands were left along the largest clearcuts (Figure 2).

In plots that experienced windthrow, the level of damage was a function of height, fetch index, stand age, stand density, and percentage of balsam fir (Figure 3). Damage was more important in larger clearcuts and for taller stands. The predicted proportion of windthrow varied between 24% and 48%, whereas the observed proportion varied between 2% and 100%. Hence, even though the model was able to depict the general trends, it was not able to capture the extremes.

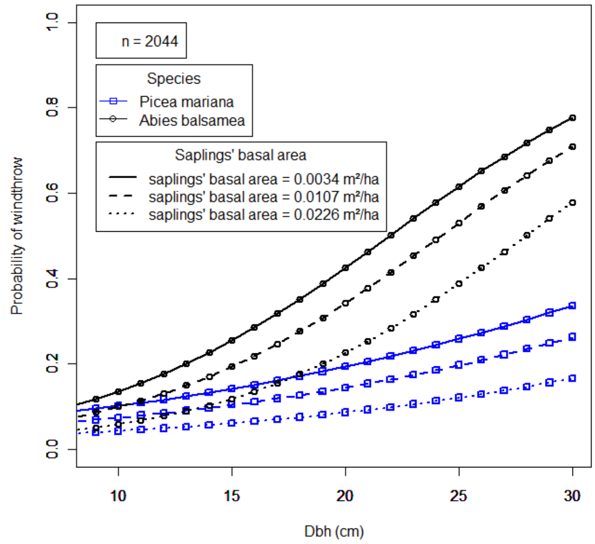

Figure 1. Probability of windthrow for group retention on mesic shallow tills of eastern Quebec (From Lavoie et al. [28]).

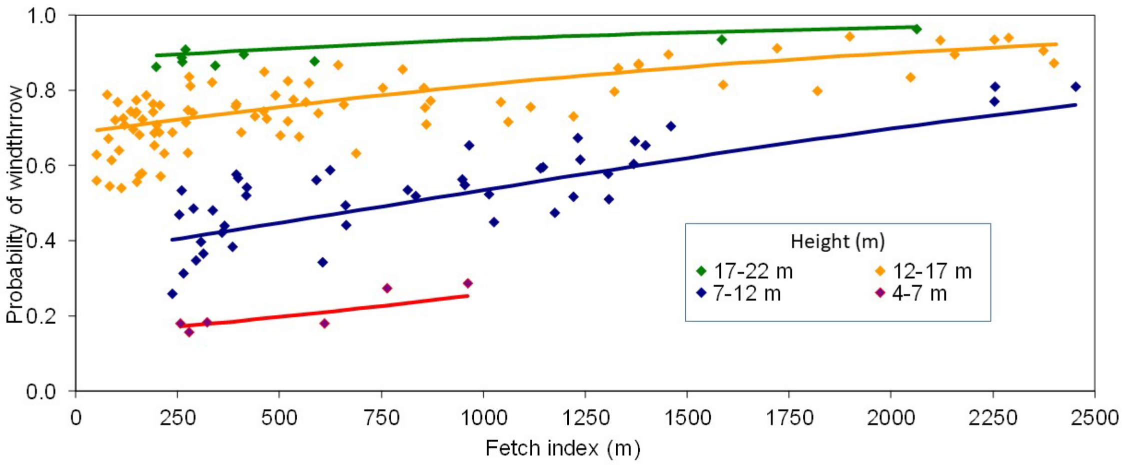

Figure 2. Probability of the occurrence of windthrow in clearcut edges, in relation with the fetch index and stand height [30].

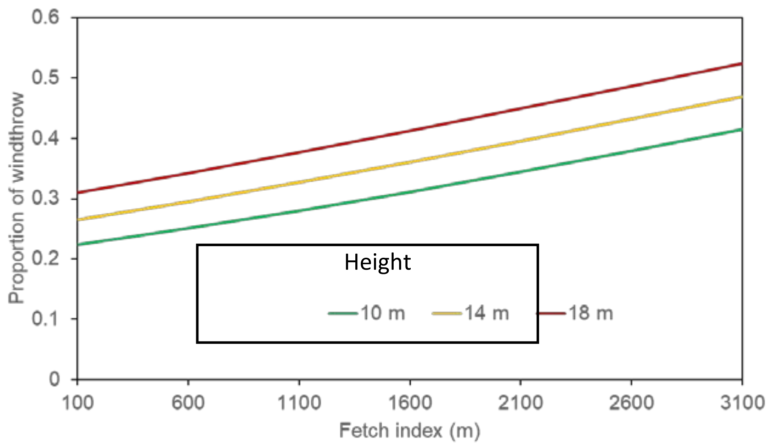

Figure 3. Effect of the fetch index and tree height on the predicted amount of windthrow, considering only cases where windthrow is present. The graph is for an edge stand made of 60% balsam fir, 55 years old, and with a stand density of 2000 stems/ha. Derived from equations of Larouche [30].

Both types of spatial clearcutting patterns differ, not only in terms of clearcut size, but also in terms of the use of buffer strips. In order to compare both strategies, it is necessary to compare them on a common basis. The amount of damage per unit of edge length was quite similar for both types (Table 1). However, the edge length per ha harvested was more than double for the dispersed cuts. This led to much higher losses in dispersed cuts relative to large clearcuts. Losses in dispersed cuts would represent around 10% of stand volume at maturity. Complete results can be found in Larouche [30] and Larouche et al. [31].

Table 1. Synthesis of wind damage for two different clearcutting patterns in balsam fir stands [30].

	Windthrow (m^3)/100 m Edge	Edge Length (m)/ha Harvested	Windthrow (m^3)/ha Harvested
Dispersed cuts	4.9	325	16.0
Large clearcuts	4.3	146	6.3

3.3. Case Study 3: Partial Cutting in Old-Growth Stands

Mortality was rather high, reaching 22% for balsam fir and 16% for black spruce. It was composed of similar proportions of trees that uprooted or died standing; only few trees broke. Total mortality was a function of species, dbh, the local treatment, and the treatment applied around the surveyed area. Mortality increased with dbh and was higher for the most intensive harvests. When applied in the middle of an untreated environment, mortality for these could reach half of the residual trees for the largest balsam fir trees (Figure 4). The environment around the partial cuts can also have an important effect on the level of damage for a given treatment, involving greater losses when locally treated areas are surrounded by clearcuts (Figure 5). Critical wind speeds for overturning calculated with the ForestGales approach were not retained in the best models.

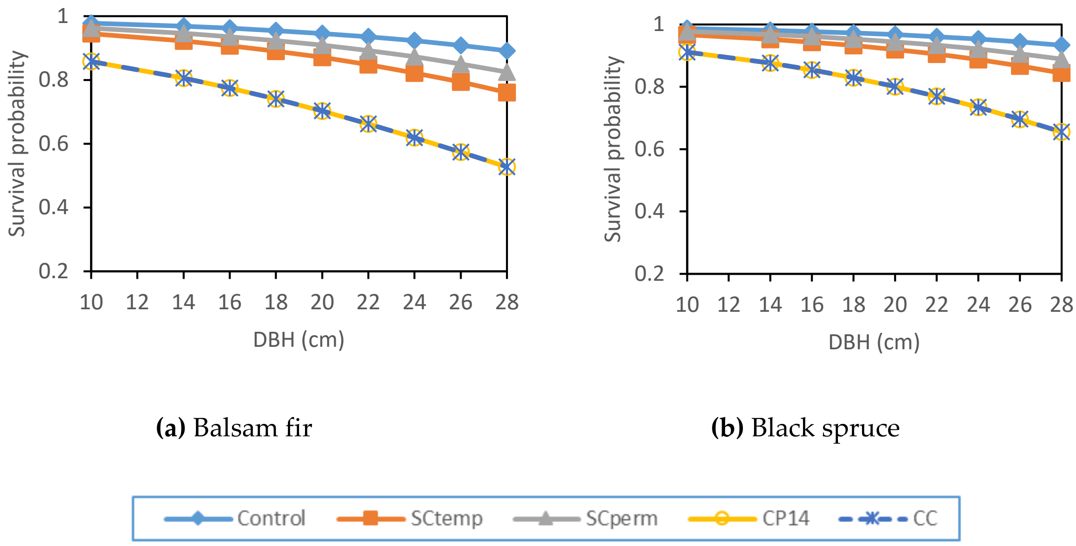

Figure 4. Probability of survival for (**a**) balsam fir and (**b**) black spruce trees according to treatment, when treatments are applied in uncut landscapes [36].

The next step consisted in trying to identify which variables were governing the type of mortality, using only trees that overturned and those that died standing. The best model includes the locally applied treatment and dbh, both factors being significant ($p < 0.05$). The Area Under the Receiver Operating Characteristic (AUC) for this model is 0.6580. According to the best model, the most prominent treatment effect would be a higher relative incidence of standing dead trees in the control where it predominates. In treated stands, overturning predominates, regardless of dbh (Figure 6).

The relative importance of overturning increased with dbh. When looking at absolute mortality values, we see that standing dead tree mortality varied little with dbh, but that the amount of overturning increased with dbh, especially above 22 cm.

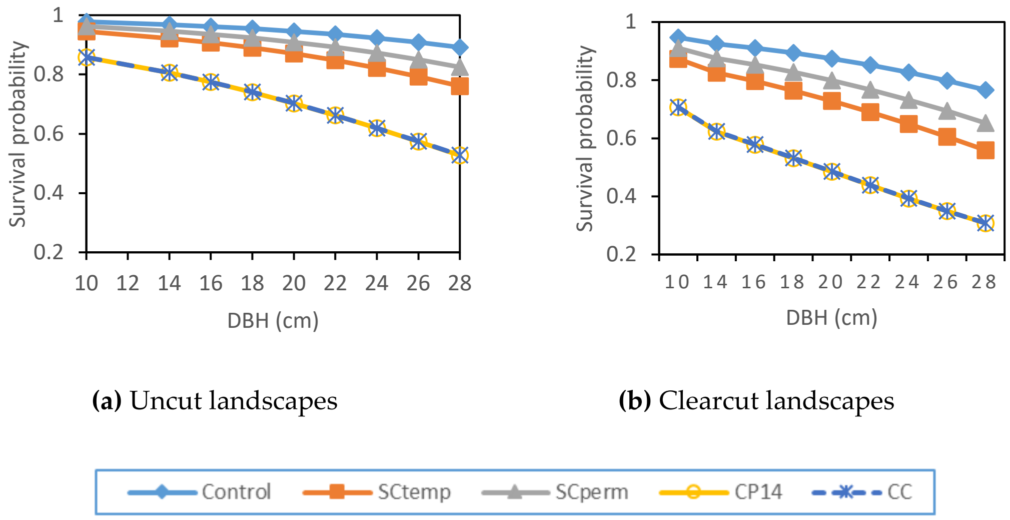

Figure 5. Balsam fir survival probability according to the landscape surrounding the treated area [36]. (**a**) Treated areas within an uncut landscape, (**b**) Treated areas within clearcuts.

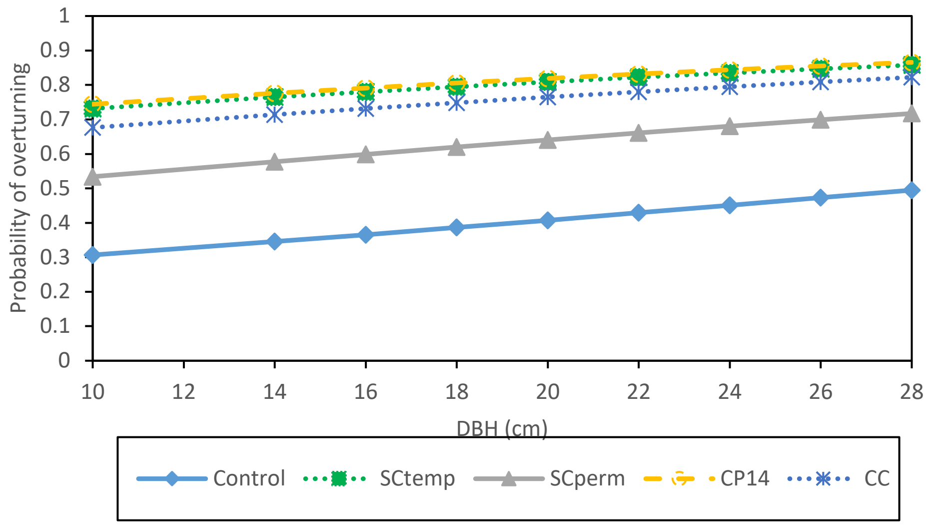

Figure 6. Probability of overturning vs. standing dead mortality, by treatment and dbh [36].

4. Discussion

Integrating natural disturbances in forest ecosystem management requires an understanding of their frequency, their spatial characteristics, and their impact on various ecosystem components. Table 2 presents an overview of these elements for wind disturbance. Even though return periods have been well documented for fire across Canada, little equivalent information is available for wind disturbance. In Quebec, permanent forest inventory plots do not enable to distinguish between mortality due to wind and other types of mortality. Some surveys have been conducted after major storms [38] but these are somewhat anecdotal and tend to focus on stands that could be salvage logged. In eastern Quebec, stand replacing wind damage is not common but partial damage may be quite significant on some sites [11,39]. Spatial characteristics of areas partially damaged by wind are difficult to precisely define due to the fuzzy nature of their limits, but their inner structure remains complex [40]. Partial opening of the canopy by windthrow increases light availability in the understory, releasing advance regeneration of tolerant species when present; more severe damage will make it possible for shade intolerant species to become established. Soil disturbance by tree uprooting will create new regeneration niches for the establishment of light-seeded species, such as birches and spruces. A detailed synthesis of the effect of windthrow on ecological processes can be found in Ulanova [41].

Table 2. Key elements of wind disturbance to guide forest ecosystem management.

	Element	Comments
Disturbance regime	Total windthrow [1]	• Lack of information for many regions; available information often related to salvage logging planning. • Long return periods for eastern Quebec.
	Partial windthrow [1]	• Lack of information for many regions. • Fuzzy limits make it difficult to spatially characterize the disturbance. • Very variable recurrence; return periods can be short in specific cases.
Impact on major ecosystem components	Main canopy	• Reduction in crown cover in relation with disturbance severity.
	Understory	• Increased light in the understory. • Minor destruction of understory unless very severe windthrow.
	Forest floor	• Soil disturbance related to uprooting; little soil disturbance if trees are mostly broken. • Increase of snags and downed woody debris.
Impact on vegetation dynamics	Shade tolerant species	• Release of advance regeneration.
	Shade intolerant species	• Increased abundance related with crown cover reduction.
	Light seeded species	• Regeneration niches created by exposure of mineral soil.

[1] Total windthrow is defined here as windthrow that removes more than 75% of the canopy, whereas partial windthrow removes between 25% and 75% of the canopy. This distinction is derived from local forest inventory standards.

Disturbances are highly variable in space and time, so that a range of measures need to be considered when designing an ecosystem management strategy [42]. Based on the framework of Bergeron and Harvey [12] and Moussaoui et al. [13], stand replacing disturbances should determine the relative abundance of the different stand structures to be maintained across the landscape. The clearcutting system could be used to regenerate stands corresponding to the first cohort type. However, either fire or total windthrow will leave some residual structure and large scale application of this system has been the center of many controversies over the years [43,44]. Among these is the idea that this approach is simplifying stands and landscapes, especially in ecosystems where the fire cycle exceeds species longevity. Such conditions occur in eastern Quebec where the combined occurrence of fire and clearcut harvesting has driven some ecosystems outside their natural range of variability [45]. Even though large fires do occur, mimicking these would likely not be socially acceptable. Thus, moves were made to maintain a minimal structure in harvested areas and to reduce their size.

Green tree retention in clearcuts has been promoted in a variety of boreal or hemiboreal ecosystems over the world [14,46,47]. It should help to maintain some structure in the regenerating stands and provide lifeboating opportunities [46]. Retained trees can also generate dead wood at a time where dead wood production would otherwise be reduced. Hence, these trees are not necessarily expected to contribute to the volume harvested at the next rotation and some wind damage can be tolerated as long as it does not occur too early after harvest when living trees are scarce and dead wood abundant.

Group retention was shown to help reducing losses relative to dispersed retention in one region but the effect was less clear in the other. Studies elsewhere also suggested that aggregating residual trees, especially in larger patches, could reduce mortality [14]. In this study, losses were higher in the western region even though aspen, which was the main species, is considered more resistant that black spruce and balsam fir that dominated in the eastern region. Studies conducted elsewhere did not find regional differences in survival in retention cuts [47]. Regional differences could arise from variations in wind climate or in stand characteristics. According to Saad et al. [48], there would be no marked difference in wind climate between these zones. In the western region, retention was applied in even-aged stands originating from fires, corresponding to the first-cohort stand type. In the eastern region, the treatment was applied in old-growth stands that had escaped fire for long periods and that were governed mostly by gap dynamics (third-cohort stands mostly). Previous work has shown that these different growth environments lead to differences in stem and crown characteristics [49]. Trees in third-cohort stands have been exposed to some extent to the action of wind and other studies have shown that such a preadaptation can be beneficial for survival [47]. Gaps also favor the development of a sapling layer that can reduce the wind load on merchantable trees and, hence, reduce wind damage [50,51]. Recommendations to reduce wind damage and those for ecosystem management could, however, become contradictory. In this sense, the condition that caused the least damage was when group retention was applied when clearcutting old-growth stands. However, in an ecosystem management framework, clearcutting would normally not be the favored approach in such stands since it would lead to a simplification of stand structure, converting stands from the third-cohort type to the first or second-cohort type [52].

Reducing clearcut size would be consistent with trying to mimic intermediate severity disturbances. It could reduce wind damage if residual stands or edge orientation were selected to minimize damage. In our case, stand height had a major impact on the vulnerability of edges to wind damage. Approaches to reduce wind damage along clearcuts have been suggested in many other regions and generally include considerations relative to edge orientation, wind exposure related to topography, soil types, and stand characteristics [53,54].

Reducing clearcut size could help reduce wind damage if it became small enough to prevent wind acceleration over the cut area. However, previous studies have shown that the maximum acceleration is reached with even rather small cutblocks. Gardiner et al. [55] have shown that the effect of gap size is minor beyond 2 tree lengths, so that the reduced size did not lead to a reduction of mortality for a given length of edge in Case study 2. In fact, when the edge trees are mature or close to maturity (height > 12 m), the likelihood of damage is high and the predicted amount of loss is also high, even with small fetch indices. Given the increased amount of edges, losses for a given harvested volume increase, even though the small dispersed patch cutting does not rely on buffer strips to separate harvested areas. In most cases, wind damaged trees in dispersed cuts will not be salvaged since volumes are often not sufficient and harvesting them could mean bringing machinery back through the regenerated area. Complete results can be found in Larouche et al. [31] and Larouche [30].

Different forms of partial cutting have been proposed to manage two-storied, irregular, or uneven-aged stands. The CP14 approach is one way to maintain a minimal structure in these stands, leading to stands of the second-cohort type. However, the impact at the stand level still remains substantial when applied to stands of the third-cohort type [37]. Our study, as well as previous ones, have shown that mortality can be substantial after this treatment [51,56]. Since these trees are expected to survive, respond to treatment, and contribute to the volume at the next harvest, losses need to be maintained within reasonable limits. However, if we consider that most retained trees would be between 9 and 13 cm dbh, losses should not reach very high levels, providing the treatment is not surrounded by clearcuts. This approach is already used operationally in eastern Canada in an attempt to maintain structure, reduce harvesting cost by focusing on larger trees, and shorten the rotation [32].

Lower levels of harvesting, such as those in SCperm and SCtemp, are efficient in maintaining many stand attributes of old forests, and especially of the third cohort type [52] and reducing wind damage to some extent. However, damage still remains substantial and could pose a major challenge when the treatments are applied in old-growth stands with slow growth, located far from markets, such as in Case study 3. In these conditions, overall profitability may be lower than current practices but future cuts may become profitable [57]. Barrette et al. [58] have shown that the amount of advanced decay increases with time since fire, so that stands of the second and third cohort types would present decay levels that could render them more vulnerable to wind damage. Future profitability may become jeopardized with the high mortality experienced in Case study 3.

When considering all three case studies, some common trends relative to the variables influencing the amount of windthrow emerge (Table 3). Species is an important factor to consider; balsam fir has been found to consistently suffer higher damage in all case studies. Since it is also the most affected by both the spruce budworm and the natural windthrow regimes, it would make sense to remove it preferentially during partial cutting. By doing so, we would reach economic objectives by reducing short term losses to wind as well as preventing an increase in balsam fir relative to the initial stand composition. Since the species reproduces prolifically, this would also help to reduce future vulnerability to wind and insect disturbances.

Table 3. Synthesis of variable effects in the three case studies and related considerations for the design of forest ecosystem management strategies. Significance of effects is mentioned for case studies where it was tested.

Variable	Effect in Case Studies	Considerations for Ecosystem Management Strategies
Topography	• Not retained in Case studies 1 and 2	• Tools are available to map the general effect of topography [21].
Surface deposit	• Significant in Case study 1; not retained in Case study 2	• Possible confounded effects.
Soil drainage	• Significant in Case study 1; not retained in Case study 2	• Possible confounded effects.
Species	• Significant in all Case studies	• Balsam fir more vulnerable.
Tree size	• Significant in all Case studies	• Taller stands generally more vulnerable. • Effect of dbh less consistent.
Height/diameter ratio	• Retained in some retention types in Case study 1; not retained in Case study 2	• Slender stems are considered more vulnerable but in interaction with height [59].
Age	• Not retained in Case study 1 • Significant in Case study 2	• Confounded with height in some cases. • Could influence tree vigor and decay incidence.
Clearcut size	• Fetch index not retained in Case study 1, retained in Case study 2 • Effect of overall treatment in Case study 3	• Reduced clearcut size involves more exposed edges. • Smaller clearcut size does not necessarily reduce wind load on new edges. • Partial cuts should not be placed directly beside recent clearcuts.
Tree retention	• Case study 1: effect of retention type not constant between regions • Case study 3: increases in mortality in relation with harvesting intensity	• Group retention could potentially reduce wind damage relative to dispersed retention. • Partial cuts still increase mortality in old stands.

Tree size is also a variable that came out as significant, but the effect of dbh sometimes differed with species. Rosenvald et al. [47] suggested that the effect of dbh is species dependent. However, it is also possible that it could reflect differences in stem shape when tree height does not vary much within even aged stands. The relation between vulnerability to wind and height is clearer, taller trees being more vulnerable. Removing larger trees would enable to reduce wind damage and reproduce some aspects of natural dynamics where old trees are more vulnerable to the spruce budworm and that tall and old firs are more vulnerable to wind damage. It would also have a beneficial effect on treatment profitability, since the largest trees provide the best financial return. The CP14 does that, but the harvest intensity is somewhat high to reproduce the effects of partial windthrow or intermediate severity outbreaks. Stand selection for this treatment needs also to be made to ensure that the small trees have a good response potential and that it does not lead to high-grading. Such a practice when applied in third-cohort stands would involve converting these to the second-cohort type.

The local intensity of tree removal as well as the level of harvesting over larger areas would be important to consider in order to reduce wind damage. Partial cuts should avoid to be placed adjacent to recent total cuts. Previous work has shown that the impact of adjacent cuts could even be more important than the effect of treatment [60]. Topography, as measured by the Topex was not retained in the models of case studies 1 and 2; it was not tested in case study 3. This could be because variations were small, such as in western Quebec, or that other factors, such as harvest intensity had a prevailing effect. Even though soil drainage and surface deposit are known to play a role and have been retained in Case study 1, their effect is not always easy to describe. In fact, confounded effects can be present, taller stands being more frequent in deep, well drained soils associated with the most productive sites. In addition, the resistance to uprooting of some species, such as black spruce, is not very responsive to soil conditions [26].

Models have been developed in many parts of the world to predict the amount of damage along clearcuts or after partial cutting [34]. Empirical models can be developed rapidly but have some limits in terms of potential for extrapolation. Hybrid mechanistic models such as ForestGales have been developed and present a better potential for extrapolation [34]. In Case study 3, the critical wind speeds calculated with a modified version of ForestGales were not retained in the models. There could be many reasons for this. Firstly, the treatment effect includes huge differences in spacing, so that it already captures an important variable in ForestGales calculations. Secondly, the competition index used has performed well in previous tests but may not be as efficient as distance-dependent indices in complex stands such as those studied here. Very important also, the model uses the mechanics of tree uprooting and stem breakage. However, wind damage has been shown to represent roughly half of the mortality only. If we want to improve mortality predictions, we need to better understand factors that are responsible for trees that die standing. It would have been interesting to test individual tree vigor but this variable was not available.

In a context of ecosystem management, windthrow could be seen as a disturbance to mimic or an interference with adapted practices. Past efforts in emulating natural disturbances have focused mostly on fire effects [9]. Less effort has been placed in emulating other disturbances, even though they may play an important role in some regions [5]. Using windthrow as a template for forest management would also be quite challenging. The importance of total windthrow is relatively minor in the ecosystem studied and some of its consequences are shared with fire, major differences being present for the understory layer and the forest floor. Clearcutting could be used to reproduce cohort 1 stand type or to convert other stand types to this structure. However, large scale use of the clearcutting system may lead to important changes in landscape patterns, especially if used in ecosystems with a low fire recurrence [45].

Partial windthrow imprints are often very diffuse and their limits, difficult to identify, making it difficult to decide what characteristics to reproduce. In addition, wind damage does not occur at random but is very specific to some soil and stand types [11]. Nevertheless, various partial cuttings methods could be used to regenerate cohort 2 or 3 stand types and maintain their pre-industrial importance in the landscape. However, using partial cuts in stands where wind damage is more typical may lead to undesired results, since these stands would already be more vulnerable to wind damage. Adopting forest ecosystem management also needs to consider that harvesting does not necessarily replace natural disturbances but can be additive [45] or interact with them [9]. Taylor and MacLean [61] have shown that spruce budworm outbreaks could make residual stands more vulnerable to wind damage, even long after the outbreak. They suggested that the level of partial harvesting be reduced to compensate for expected losses. However, the impact on wood procurement and treatment profitability should be examined.

Using knowledge of past disturbance regimes has also some drawbacks in a context of climate change [62]. Saad et al. [48] have shown that climate change could extend the period when the soil is unfrozen and expose trees to strong late fall winds. In addition, the current spruce budworm outbreak is affecting ecosystems previously not impacted, probably in relation with climate change. Thus, ecosystems may naturally have to face conditions that are outside the historical range of disturbances, posing an important additional challenge in designing forest ecosystem management strategies.

5. Conclusions

Regardless of whether we want to use windthrow dynamics as a template in windthrow prone areas or not, it is worth considering this disturbance when designing ecosystem management strategies. Methods to maintain or convert stands to the first-cohort type are easy to design since they would be based on clearcutting. The options that minimize wind damage would be to retain groups of less vulnerable species, avoiding leaving balsam fir trees as residuals. Wind damage would also be reduced if the practice was applied to convert third-cohort type stands to first-cohort type. However, such a conversion would have a strong effect on species associated with uneven-aged forests [37].

The second-cohort stand type can be maintained in that condition through the use of partial cutting methods such as the CP14 (regionally known as CPPTM or HARP). This approach is feasible and has already been integrated into regular operations. Recommendations on how to minimize losses are available from this study and others [51,63].

Third-cohort type stands could be perpetuated with irregular shelterwood or selection cutting approaches such as those presented here. However, these are more difficult to apply and objectives may not be reached if mortality is too high. Although some of the driving variables have been identified, research is still required to better understand mortality processes in these stands. Much effort has been placed on modeling wind damage in these stands but this type of mortality may represent only half of the mortality. Some stands where vigor is reduced may be converted to the second-cohort type. A more complex structure could be developed in second-cohort stands to move them into conditions of the third-cohort type, enabling to maintain their proportion in the landscape. Research remains needed to test this kind of approach.

Using partial windthrow as a template remains more challenging than the use of fire because of, among other things, its specificity. Reproducing partial windthrow attributes in places where it is a significant disturbance would mean applying partial cuts in vulnerable stands, which could lead to undesired outcomes, both in terms of maintaining a residual canopy and wood production. The level of harvesting could be reduced to control this risk but the overall impact on wood procurement and costs, as well as the protection benefits, would need to be assessed.

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References

1. Keane, R.E.; Hessburg, P.F.; Landres, P.B.; Swanson, F.J. The use of historical range and variability (HRV) in landscape management. *For. Ecol. Manag.* **2009**, *258*, 1025–1037. [CrossRef]
2. Hunter, M.L.J. Natural fire regimes as spatial models for managing boreal forests. *Biol. Conserv.* **1993**, *65*, 115–120. [CrossRef]
3. Bergeron, Y.; Gauthier, S.; Kafka, V.; Lefort, P.; Lesieur, D. Natural fire frequency for eastern Canadian boreal forest: Consequences for sustainable forestry. *Can. J. For. Res.* **2001**, *31*, 384–391. [CrossRef]
4. Bélisle, A.C.; Gauthier, S.; Cyr, D.; Bergeron, Y.; Morin, H. Fire regime and old-growth boreal forests in central Quebec, Canada: An ecosystem management perspective. *Silva Fenn.* **2011**, *45*, 889–908. [CrossRef]
5. Long, J.N. Emulating natural disturbance regimes as a basis for forest management: A North American view. *For. Ecol. Manag.* **2009**, *257*, 1868–1873. [CrossRef]
6. Madoui, A.; Leduc, A.; Gauthier, S.; Bergeron, Y. Spatial pattern analyses of post-fire residual stands in the black spruce boreal forest of western Quebec. *Int. J. Wildland Fire* **2010**, *19*, 1110–1126. [CrossRef]
7. Harvey, B.; Leduc, A.; Gauthier, S.; Bergeron, Y. Stand-landscape integration in natural disturbance-based management of the southern boreal forest. *For. Ecol. Manag.* **2002**, *155*, 369–385. [CrossRef]
8. Foster, D.R. The history and pattern of fire in the boreal forest of southeastern Labrador. *Can. J. Bot.* **1983**, *61*, 2459–2470. [CrossRef]
9. DeGrandpré, L.; Waldron, K.; Bouchard, M.; Gauthier, S.; Beaudet, M.; Ruel, J.C.; Hébert, C.; Kneeshaw, D.D. Incorporating insect and wind disturbances in a natural disturbance-based management framework for the boreal forest. *Forests* **2018**, *9*, 471. [CrossRef]
10. Navarro, L.; Morin, H.; Bergeron, Y.; Montoro-Girona, M. Changes in spatiotemporal patterns of 20th century spruce budworm outbreaks in Eastern Canadian Boreal forests. *Front. Plant Sci.* **2018**, *9*, 1905. [CrossRef]
11. Waldron, K.; Ruel, J.-C.; Gauthier, S. The effects of site characteristics on the landscape-level windthrow regime in the North Shore region of Quebec, Canada. *Forestry* **2013**, *86*, 159–171. [CrossRef]
12. Bergeron, Y.; Harvey, B. Basing silviculture on natural ecosystem dynamics: An approach applied to the southern boreal mixedwood forest of Quebec. *For. Ecol. Manag.* **1997**, *92*, 235–242. [CrossRef]
13. Moussaoui, L.; Leduc, A.; Fenton, N.J.; Lafleur, B.; Bergeron, Y. Changes in forest structure along a chronosequence in the black spruce boreal forest: Identifying structures to be reproduced through silvicultural practices. *Ecol. Indic.* **2019**, *97*, 89–99. [CrossRef]
14. Aubry, K.B.; Halpern, C.B.; Peterson, C.E. Variable-retention harvests in the Pacific Northwest: A review of short-term findings from the DEMO study. *For. Ecol. Manag.* **2009**, *258*, 398–408. [CrossRef]
15. Kneeshaw, D.; Harvey, B.D.; Reyes, G.P.; Caron, M.-N.; Barlow, S. Spruce budworm, windthrow and aprtial cutting: Do different partial disturbances produce different forest structures? *For. Ecol. Manag.* **2011**, *262*, 482–490. [CrossRef]
16. Ruel, J.-C. Understanding windthrow: silvicultural implications. *For. Chron.* **1995**, *71*, 443–445. [CrossRef]
17. Robitaille, A.; Saucier, J.-P. *Paysages Régionaux du Québec Méridional*; Publications du Québec: Quebec City, QC, Canada, 1998; p. 213.
18. Boucher, D.; De Grandpré, L.; Gauthier, S. Développement d'un outil de classification de la structure des peuplements et comparaison de deux territoires de la pessière à mousses du Québec. *For. Chron.* **2003**, *79*, 318–328. [CrossRef]
19. Leblanc, M. *La CPRS à Rétention de Bouquets: Un Nouveau Traitement Sylvicole à Expérimenter*; Ministère des Ressources naturelles de la Faune et des Parcs, Direction de L'environnement Forestier: Quebec City, QC, Canada, 2004.
20. Leblanc, M. *La Coupe Avec Protection de Tiges à Diamètre Variable: Un Traitement Sylvicole Flexible Ministère des Ressources Naturelles et de la Faune*; Direction de L'environnement Forestier: Quebec City, QC, Canada, 2005; p. 6.

21. Ruel, J.-C.; Mitchell, S.J.; Dornier, M. Mapping wind exposure for windthrow hazard mapping. *North. J. Appl. For.* **2002**, *19*, 183–187. [CrossRef]
22. Scott, R.E.; Mitchell, S.J. Empirical modelling of windthrow risk in partially harvested stands using tree, neighbourhoood, and stand attributes. *For. Ecol. Manag.* **2005**, *218*, 193–209. [CrossRef]
23. Aitchison, J.; Silvey, S.D. The generalization of probit analysis to the case of multiple responses. *Biometrika* **1957**, *44*, 131–140. [CrossRef]
24. R_Development_Core_Team. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2009.
25. Achim, A.; Ruel, J.-C.; Gardiner, B.A.; Laflamme, G.; Meunier, S. Modeling the vulnerability of balsam fir forests to wind damage. *For. Ecol. Manag.* **2005**, *204*, 35–50. [CrossRef]
26. Élie, J.-G.; Ruel, J.-C. Windthrow hazard modelling in boreal forests of black spruce and jack pine. *Can. J. For. Res.* **2005**, *35*, 2655–2663. [CrossRef]
27. Burnham, K.P.; Anderson, D.R. Multimodel inference—understanding AIC and BIC in model selection. *Sociol. Method. Res.* **2004**, *33*, 261–304. [CrossRef]
28. Lavoie, S.; Ruel, J.-C.; Bergeron, Y.; Harvey, B.D. Windthrow after group and dispersed tree retention in eastern Canada. *For. Ecol. Manag.* **2012**, *269*, 158–167. [CrossRef]
29. Bélanger, L. La forêt mosaïque comme stratégie de conservation de la biodiversité de la sapinière boréale de l'Est: l'expérience de la forêt Montmorency. *Le Nat. Can.* **2001**, *125*, 18–25.
30. Larouche, C. *Effet du Patron de Répartition des Coupes et des Variables du Milieu Sur Les Pertes Par Chablis Dans Les Lisières: Cas ce la Sapinière à Bouleau Blanc de l'Est*; Université Laval: Quebec City, QC, Canada, 2005.
31. Larouche, C.; Ruel, J.-C.; Bélanger, L. L'effet du patron de répartition des coupes sur les pertes par chablis: Le cas de la sapinière à bouleau blanc de l'Est. *For. Chron.* **2007**, *83*, 84–91. [CrossRef]
32. Groot, A.; Lussier, J.-M.; Mitchell, A.K.; MacIsaac, D.A. A silvicultural systems perspective on changing forestry practices. *For. Chron.* **2005**, *81*, 50–55. [CrossRef]
33. Riopel, M.; Bégin, J.; Gingras, J.-F. *Une Option Pour la Récolte en Forêt Boréale: La Coupe Avec Protection des Petites Tiges Marchandes*; FERIC: Pointe-Claire, QC, Canada, 2000; Volume 1, p. 8.
34. Gardiner, B.; Byrne, K.; Hale, S.; Kamimura, K.; Mitchell, S.J.; Peltola, H.; Ruel, J.-C. A review of mechanistic modelling of wind damage risk to forests. *Forestry* **2008**, *81*, 447–463. [CrossRef]
35. Hale, S.E.; Gardiner, B.A.; Wellpott, A.; Nicoll, B.C.; Achim, A. Wind loading of trees: Influence of tree size and competition. *Eur. J. For. Res.* **2012**, *131*, 203–217. [CrossRef]
36. Ruel, J.C.; Gardiner, B. Mortality patterns after different levels of harvesting of old-growth boreal forests. *For. Ecol. Manag.* **2019**, *448*, 346–354. [CrossRef]
37. Ruel, J.-C.; Fortin, D.; Pothier, D. Partial cutting in old-growth boreal stands: An integrated experiment. *For. Chron.* **2013**, *89*, 360–367. [CrossRef]
38. Ruel, J.-C.; Benoit, R. Analyse du chablis du 7 novembre 1994 dans les régions de Charlevoix et de la Gaspésie, Québec, Canada. *For. Chron.* **1999**, *75*, 293–301. [CrossRef]
39. Bouchard, M.; Pothier, D.; Ruel, J.-C. Stand-replacing windthrow in the boreal forests of eastern Quebec. *Can. J. For. Res.* **2009**, *39*, 481–487. [CrossRef]
40. Waldron, K.; Ruel, J.-C.; Gauthier, S.; Goulet, P. Comparison of spatial patterns between windthrow and logging at two spatial scales. *Can. J. For. Res.* **2014**, *44*, 740–749. [CrossRef]
41. Ulanova, N.G. The effects of windthrow on forests at different spatial scales: A review. *For. Ecol. Manag.* **2000**, *135*, 155–168. [CrossRef]
42. Mitchell, R.J.; Palik, B.J.; Hunter, M.L.J. Natural disturbance as a guide to silviculture. *For. Ecol. Manag.* **2002**, *155*, 315–317. [CrossRef]
43. Bergeron, Y. Is regulated even-aged management the right strategy for the Canadian boreal forest? *For. Chron.* **2004**, *80*, 458–462. [CrossRef]
44. Yelle, V.; Bélanger, L.; Pâquet, J. Acceptabilité visuelle de coupes forestières pour la pessière noire: Comparaison de la coupe à blanc traditionnelle et de différents types de rétention végétale chez divers groupes d'intérêt issus d'une région ressource forestière. *Can. J. For. Res.* **2008**, *38*, 1983–1995. [CrossRef]
45. Bouchard, M.; Pothier, D. Long-term influence of fire and harvesting on boreal forest age structure and forest composition in eastern Quebec. *For. Ecol. Manag.* **2011**, *261*, 811–820. [CrossRef]
46. Rosenvald, R.; Lohmus, A. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. *For. Ecol. Manag.* **2008**, *255*, 1–15. [CrossRef]

47. Rosenvald, R.; Lohmus, A.; Kiviste, A. Preadaptation and spatial effects on retention-tree survival in cut areas in Estonia. *Can. J. For. Res.* **2008**, *38*, 2616–2625. [CrossRef]
48. Saad, C.; Boulanger, Y.; Beaudet, M.; Gachon, P.; Ruel, J.C.; Gauthier, S. Potential impact of climate change on the risk of windthrow in eastern Canada's forests. *Clim. Chang.* **2017**, *143*, 487–501. [CrossRef]
49. Bergeron, C.; Ruel, J.-C.; Élie, J.-G.; Mitchell, S.J. Root anchorage and stem strength of black spruce (*Picea mariana*) trees in regular and irregular stands. *Forestry* **2009**, *82*, 29–41. [CrossRef]
50. Mason, W.A. Are irregular stands more windfirm? *Forestry* **2002**, *75*, 347–355. [CrossRef]
51. Riopel, M.; Bégin, J.; Ruel, J.-C. Probabilités de pertes des tiges individuelles, cinq ans après des coupes avec protection des petites tiges marchandes, dans des forêts résineuses du Québec. *Can. J. For. Res.* **2010**, *40*, 1458–1472. [CrossRef]
52. Cimon-Morin, J.; Ruel, J.-C.; Darveau, M. Effect of alternative silvicultural treatments on stand attributes in irregular balsam fir-black spruce stands. *For. Ecol. Manag.* **2010**, *260*, 907–914. [CrossRef]
53. Mitchell, S.J.; Hailemarian, T.; Kulis, Y. Empirical modeling of cutblock edge windthrow risk on Vancouver Island, Canada, using stand level information. *For. Ecol. Manag.* **2001**, *154*, 117–130. [CrossRef]
54. Zeng, H.; Peltola, H.; Talkkari, A.; Venäläinen, A.; Strandman, H.; Kellomäki, S.; Wang, K. Influence of clear-cutting on the risk of wind damage at forest edges. *For. Ecol. Manag.* **2004**, *203*, 77–88. [CrossRef]
55. Gardiner, B.A.; Stacey, G.R.; Belcher, R.E.; Wood, C.J. Field and wind tunnel assessments of the implications of respacing and thinning for tree stability. *Forestry* **1997**, *70*, 233–252. [CrossRef]
56. Thorpe, H.C.; Thomas, S.C.; Caspersen, J.P. Tree mortality following partial harvests is determined by skidding proximity. *Ecol. Appl.* **2008**, *18*, 1652–1663. [CrossRef]
57. Moore, T.Y.; Ruel, J.-C.; Lapointe, M.-A.; Lussier, J.-M. Evaluating the Profitability of Selection Cuts in Irregular Boreal Forests: An Approach Based on Monte Carlo Simulations. *Forestry* **2012**, *85*, 64–77. [CrossRef]
58. Barrette, J.; Pothier, D.; Ward, C. Temporal changes in stem decay and dead and sound wood volumes in the northeastern Canadian boreal forest. *Can. J. For. Res.* **2013**, *43*, 234–244. [CrossRef]
59. Becquey, J.; Riou-Nivert, P. L'existence de zones de stabilité des peuplements. Conséquences sur la gestion. *Rev. For. Fr.* **1987**, *39*, 323–334. [CrossRef]
60. Ruel, J.-C.; Raymond, P.; Pineau, M. Windthrow after shelterwood cutting in balsam fir stands. *North. J. Appl. For.* **2003**, *20*, 5–13. [CrossRef]
61. Taylor, S.L.; MacLean, D.A. Legacy of insect defoliators: Increased wind-related mortality two decades after a spruce budworm outbreak. *For. Sci.* **2009**, *55*, 256–267.
62. LeGoff, H.; Leduc, A.; Bergeron, Y.; Flannigan, M. The adaptative capacity of forest management to changing fire regimes in the boreal forest of Quebec. *For. Chron.* **2005**, *81*, 582–592. [CrossRef]
63. Anyomi, K.; Ruel, J.-C. A multi-scale analysis of the effects of alternative silvicultural treatments on windthrow within balsam fir dominated stands. *Can. J. For. Res.* **2015**, *45*, 1739–1747. [CrossRef]