Elevated mortality of residual trees following structural retention harvesting in boreal mixedwoods

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ABSTRACT

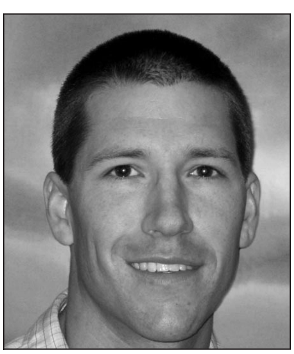
In recent years boreal forests have been harvested to retain a portion of the original canopy, thereby providing forest structure, mostly for biodiversity reasons. Boreal mixedwood cutovers were surveyed at one and five years after harvesting with approximately 10% structural retention, to quantify the mean annual mortality rates of the residual trembling aspen, balsam poplar, paper birch and white spruce trees. For comparison, "natural" mortality rates by species were estimated from permanent sample plots in stands of similar composition. Species ranking of the annual mortality rates of residuals in areas harvested with structural retention were: poplar (10.2%) > birch (8.7%) > aspen (6.1%) > spruce (2.9%). Annual mortality rates were 2.5 to 4 times greater than in the reference stands. The majority of broadleaved species died as snags (~70%–90%), while most spruce died due to windthrow (80%). Mortality rates increased with slenderness coefficient for codominant and understory poplar and for understory birch. For aspen, codominants were most likely to die, while in spruce, dominant trees and trees with the greatest damage to the bole from harvesting operations had the highest mortality.

Key words: Alberta, *Betula papyrifera*, dieback, harvesting damage, mixedwood forests, variable retention, *Picea glauca*, *Populus balsamifera*, *Populus tremuloides*, structural retention, sustainable forest management

RÉSUMÉ

Au cours des dernières années, les forêts boréales ont été récoltées pour retenir une portion de la canopée originale, fournissant ainsi une structure forestière, surtout pour des raisons de biodiversité. À la suite de la récolte, on a vérifié le parterre de la coupe après un an et après cinq ans en ayant conservé environ 10 % de la structure, afin de quantifier la moyenne des taux annuels de mortalité des peupliers faux-trembles, des peupliers baumiers, des bouleaux à papier et des épinettes blanches. Aux fins de comparaison, les taux de mortalité « naturelle » par les espèces ont été estimés à partir des lotissements permanents représentatifs dans des peuplements de composition semblable. Le classement par espèce des taux de mortalité annuelle des arbres rémanents dans les zones récoltées avec une rétention structurelle étaient : le peuplier (10,2 %) > le bouleau (8,7 %) > le tremble (6,1 %) > l'épinette (2,9 %). Les taux de mortalité annuelle étaient de 2,5 à 4 fois plus élevés que les taux dans les peuplements de référence. La majorité des espèces caducifoliées est morte comme chicot (~70 % – 90 %), alors que la plupart des épinettes sont mortes à cause du déracinement par le vent (80 %). Les taux de mortalité ont augmenté avec le coefficient de sveltesse pour le peuplier codominant et dominé et pour le bouleau dominé. Pour le tremble, il était plus probable que les codominants meurent tandis que pour l'épinette, les arbres dominants et les arbres ayant le plus de dommages sur le tronc à cause des opérations de récolte étaient ceux qui avaient le taux le plus élevé de mortalité.

Mots clés : Alberta, *Betula papyrifera*, dépérissement de la cime, dommage de récolte, forêts mixtes, rétention variable, *Picea glauca*, *Populus balsamifera*, *Populus tremuloides*, rétention structurelle, gestion durable de la forêt



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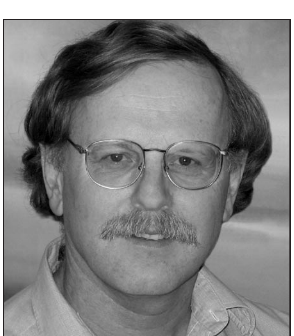

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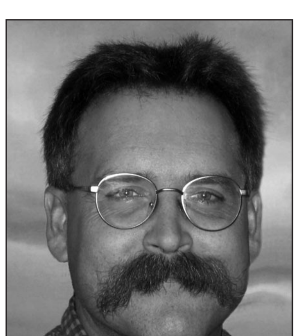

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Introduction

Structural retention harvesting involves the preservation of large, mature trees in dispersed or aggregated patterns in forest cutovers. This silvicultural approach has been inspired by the patterns of natural disturbance (i.e., fire, wind, or disease) in unmanaged forests, which generally leave some standing structure. The stand-level objectives of partial harvesting are to leave a high degree of structural heterogeneity with varying amounts, types and spatial patterns of living and dead trees to address a broad array of forest management goals (Mitchell and Beese 2002). It is assumed that the retention of later seral conditions will contribute to sustaining ecosystem functions and biological diversity at the stand level (Franklin *et al.* 1997, Bergeron *et al.* 2002). Thus, it has been widely promoted and adopted as a forest management strategy in boreal and temperate forest ecosystems (Bourgeois *et al.* 2007, Thorpe and Thomas 2007). However, since the policies and guidelines of this harvesting practice have been developed predominantly from expert opinion (Halpern *et al.* 2005), many details regarding how to best implement retention systems are still unclear.

An accelerated rate of mortality shortly after cutting could compromise the benefits of leaving residual trees. Retention trees are exposed to increased environmental stress from greater evaporative demand, wind exposure and high soil moisture (Franklin *et al.* 1997, Bladon *et al.* 2006). Species that are sensitive to these stresses could die prematurely. Several authors, primarily examining Douglas-fir (*Pseudotsuga menziesii* (Mirb.)) dominated forests of the Pacific Northwest, have noted increased mortality of retention trees following partial harvesting (Franklin *et al.* 2002, Maguire *et al.* 2006). Others have observed relationships between stress and / or mortality with increasing tree height, crown class and slenderness (Coates 1997, Ruel *et al.* 2000, Liu *et al.* 2003). However, mortality of retention trees has been quantified only for a few species and regions. Additionally, the majority of studies have focused on mortality due to blowdown, resulting from wind exposure after partial harvesting (Beurmeyer and Harrington 2002, Scott and Mitchell 2005). Thus, the rates of mortality of retention trees, especially standing mortality (i.e., snag creation), remain poorly understood. Further, little is known if structural characteristics of trees could be used to predict the probability of mortality. It has also been noted that logging damage to residual trees can be extensive, and could lead to increased mortality (Moore *et al.* 2002). However, to our knowledge there have been no attempts to quantify mortality rates of residual trees due to machine damage in tree retention systems.

Our objectives were to quantify the mortality of residual trees for trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*Populus balsamifera* L.), paper birch (*Betula papyrifera* Marsh.), and white spruce (*Picea glauca* (Moench) Voss) five years following harvesting in boreal mixedwood stands, and compare these to mortality rates in similar undisturbed stands. Further, we identified the nature of their mortality (e.g., windthrow, standing dead) and investigated whether mortality of residual trees could be related to variables such as crown class, diameter-at-breast-height (dbh), tree height, slenderness coefficient (height/dbh), and logging damage to the lower bole.

Materials and Methods

Study sites and experimental design

Initial surveys were conducted in 2001, one year after structural retention harvesting of boreal mixedwood sites near Drayton Valley and Rocky Mountain House, Alberta, Canada, in the Lower Foothills Natural Subregion (52° 61′ to 53° 09′ N and 114° 96′ to 115° 32′ W). All sites were harvested during summer by the same contractor. Elevation ranged from approximately 800 m to 1200 m with rolling topography and gentle slopes. The dominant soil types were Orthic and Brunisolic Grey Luvisols. Climate is subhumid and continental, with long, cold winters and mild summers. The mean annual temperature (1980–1999) before the partial harvest was 2.1°C ± 0.5 (95% CI), with a mean monthly growing season (May to September) temperature of 11.7°C ± 0.4. The mean annual precipitation was 555.4 mm ± 29.2, with the majority (~76%) falling as rain during summer. For comparison, the mean annual temperature during the survey period was 1.8°C ± 0.4, with a mean monthly growing season (May to September) temperature of 11.3°C ± 0.6. The mean annual precipitation during the survey period was 561.1 mm ± 96.9.

In May 2001, 55 independent sampling plots (100-m radius) were established within areas which had approximately 10% of the original stand structure and species composition retained during harvesting. Plots were well distributed over an area of approximately 6900 ha. Stands were boreal mixedwoods of fire origin and typical of this region. The dominant species were either trembling aspen or white spruce but there were lesser amounts of balsam poplar and paper birch. Cutting was done in a typical operational manner for this area and both deciduous and conifer species were utilized. Cutover areas contained primarily dispersed retention trees, which were considered trees that were isolated on the landscape or in small patches of fewer than 10 trees. Within each plot, one healthy residual tree from each species (aspen, poplar, birch and spruce) and each crown class was chosen if present. Crown class was identified as either dominant (D; crown in the upper canopy at the time of harvest), co-dominant (CD; slightly shorter, with narrower crowns than D trees), or understory (U; under the main canopy at time of harvest). Trees displaying signs of pathogens, insect defoliation, crown dieback or stem form defects were not included. Measurements of independent variables, including diameter-at-breast-height (dbh; 1.3 m), percent live crown and damage from harvesting (as a percentage of the circumference of the bole) were recorded for each tree. The operator of the harvesting equipment left trees of a range of sizes and species. For all species, the largest trees in our sample were representative of the largest trees in the original stand, based upon observations of stump size and trees in adjacent stands. Thus, the sample that we collected represented the range of tree sizes in the original stand. A total of 505 residual trees, or approximately 40 trees from each species and crown class, were surveyed (Table 1).

Tree condition was re-evaluated in May 2005. Each tree was assigned a condition code, adapted from McCune *et al.* (1988): 0 = healthy tree, all leaves present; 1 = weakened, but mostly healthy, minimal crown dieback (< 20%); 2 = declining, heavy crown dieback (> 20%), small twigs intact; 3 = full crown dieback, dead tree. Dead trees were further categorized similar to Senecal *et al.* (2004): 1 = snag (dead tree, still stand-

Table 1. Characteristics of residual trees surveyed following partial harvesting (mean ± 95% confidence interval)

Species	n	Height (m)	htlc (m)	dbh (m)	SC
Aspen	152	21.9 ± 0.8	10.1 ± 1.1	36.2 ± 2.8	71.0 ± 3.9
Birch	108	17.0 ± 1.2	3.5 ± 0.7	23.8 ± 2.4	79.6 ± 3.8
Poplar	130	20.9 ± 1.1	4.0 ± 0.8	34.0 ± 2.9	69.1 ± 3.3
Spruce	115	18.7 ± 1.3	2.4 ± 0.4	35.4 ± 2.7	54.6 ± 1.6

Abbreviations: htlc = height to live crown; dbh = diameter-at-breast-height; SC = slenderness coefficient.

ing); 2 = broken after death; 3 = broken when alive (trees that were bent, snapped or crushed by other trees); 4 = uprooted (trees that blew over exposing the root system). We distinguished between trees in the dead tree categories 2 and 3 by examining the degree of decomposition and wood structural differences at the point of breakage. The presence of fine branches and leaves, no discoloration of wood and splintering or uneven/jagged wood at the point of breakage were indicative of trees that were alive when broken. All standing trees were measured for dbh, as well as tree height and height-to-live-crown using a Vertex III hypsometer (Haglöf Sweden AB, Langsele, Sweden). Slenderness coefficient was calculated as the ratio of tree height/dbh.

The natural mortality rates of the four species were determined from the long-term permanent sample plot (PSP) data collected by the Alberta Land and Forest Service. The data were collected from 699 locations from a range of stand ages, densities, compositions and site conditions. We selected a subset of plots with similar composition, elevation, slope and location (between 52° 42' and 53° 33'N and 115° 10' and 115° 71'W) to facilitate comparison with our field plots. Similar to our field study, any trees with pathogens, insect defoliation, crown dieback or stem form defects in their first year of measurement were removed from the data set. The final analysis was performed using 29 plots within mature stands, containing 9806 trees.

Mortality rate

The mean annual mortality rate was calculated for each species and each plot separately, and then averaged across all plots. It was also determined for the three canopy classes and for classes of slenderness coefficient. The mean annual mortality rate (m) was calculated as:

$$[1] \qquad m = \left[\left(\frac{N_t}{N_0}\right)/t\right] \times 100$$

where N_t represents the number of dead trees, N_0 is the total number of stems and t is the time interval (years).

Statistical analyses

The experimental designs for comparing the effects of both crown class and the presence or absence of a harvest injury on mortality were essentially randomized complete block (RCB) designs. Plots were considered blocks and all crown classes and typically both injured and uninjured trees were present in each plot. PROC GLIMMIX (Littell *et al.* 2006) was used to capture the RCB design for modeling mortality, which was a binary variable (trees were either living or dead). Multiple comparisons tests among crown classes were made with the Tukey test. Logistic regression (PROC LOGISTIC; SAS Institute Inc. 1999) was used to determine 1) if mortality increased with slenderness coefficient; this analysis was done separately for each of the crown classes and 2) if the probability of mortality increased with increasing proportion of the circumference damaged, for those trees that had been injured. Logistic regression was used because there was typically only one observation per plot. In all analyses, the critical value for statistical significance was $\alpha = 0.05$.

Results

Four years after the initial survey of the trees in the structural retention plots, 42.3% of poplar, 34.3% of birch, 23.7% of aspen and 13.0% of spruce residuals were found dead. The mean annual mortality rates were 9.4% ± 2.3 for poplar, 8.7% ± 2.8 for birch, 5.8% ± 1.9 for aspen and 2.6% ± 1.3 for spruce (Fig. 1). Compared to mortality rates in similar reference plots (PSPs), average annual mortality for residual poplar was 4.0 times greater, for aspen 3.4 times greater, for birch 2.8 times greater, and for spruce 2.3 times greater.

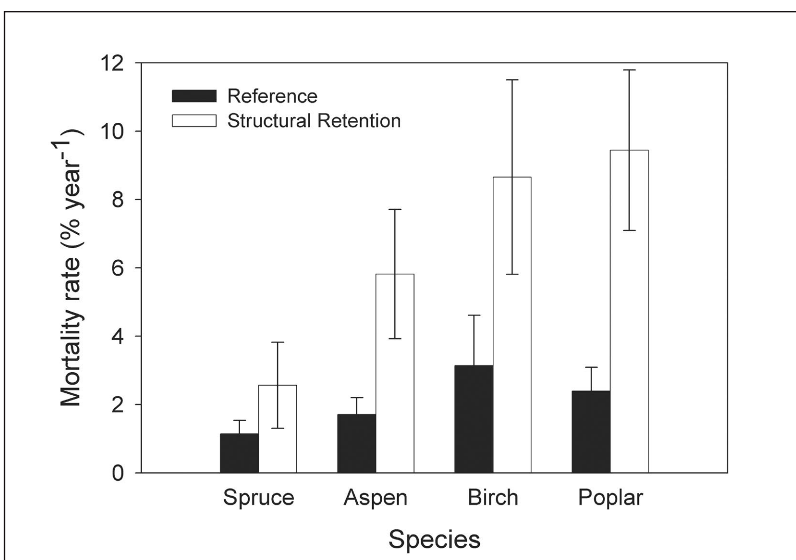

Fig. 1. Mean annual mortality rates (± 95% confidence intervals) from the reference and structural retention harvested plots for white spruce, trembling aspen, paper birch, and balsam poplar.

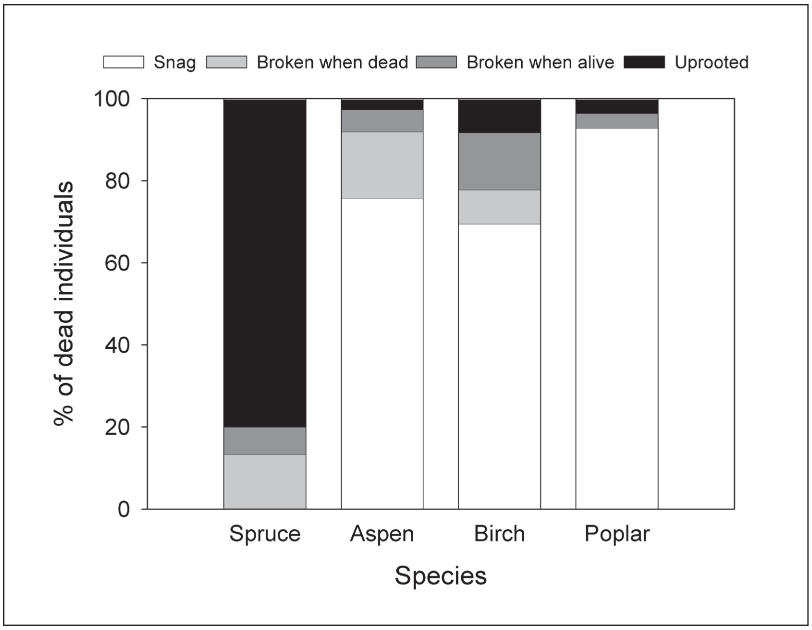

Fig. 2. A comparison of the different forms of mortality for white spruce, trembling aspen, paper birch, and balsam poplar residuals five years after structural retention harvesting.

In the structural retention plots there were large differences between the residual broadleaf species and white spruce in the cause of mortality. The majority of dead aspen (75.7%), birch (69.4%) and poplar (92.7%) were found as standing dead (snags) (Fig. 2). An additional 16.2% of aspen and 8.3% of birch were found broken, either before or after death. Few aspen (2.7%), birch (8.3%) and poplar (3.6%) died due to windthrow. In contrast, 80.0% of the dead spruce had been uprooted due to windthrow.

For spruce, crown class and bole damage were strong predictors of both windthrow and standing mortality. Dominant residual spruce were most likely to die (m = 6.9%), followed by co-dominants (m = 3.3%) and understory residuals (m = 1.5%). If the bole of a residual spruce tree was damaged during the harvesting, it was 2.6 times more likely to die shortly after harvest than an undamaged tree ($P = 0.04$). Additionally, the frequency of mortality increased as the percentage of the circumference of the bole that was damaged increased ($P = 0.04$). Tree boles were damaged on 44.8% of spruce residuals. For trees that died, the mean extent of damage to the bole was 24.0% of the circumference of the tree, while an average of 13.2% of the bole was damaged in surviving spruce residuals.

For poplar, probability of mortality increased with greater slenderness for both U ($P = 0.02$) and CD trees ($P = 0.02$). However, neither crown class or harvesting damage were significant predictors of mortality. For birch, there was a significant relationship between crown class and mortality ($P = 0.04$). Multiple comparisons indicated that U birch had higher mortality than CD birch. Additionally, the mortality of U birch was positively related to slenderness coefficient ($P = 0.04$). For aspen, CD trees were more likely to die than D trees ($P = 0.05$).

Of the trees still alive five years after the partial harvest, 17.8% of aspen, 36.1% of birch, 28.5% of poplar and 5.2% of spruce residuals were categorized as declining, with extensive crown dieback (Fig. 3). An additional 11.8% of aspen, 19.4% of birch, 8.5% of poplar and 8.7% of spruce in the structural retention plots showed evidence of weakening, with low to moderate crown dieback.

Discussion

This study showed high mortality rates of dispersed residual trees following structural retention harvesting (~10% retention) of boreal mixedwood stands. The broadleaf residuals (balsam poplar, paper birch, and trembling aspen) were the most vulnerable species to crown dieback and whole-tree mortality. Annual mortality of these residual species was approximately three to four times greater in the structural retention sites than in reference sites. If retention systems are designed to maintain living biological legacies that facilitate "lifeboating" of species and processes, or to enhance connectivity (Franklin *et al.* 1997), then these high rates of mortality could compromise such objectives, since the living, mature trees are likely to be lost before the regenerating stand will be able to replace their structural characteristics. Conversely, if snag creation is a management priority, our results indicate that retention of isolated hardwood trees is more likely to provide this necessary ecological function (69.4% to 92.7% of dead trees were found standing), rather than conifers (fewer than 20% of dead trees were standing).

Studies of post-harvest mortality of residual trees remain rare, and generally focussed on windthrow. Additionally, the majority of observations have been from the U.S Pacific Northwest and British Columbia (Thorpe and Thomas 2007). To our knowledge, this study is the first in this region to demonstrate elevated mortality rates of retention trees following harvesting. However, there was a much higher rate of mortality in our study than had been found in other partial-cut studies (Beurmeyer and Harrington 2002, Walter and Maguire 2004, Maguire *et al.* 2006). High mortality may be related to two factors: (1) Our study sites are located in a region with relatively low precipitation; therefore, the residual trees could already be stressed prior to harvest. (2) The low level of structural retention created more open conditions than a higher level of retention or other partial-cut systems, such as shelterwood or single-tree selection, and would therefore, be expected to produce a more stressful microclimate.

Our observations of heavy crown dieback among the hardwood species prior to mortality, suggests that the majority of hardwood mortality was probably related to xylem dysfunction (Sperry *et al.* 1994, Maherali *et al.* 2004) from the abrupt increase in evaporative demand that occurs after removing most of the canopy (Bladon *et al.* 2006). Additionally, the more slender poplar (U and CD trees) and birch (U trees) residuals had a greater probability of suffering mortality. This supports the idea that increased wind exposure and bending may damage the xylem of tree stems, reducing hydraulic conductivity and intensifying moisture stress (Fredericksen *et al.* 1994, Liu *et al.* 2003). Bending and damage to water-conducting tissue is potentially a greater problem for more slender stems due to wider oscillations of the crowns than for trees with stout boles (Rudnicki *et al.* 2003).

Residual spruce trees were more susceptible to windthrow than the hardwoods, likely for a couple of reasons: 1) shallower rooting depths (Strong and La Roi 1983) and 2) higher drag coefficients (Rudnicki *et al.* 2004). Additionally, the dominant spruce residuals were more likely to be windthrown than CD or U spruce, which is consistent with others (Ruel *et al.* 2000). However, we did not observe greater windthrow of trees with a high slenderness coefficient, as expected (Coates 1997, Meunier *et al.* 2002, Scott and Mitchell 2005). This may be due to the rapid increase in vulnerability of trees to windthrow when they reach a height of 10 m to 12 m (Ruel 1995, Ruel *et al.* 2003). The average height of the windthrown spruce in the structural retention plots was 22.2 m. Further studies on retention trees that focus on stand factors such as height, root systems, soil properties, age, root rot, crown length, pre-harvest stand density and/or slope position may help to determine the most important factors influencing windthrow risk in these sites.

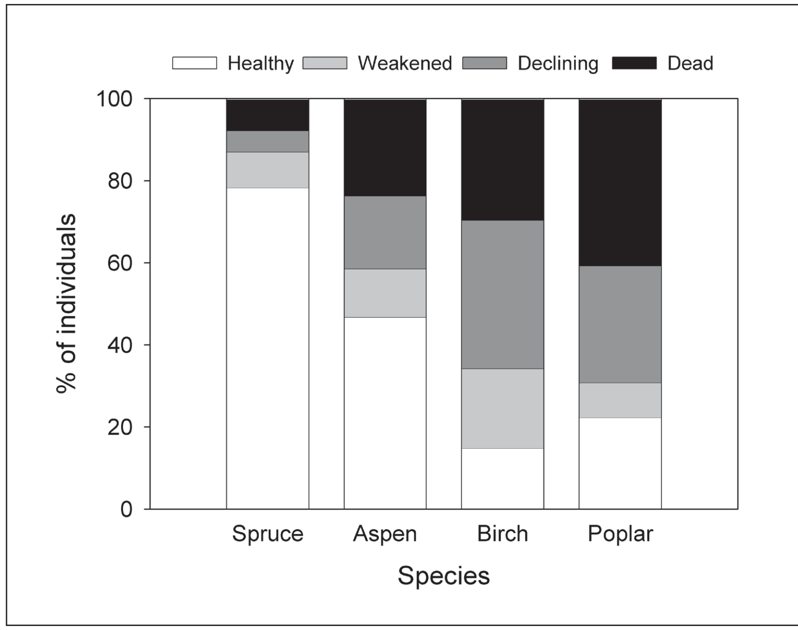

Fig. 3. Health of residual white spruce, trembling aspen, paper birch, and balsam poplar residuals five years after structural retention harvesting.

The likelihood of spruce windthrow was greater for trees that were damaged significantly during the harvesting. This suggests that the force of impact of the harvesting equipment, coupled with damage to structural roots, could potentially compromise tree stability and root anchorage, increasing the probability of windthrow. Also, wounds that expose the cambium or wood could promote additional tree death by increasing the susceptibility to fungal attack, causing stain, decay and reduced vigour (Franklin *et al.* 1987, Nichols *et al.* 1993).

Our observations show that it is important for managers to place some emphasis or forethought on the desired objectives when using the silvicultural approach of tree retention. If the primary objectives are to provide live trees for critical habitat elements and basic ecosystem functions (Franklin *et al.* 2002), then we recommend caution in applying retention systems. High mortality rates could be a problem, especially if retention levels are low, creating more open conditions. Retention of more dominant aspen and birch, more stout poplar, or CD and U spruce residuals may achieve the objectives of providing a living legacy, as these trees appeared less susceptible to mortality. However, if recruitment of snags is the objective, leaving many dispersed hardwood residuals may be a good strategy. Conversely, large spruce trees, particularly those damaged from the logging may be more at risk to windthrow, providing immediate inputs of coarse woody debris. The increased wind speeds in the open cuts appears to be problematic, both for windthrow of the residual conifers and the stresses related to increased evaporative demand—linked to xylem damage in the residual hardwoods (Sperry and Sullivan 1992, Hacke and Sauter 1995). Therefore, managers could use strategies to reduce wind around residual trees such as leaving residuals in clusters, near stand edges, or in sheltered landscape positions.

While this study has provided much new evidence of high mortality rates of residual trees for five years after harvesting, it would be advantageous for land managers to have some idea of how long the mortality rates will remain elevated. Additionally, it would be valuable to determine if increased patch retention or increased levels of isolated retention result in lower rates of mortality. Currently, there is little reliable empirical data comparing the responses of residual trees in dispersed and aggregated retention. It also remains uncertain why some trees remain healthy and persist, while others die prematurely following structural retention harvesting. This quantitative information will be vital for land managers to make informed decisions about sustainability, in terms of the desired retention levels and spatial patterns to achieve various objectives.

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