Fifty years of forest dynamics following diameter-limit cuttings in balsam fir – yellow birch stands of the Lower St. Lawrence region, Quebec[1]

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Abstract: The long-term effects of high-intensity diameter-limit cuttings conducted in the winter and summer of the 1940s and 1950s on the dynamics of softwood and mixedwood stands in southeastern Quebec were compared. Changes in composition and stand structure over a 50 year period were studied using 18 permanent sample plots located in the Lac-Métis Seigneury observation area measured in 1950, 1960, 1970, and 2003. Winter logging operations were conducted between 1942 and 1949, and summer logging operations were conducted between 1958 and 1960. The interaction between cuttings and the 1950s and 1970s spruce budworm (*Choristoneura fumiferana* (Clemens)) outbreaks that occurred in the area affected forest dynamics. For the two logging operation types, the most abundant softwood species observed before logging and in 2003 was balsam fir (*Abies balsamea* (L.) Mill.), whereas white birch (*Betula papyrifera* Marsh.) and yellow birch (*Betula alleghaniensis* Britt.) were the most abundant hardwood species. Changes in the overstory composition were more significant in the summer logging operations than in the winter ones. The softwood cover type observed before winter logging was maintained in 2003. Following summer logging, an important increase in the proportion of birch species was observed. The softwood cover type observed before logging had changed to a mixedwood cover type by 2003. Forest dynamics differences between the two types of logging were the result of interactions between the density and composition of advance regeneration, the microsite conditions after logging, and the length and severity of spruce budworm outbreaks.

Résumé : Les effets à long terme de coupes à diamètre limite de forte intensité réalisées en hiver et en été dans les années 40 et 50 sur la dynamique forestière de peuplements résineux et mixtes du sud-est du Québec ont été comparés. Les changements de composition et de structure des peuplements sur une période de 50 ans ont été étudiés à l'aide de 18 parcelles-échantillons permanentes situées dans l'aire d'observation de la Seigneurie du Lac-Métis ayant fait l'objet de mesures répétées en 1950, 1960, 1970 et 2003. Deux types de coupe ont été considérés : (*i*) des coupes d'hiver réalisées entre 1942 et 1949 et (*ii*) des coupes d'été réalisées entre 1958 et 1960. L'interaction entre les coupes et les épidémies de tordeuse des bourgeons de l'épinette (C*horistoneura fumiferana* (Clemens)) survenues dans les années 50 et 70 a affecté la dynamique forestière. Pour les deux types de récolte, l'espèce résineuse la plus abondante observée avant la coupe et en 2003 était le sapin baumier (*Abies balsamea* (L.) Mill.), alors que le bouleau blanc (*Betula papyrifera* Marsh.) et le bouleau jaune (*Betula alleghaniensis* Britt.) étaient les espèces feuillues les plus abondantes. Les changements observés dans la composition du couvert arborescent ont été plus importants à la suite des coupes d'été qu'à la suite des coupes d'hiver. Le couvert forestier de type résineux observé avant les coupes d'hiver a été maintenu en 2003. À la suite des coupes d'été, une augmentation notable de la proportion des bouleaux a été observée. Le type de couvert est passé de résineux avant la coupe à mélangé à prédominance résineuse en 2003. Les différences de dynamique forestière selon les deux types de coupe ont été le résultat des interactions entre la densité et la composition de la régénération préétablie, les caractéristiques des micro-sites après coupe et la longueur et l'intensité des épidémies de la tordeuse des bourgeons de l'épinette.

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Introduction

The eastern balsam fir (*Abies balsamea* (L.) Mill.) – yellow birch (*Betula alleghaniensis* Britt.) forest types occupy a large portion of the northeastern North American continent. Until the early 1990s, these forest types were managed mostly by clear-cutting and diameter-limit cutting in southern Quebec. Diameter-limit cutting has also been commonly applied in various mixed forest types in North America (Kenefic et al. 2005; Nyland 2005). It was observed that both cutting methods can have negative or undesirable effects for several mixed-forest types (Harvey and Bergeron 1989; Sendak et al. 2003; Angers et al. 2005), including mixed stands of balsam fir and yellow birch (Archambault et al. 1998, 2003; De Grandpré et al. 2000). In particular, diameter-limit cutting can engender huge expanses of degraded stands with small trees of inferior quality in several mixed-forest types (Kenefic et al. 2005; Blum and Filip 1963) or severely modify the proportion of one of the dominant species (Archambault et al. 2003; Sendak et al. 2003; Angers et al. 2005). Clear-cutting can change the long-term species dynamics or reduce species diversity (Hix and Barnes 1984; Meier et al. 1995; Archambault et al. 1998). Furthermore, aggressive competing vegetation can invade cutovers and decrease site productivity for several decades (Archambault et al. 1998; Laflèche et al. 2000).

The dynamics of natural and anthropogenic disturbances in mixed ecosystems is a complex issue. There is still much uncertainty surrounding the usefulness of the knowledge gathered over time to guide management practices. For instance, the extent to which silvicultural treatments, such as diameter-limit cutting, can emulate the gap dynamics of mixed-forest ecosystems or be applied while minimizing the long-term negative or undesirable effects remains largely unknown. There is a need to examine the application of such treatments under different operational conditions using historical datasets obtained from repeated measurements. For instance, logging activities conducted when there is snow cover or during the dormant season can have different long-term effects on species dynamics compared with logging during summer conditions (Tubbs and Reid 1984; Stone and Elioff 2000; Stone 2002). This information is essential to improve the information that forest managers can use for developing appropriate forest management scenarios and silvicultural techniques that can be incorporated into a suitable ecosystem management approach (Bergeron and Harvey 1997; Hunter 1999; Kimmins 2002). It is also essential that silvicultural techniques be based on a better understanding of natural processes to preserve biodiversity, ensure successful regeneration of desired species, and prevent competing species invasion.

The objective of this study was to determine if the application of high-intensity diameter-limit cuttings significantly changed the species composition in balsam fir – yellow birch forest ecosystems. Postharvest vegetation dynamics were studied in the context of two types of harvesting operations: (*i*) diameter-limit cuttings conducted in winter between 1942 and 1949 and (*ii*) diameter-limit cuttings conducted in summer between 1958 and 1960. The effects of the two types of diameter-limit cuttings (winter and summer) on stand composition and structure were compared. The interaction of spruce budworm outbreaks with diameter-limit cuttings was studied. The study emphasized development pathways within each type of harvesting operation to provide insight into the vegetation dynamics of these complex and poorly understood ecosystems.

Methods

Study area

The study was conducted in the area southeast of Rimouski (48°23′N, 67°55′W) in eastern Quebec, in the Bas-Saint-Laurent Model Forest (Lac-Métis Seigneury). This area belongs to forest section L.6 (Témiscouata-Restigouche), of the Great Lakes – St. Lawrence forest region identified by Rowe (1972). The mixedwood forests found in this area are largely dominated by balsam fir. Yellow birch, white spruce (*Picea glauca* (Moench) Voss), and white birch (*Betula papyrifera* Marsh.) can also occupy a large portion of the stands but in smaller proportions than balsam fir. Other species that can be found in relatively small proportions include red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marsh.), black ash (*Fraxinus nigra* Marsh.), and eastern white cedar (*Thuja occidentalis* L.). Volume at stand maturity can be as high as 200 m^3/ha. Spruce budworm (C*horistoneura fumiferana* (Clemens)) outbreaks (Blais 1983; Pominville et al. 1999), windthrow (Lorimer 1977), and fire are the most frequent natural disturbances that affect the dynamics of this forest type (Grondin 1996; Prévost et al. 2003). While fire-related disturbances may occasionally occur over large areas, spruce budworm and windthrow generally create canopy openings that generate gap dynamics conditions, which characterize the dynamics of temperate uneven-aged northern hardwood forest types (De Grandpré et al. 2000). One of the main benefits of gap dynamics is to maintain high species diversity (Clebsch and Busing 1989; Houle 1994; Brokaw and Busing 2000).

The moderately rugged relief consists of rounded hills with broad summits and gentle to moderately steep slopes. The mean altitude is 365 m. The bedrock consists of sedimentary rocks (limestone, shale, sandstone, conglomerate, and pyroclastic rocks). Surficial deposits vary in thickness and consist primarily of weathering products of bedrock and tills. The mean annual temperature is 2.5 °C, growing degree-days range from 2000 to 2200 (base temperature of 5.6 °C), and the growing season lasts between 150 and 160 days. Mean annual precipitation varies between 900 and 1100 mm (Robitaille and Saucier 1998). The most common potential climax vegetation on mesic sites consists of balsam fir – yellow birch ecosystems. Stands located at altitudes higher than 600 m are colonized mostly by balsam fir – white birch ecosystems.

The study area experienced two major spruce budworm outbreaks in the second half of the 20th century (Table 1). The first one occurred in the 1950s (Webb et al. 1961; Hatcher 1963), and the second one occurred in the 1970s (Hardy et al. 1986). Four years of light defoliation (1%–34% defoliation) and 5 years of moderate to severe defoliation (35%–100% defoliation) were recorded during the first outbreak between 1948 and 1957. Thirteen years of light defoliation and 6 years of moderate to severe defoliation were recorded during the second outbreak between 1974 and 1990. The years and intensities of spruce budworm annual

Table 1. Summary of sample plot characteristics.

Characteristic	Winter logging operations (1942–1949) (n = 10)[a]	Summer logging operations (1958–1960) (n = 8)
Altitude (m)	318±13[b]	376±6
Surficial deposit[c]	1A (9), 1AY (1)[d]	1A (7), 1AY (1)
Aspect[e]	N (5), S(3), W (3), FL (1)	S (1), E (2), W (5)
Slope position[f]	LS (6), MS (3), US (1)	MS (7), US (1)
Slope gradient (%)	13±3	12±4
Thickness of humus (cm)	10±2	14±5
Thickness of the A and B horizons (cm)	42±4	49±3
Texture of the C horizon[g]	LS (3), SL (2), SCL (3), SiL (2)	S (1), LS (3), SL (3), SiL (1)
Drainage[h]	1 (1), 2 (5), 3 (4)	2 (7), 3(1)
Ecotype[i]	MS11 (2), MS12 (4), MS21 (1), MS22 (3)	MS11 (4), MS12 (1), MS22 (2), RS52 (1)
Spruce budworm outbreak years	1948–1957, 1974–1990	1948–1957, 1974–1990
Sampling years	1950, 1960, 1970, 2003	1950, 1960, 1970, 2003

[a] Number of plots.

[b] Mean ± SE.

[c] 1A, undifferentiated till > 1 m thick; 1AY, undifferentiated till 50 cm to 1 m thick.

[d] The numbers in parentheses indicate the number of plots that fall into this category.

[e] N, north; S, south; E, east; W, west; FL, flat.

[f] LS, lower slope; MS, midslope; US, upper slope.

[g] S, sand; LS, loamy sand; SL, sandy loam; SCL, sandy clay loam; Silt, silt loam. Results of field tests.

[h] 1, Rapid; 2, well-drained; 3, moderately well-drained.

[i] MS11, balsam fir – yellow birch stands on thin to thick deposits, with a coarse texture and xeric or mesic drainage; MS12, balsam fir – yellow birch stands on thin to thick deposits, with medium texture and mesic drainage; MS21, balsam fir – white birch stands on thin to thick deposits, with coarse texture and xeric or mesic drainage; MS22, balsam fir – white birch stands on thin to thick deposits, with medium texture and mesic drainage; RS52, balsam fir – red spruce stands on thin to thick deposits, with medium texture and mesic drainage (Blouin and Berger 2002).

defoliation were obtained from aerial surveys conducted by the ministère des Forêts between 1948 and 1957 and the ministère des Ressources naturelles et de la Faune du Québec (MRNFQ) between 1974 and 1990.

In 1947, the Department of Northern Affairs and National Resources of Canada initiated a new long-term research project to study the postharvest evolution of forest stands located in different regions of Quebec (Pfalzgraf 1970). To this end, 15 observation areas were established, each covering approximately 13 km^2 and comprising some 250 permanent sample plots. Most of the plots were measured at 10 year intervals, with the last inventory being completed in the late 1960s. This study uses inventory data from one of these observation areas, the Lac-Métis Seigneury observation area, which was established in 1950 within the current territory of the Bas-Saint-Laurent Model Forest (Boynton 1954).

Data collection

For the present study, 18 square sample plots (20.1 m × 20.1 m) from the Lac-Métis Seigneury observation area were examined. The plots were established in 1950 and remeasured in 1960 and 1970. The original study plan included the analysis of more sample plots. However, the destruction of sample plots during recent harvest operations limited the availability of suitable sample plots that could be remeasured. Using historical documents from the Price Brothers Company and MRNFQ ecoforest maps, only plots that had not previously been logged before the diameter-limit cuttings were selected. The plots were selected such that the effects of permanent site characteristics (surficial deposit, topography, and pedology) on vegetation dynamics were minimized. Thus, most of the plots selected were located on mesic sites with medium soil texture, at midslope or on lower slopes, on thick undifferentiated tills (Table 1). The plots were stratified on the basis of the two types of logging operations: 10 plots in the winter cutovers from 1942 to 1949 and 8 plots in the summer cutovers from 1958 to 1960. The results of the 1950, 1960, and 1970 inventories were compared with the last inventory that was conducted in the 18 selected plots in 2003.

The preharvest forest cover composition of the stands that were winter logged between 1942 and 1949 is not known with certainty because the permanent plots were established after the harvesting operations. However, according to the establishment report of the Lac-Métis Seigneury observation area (Boynton 1954), 9 of the 10 sample plots selected for the study corresponded to the fir–spruce–birch cover type, and one sample plot corresponded to the fir–spruce type. The Lac-Métis Seigneury observation area included 46 sample plots corresponding to the fir–spruce–birch forest type that had undergone winter logging before the experimental site was established. The preharvest forest cover of these 46 sample plots was composed of 48% balsam fir (in terms of volume), 32% spruces, 3% eastern white cedar, 16% birch, and 1% other deciduous species. The basal area data before logging for each sample plot were not available. Therefore, we hypothesized that the preharvest forest cover of the 10 plots selected to study the effects of the winter operations from 1942 to 1949 was similar to that of the 46 sample plots characterized by fir–spruce–birch forest cover. The preharvest forest cover was likely coniferous with some yellow birch intermixed with white birch. In addition, the preharvest composition of the sample plots of fir–spruce–birch was similar to the composition of plots with this same forest cover that were not logged in winter between 1942 and 1949 (Boynton 1954). Therefore, the preharvest composition of the sample plots selected for the investigation of vegetation dynamics following winter logging operations should resemble the preharvest composition of the eight sample plots selected to study vegetation dynamics after the summer logging operations in 1958–1960.

Winter logging operations from 1942 to 1949 aimed at harvesting sawlogs (Boyton 1954). The operations were conducted with bow and bucking saws, beginning in early fall and continuing through the winter. Horses were used to skid the timber to the stacking area, and then horses, tractors, or tracked skidders were used to carry the wood to the main road. Afterward, the wood was transported to the lake on sleds pulled by trucks on icy roads. The lake was used for log drives. The lower diameter limit was set at 16 cm for balsam fir and 18 cm for spruces. The summer logging operations carried out from 1958 to 1960 also had the goal of harvesting for sawlogs. Harvesting was conducted with chain saws, and the operations continued until the first snowfalls. Horses were used to skid the wood before the snow came. It was then hauled to the lake by truck on roads built by bulldozers. The lower diameter limit was set at 16 cm for balsam fir and for spruces.

For both types of logging operations, the proportion of timber removed was important (Boyton 1954). About 59% of standing volume was harvested during the winter operations (1942–1949) and 58% during the summer operations (1958–1960). During the 1942–1949 operations, approximately 83%, 58%, 29%, and 23% of volume was harvested for spruce, balsam fir, eastern white cedar, and birch, respectively. In the 1958–1960 logging operations, the corresponding percentages were approximately 92%, 59%, 6%, and 7%, respectively. Information on the harvesting intensity for winter logging (1942–1949) was obtained from 46 adjacent sample plots in stands that were also winter logged and described by Boynton (1954) as fir–spruce–birch forest type. However, the exact harvesting intensity for the 10 sample plots examined in the present study could not be determined because the data from the stump inventory conducted when the experimental site was established in 1950 were no longer available. Information on the harvesting intensity from the summer logging (1958–1960) was determined based on a stump inventory conducted during the 1960 measurements.

During the 2003 inventory, a detailed ecological survey was undertaken. Physiographic and pedological characteristics were recorded using the procedure applied by the MRNFQ for the description of forest ecosystems (Saucier 1994). Diameter at breast height (DBH), number of live merchantable and nonmerchantable stems (DBH ≥ 1 cm), and number of dead merchantable and nonmerchantable stems (DBH ≥ 5 cm) present on the different plots were recorded for each species. Using the MRNFQ classification system (Létourneau et al. 2003), a cover type was classified as softwood if conifer species occupied more than 75% of the stand basal area. A stand was considered as mixedwood when there was between 25% and 75% of conifers in the basal area. If less than 25% of the basal area was in conifers, a stand was classified as hardwood.

Tree height and age were measured for dominant and codominant stems located within the plots during the 2003 inventory. In total, 80 stems were sampled to determine height, and increment cores were taken at a 1 m height from 79 stems for age determination. Tree age was determined in the laboratory by counting growth rings under a microscope. Annual growth ring widths of increment cores were measured using the automatic tree ring measurement system MacDENDRO$^{\mathrm{TM}}$ (Guay et al. 1992). Growth curves at 1 m based on annual radial increments were constructed for balsam fir, white spruce, and red spruce.

Using the methodology developed by Blais (1964), radial growth losses at breast height caused by spruce budworm defoliation were estimated for balsam fir, white spruce, and red spruce, three highly susceptible species (MacLean 1980; Coulson and Witter 1984). Radial growth losses were determined by comparing the average annual radial growth of the increment cores for the 10 years preceding the outbreaks with the average for the suppression periods, which were determined by visual analysis of the growth curves of the three species.

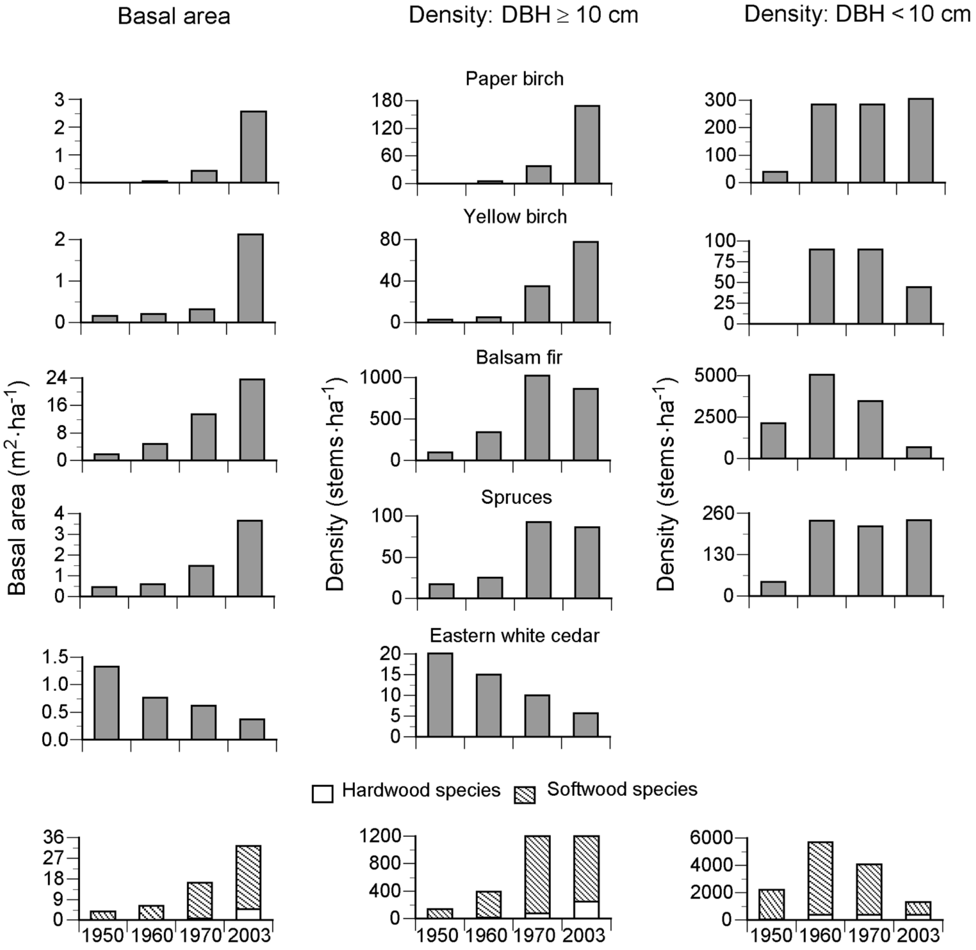
Data analysis

In each plot, the number of stems per hectare of commercial-size trees (DBH ≥ 10 cm) and saplings (DBH < 10 cm) and the basal area of commercial-size trees were computed for each commercial species, all commercial softwood and hardwood species, and all commercial species. A value of zero was assigned to these variables in plots from which the species or category of species was missing. Since the accuracy of identification of the different spruce species (*Picea* spp.) was uncertain for the inventories conducted before 2003, the data for the different spruce species were grouped. Mean values of the three variables and their standard errors were obtained for the 1942–1949 and 1958–1960 cuttings after weighting each plot by the plot area. This was necessary because plot areas, as measured in 2003, varied from 353 to 485 m^2 (mean = 403 m^2). Mean DBH for each logging type was obtained as weighted averages of the means per plot, the weight being the number of trees of the species or category of species in the plot. Standard errors were weighted accordingly.

Mean differences in basal area and stem density were computed from differences between the 1970 and 2003 inventories for both types of harvesting operations, from differences between the 1950 and 1970 inventories, and from those between the 1950 and 2003 inventories for the 1942–1949 winter cuttings, and from differences between the 1960 and 1970 and between the 1960 and 2003 inventories for the 1958–1960 summer cuttings. These differences are the mean differences per plot, weighted by the plot area. Standard errors of mean differences were weighted accordingly.

Hypotheses that the mean differences in basal area and stem density between the two inventories were zero were tested with a paired t test. Considering the large number of null values for individual species, these tests were restricted to the three categories of species described above: commercial softwoods, commercial hardwoods, and all commercial species. Because many tests were performed on potentially correlated variables, a difference was declared significant at the $\alpha = 0.05$ level only when its observed P value was smaller than $0.05/K$, where K is the number of tests performed. This is a generalized version of the Bonferroni multiple comparison procedure. It errs on the conservative side in that it prevents us from declaring a significant difference more often than necessary. A total of 27 tests were per-

Fig. 1. Changes in selected mean stand characteristics for the most abundant commercial species following winter logging operations (1942–1949).



formed: three species categories × 3 variables × 3 pairs of inventories within each cut, so that $K = 27$. The three variables were the basal area of commercial-size (DBH ≥ 10 cm) species, the density of commercial-size trees, and the density of saplings. Therefore, a P value smaller than $0.05/27 = 0.0019$ was required for a test to be declared significant at the $\alpha = 0.05$ level.

Comparisons between the two types of harvesting operations could not be made using t tests because sample plots were not assigned at random within the study area. Therefore, comparisons between the two groups of plots are qualitative rather than based on statistical comparisons.

Results

Composition and stand structure

Following winter logging operations, there was no major change in species composition. Despite the substantial increase in basal area for birches from 1970 to 2003, the proportions in basal area for hardwoods (15%) and softwoods (85%) in 2003 (Fig. 1) were comparable with the volume proportions of hardwoods (17%) and softwoods (83%) that existed before the logging operations. The importance of advance softwood regeneration (2174 saplings/ha) observed after logging (1950) probably contributed to maintaining the high softwood composition and could have prevented the sites from being invaded by intolerant hardwood species. Balsam fir, eastern white cedar, spruces, and yellow birch were the most abundant species in 2003. Whereas balsam fir and spruces increased gradually in importance from 1950 to 2003, basal area for white and yellow birches did not change very much from 1950 to 1970, but increased sharply from 1970 to 2003. Basal area for eastern white cedar decreased from 1950 to 1970. The relative abundance of some species in 2003 differed from the relative abundance in 1950. In 2003, balsam fir, yellow birch, and eastern white cedar accounted for 73%, 6%, and 1.1% of total basal area, respectively, which represented about a 1.5-fold increase for balsam fir and yellow birch relative to 1950, but a 35-fold decrease for eastern white cedar. The relative proportion of white birch and spruces remained unchanged with 11% and 12% of the total basal area in 2003, respectively, and 11% for both species in 1950. In 2003, two spruce species were tallied: white spruce and red spruce (*Picea rubens* Sarg.) (69% and 31% of the basal area of spruce, respectively). Eastern white cedar, a late-successional species (Bergeron and Dubuc 1989), was scarce in 2003 relative to the other species. Red maple and black ash were observed only in

Table 2. Mean diameter (cm) of merchantable stems of commercial species (DBH ≥ 10 cm).

	Winter logging operations (1942–1949)				Summer logging operations (1958–1960)			
Species	1950 (*n* = 10)	1960 (*n* = 10)	1970 (*n* = 10)	2003 (*n* = 10)	1950 (*n* = 8)	1960 (*n* = 8)	1970 (*n* = 8)	2003 (*n* = 8)
Red maple	—	—	—	12.0	17.0	22.0	16.4	15.3
Sugar maple	—	—	—	—	—	10.0	11.0	29.0
White birch		11.0	11.3	13.5	19.9	16.9	17.3	15.1
Yellow birch	28.0	19.7	10.9	18.6	26.1	26.7	24.3	21.6
Black ash	—	—	—		10.0	12.7	—	—
Trembling aspen	—	—	—	—	—	—	—	27.5
Hardwood species	28.0	16.8	11.1	14.0	21.6	17.7	19.4	17.1
Balsam fir	15.9	13.5	12.8	19.0	18.2	12.9	14.4	18.2
Spruce species	17.1	16.4	14.1	22.6	25.9	11.7	15.9	20.2
Eastern white cedar	24.0	23.2	27.8	29.0	26.0	32.5	26.5	42.0
Softwood species	17.2	14.6	13.3	19.4	19.6	13.4	15.0	18.9
All species	18.4	14.8	13.0	18.0	19.6	14.9	15.5	18.2

Table 3. Dendrometric characteristics of trees studied (dominant and codominant commercial species) based on the 2003 inventory.

	Winter logging operations (1942–1949)			Summer logging operations (1958–1960)		
Species	DBH (cm)	Height (m)	Age (years)	DBH (cm)	Height (m)	Age (years)
Red maple	—	—	—	20.4±20.0	13.6	45
Sugar maple	—	—	—	26.1	—	—
White birch	19.1±13.0	17.5±1.7	52±6	17.0±29.0	—	—
Yellow birch	23.3±14	17.0±1.9	50±1	34.6±44.0	—	—
Trembling aspen	—	—	—	27.5	—	—
Balsam fir	22.4±4.0	17.4±0.4	54±2	24.2±7.0	15.6±0.5	60±3
White spruce	26.9±15.0	19.6±0.5	43±9	27.5±32.0	16.3±0.9	90±15
Red spruce	22.1±22.0	17.6±0.8	74±14	24.1±21.0	16.4±0.6	47±4
Eastern white cedar	—	—	—	46.7±79.0	—	—

Note: Values are means ± SEs. The number of stems per species ranged from 1 to 148 (total = 298) for diameter, from 1 to 28 (total = 80) for height, and from 1 to 27 (total = 79) for age.

2003. Red maple basal area accounted for less than 1% of the total basal area, and there were no merchantable stems of black ash.

From 1950 to 1970, the basal area and density of merchantable stems of white birch and yellow birch were low but increased notably from 1970 to 2003 (Fig. 1). From 1950 to 2003, the density of merchantable stems of balsam fir and spruces exhibited a fairly steady upward trend. However, the density of merchantable stems of these two softwood species decreased between 1970 and 2003. The density of merchantable softwoods and of all commercial species increased significantly between 1950 and 1970 ($0.000\,6 \le P \le 0.001\,06$) and between 1950 and 2003 ($0.000\,25 \le P \le 0.001\,61$).

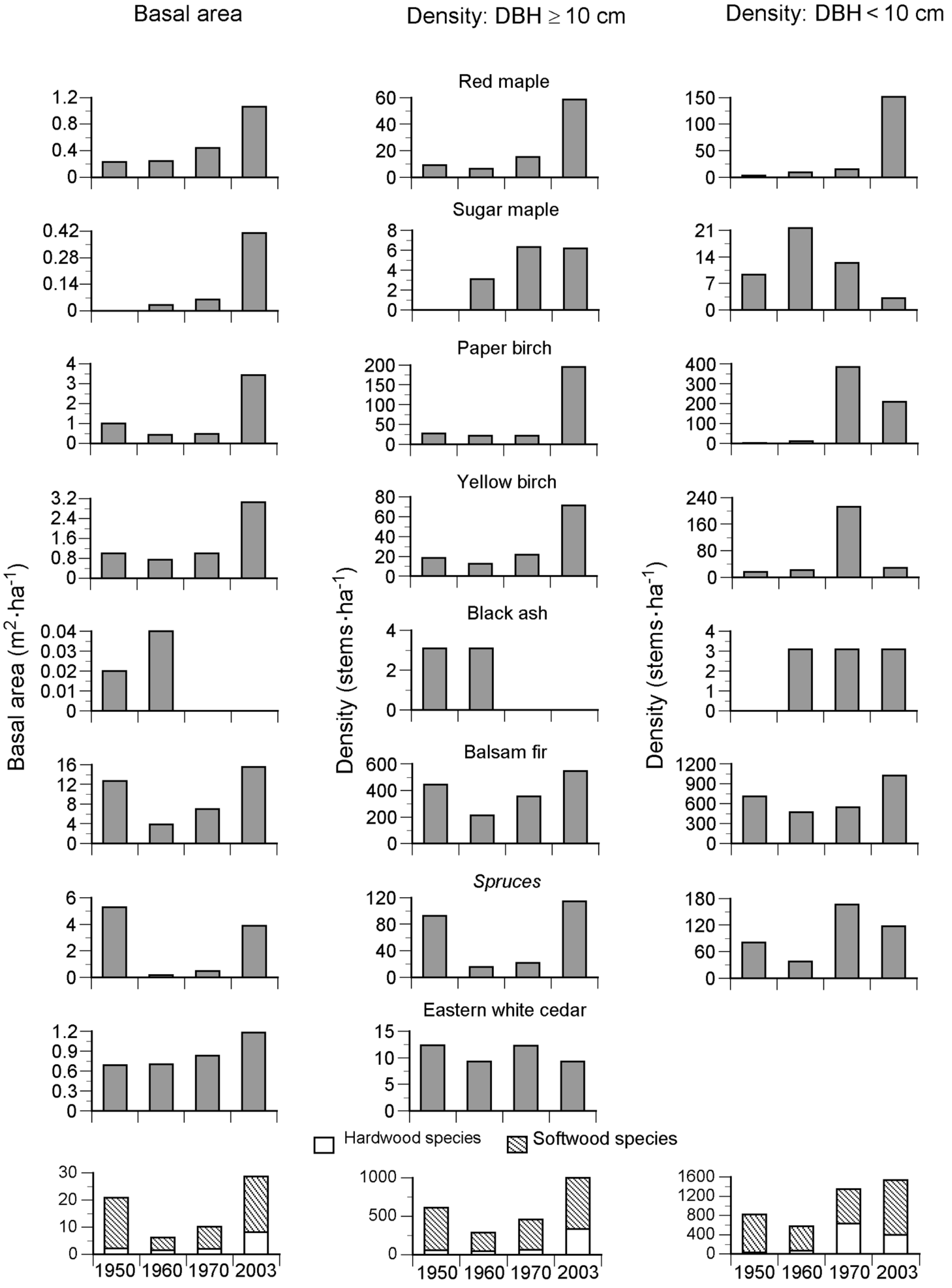
The mean diameter of merchantable balsam fir and spruces declined between 1950 and 1970 and then rose in 2003 (Table 2). The decreases in mean diameter may be attributed to massive recruitment (the point at which saplings become merchantable stems) of the regeneration that was released during the cuttings. Because most balsam fir were 1 m tall in the early 1950s, it can be hypothesized that recruitment occurred mainly during the 1960s and 1970s. Assuming an annual increment of about 5–10 mm/year, DBH would increase from 0 to 10 cm over a period of 10–20 years.

Dominant and codominant white spruce and yellow birch had the largest DBH (Table 3). Most species regenerated when the cut was undertaken. The establishment of red spruce took place approximately 20 years before logging. Therefore, most of the stems that were dominant and codominant in 2003 had reached 1 m in height in the first two decades after harvesting, except for red spruce, which probably reached this height before the intervention.

Following summer logging operations, there was a substantial increase in the proportion of basal area for hardwood species, which changed from 11% before harvesting to 29% in 2003 (Fig. 2). This was a pronounced difference compared with winter logging operations. The change in the relative proportions of hardwoods and softwoods before and after harvesting was less pronounced during winter logging. As a consequence, the forest cover changed from a softwood type before logging to a conifer-dominated mixedwood cover type afterwards. Fifty years after summer logging, the sites that had been logged experienced a substantial increase in white birch and yellow birch abundance. On the other hand, the invasion of the cutover sites by these two hardwood species was less pronounced following winter logging operations.

Both in 1960 and 2003, balsam fir, spruces, white birch, and yellow birch were the most abundant species. However, white birch and yellow birch were more abundant in 2003 with 23% of the total basal area compared with 10% before logging. Maples were also more abundant with 5% of the total basal area in 2003 compared with 1% before logging. In

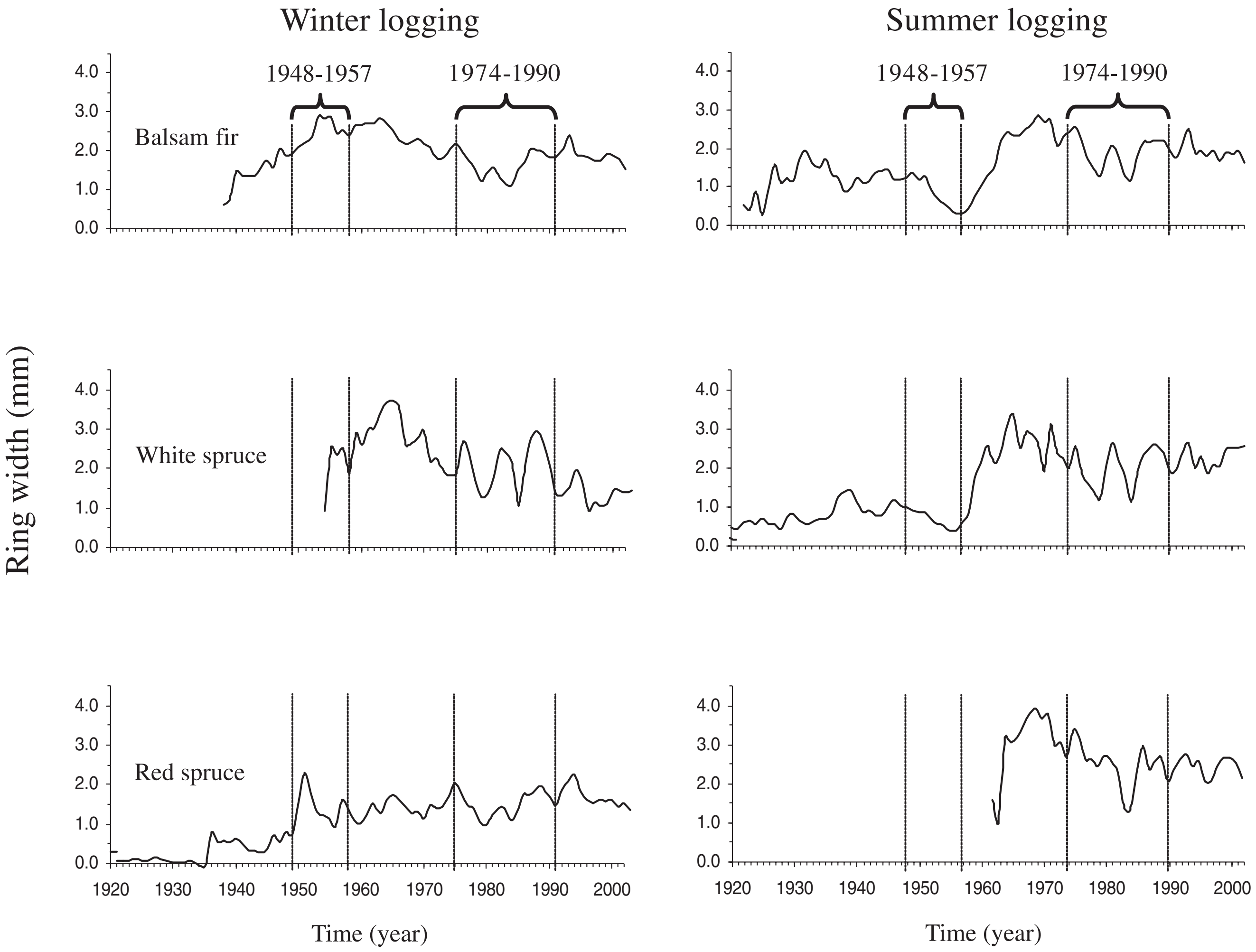
Fig. 2. Changes in selected mean stand characteristics for the most abundant commercial species before harvesting (1950 data) and following summer logging operations (1960, 1970, and 2003 data).



2003, red spruce and white spruce accounted for 63% and 37%, respectively, of the basal area for spruces. Sugar maple was observed from 1960 to 2003, with a basal area of less than 2% of the total basal area. Black ash was observed in 1950 and 1960 only, with a basal area of less than 1% of the total basal area at all times. The basal area of trembling aspen (*Populus tremuloides* Michx.) represented less than 1% of the total basal area in 2003, the only year it was observed.

The basal area and density of merchantable stems for red maple, white birch, and yellow birch were fairly stable from 1960 to 1970 but increased sharply between 1970 and 2003 (Fig. 2). The basal area and density of merchantable stems for balsam fir and spruces increased markedly between 1960 and 2003. Between 1960 and 2003, the basal area for both softwood and hardwood species and of all commercial species, along with the density of merchantable stems of commercial species, increased significantly ($0.000\,1 \leq P \leq 0.001\,76$) (Fig. 2). From 1970 to 2003, the basal area of softwood species and of all commercial species likewise increased significantly ($P = 0.001\,73$ and $P = 0.000\,17$, respectively). The

Fig. 3. Mean annual radial growth curves at breast height (1.3 m) for balsam fir, white spruce, and red spruce for both winter and summer logging operations. Vertical broken lines enclose the periods of spruce budworm outbreaks.



mean diameter of balsam fir and spruces increased steadily between 1960 and 2003.

Dominant and codominant white spruce and yellow birch had the largest DBH in 2003 (Table 3). Most of the dominant and codominant stems of balsam fir, white spruce, and red spruce that were present in 2003 had reached 1 m in height before the logging operations.

In 1970, i.e., about 25 years after the winter logging operations and 10 years after the summer ones, there were 872 stems/ha of mountain maple (*Acer spicatum* Lamb.) in stands that had originated from winter logging and 988 stems/ha in stands that had originated from summer logging. In 2003, the density of mountain maple saplings was 1354 stems/ha in stands that originated from winter logging and 1661 stems/ha in those that resulted from summer logging. It appears that growth in commercial species has not been reduced as it can occur following following mechanized clearcuts. In these forest types, mountain maple can completely colonize the cutovers and cause fierce competition for 30–60 years (Bell 1991; Jobidon 1995; Archambault et al. 1998).

Spruce budworm outbreak interaction

Spruce budworm has the capacity to defoliate and kill balsam fir and spruce trees over large areas (MacLean 1980; Coulson and Witter 1984). The basal area and density of merchantable balsam fir and spruces increased appreciably between 1950 and 1960 (Fig. 1) in the sample plots logged during winter (1942–1949). Therefore, it appears that the spruce budworm outbreak that occurred in the area in the 1950s had relatively limited effects in these sample plots. Although some radial growth suppression could be observed between 1956 and 1958 for balsam fir and between 1951 and 1959 for red spruce, there was no clear pattern of radial growth suppression during this outbreak (Fig. 3). Spruce-budworm-induced defoliation and mortality were likely limited because of the young age of the stems at the time of the outbreak. In general, young stands are less vulnerable to defoliation by this insect than mature or overmature stands (MacLean 1980). However, the impact of the 1970s spruce budworm outbreak appears to have been greater than the 1950s outbreak. Between 1976 and 1992, there was a period of growth reduction (Fig. 3), and the mean annual radial growth at breast height was reduced by 22% for balsam fir and 16% for white spruce as compared with the 10 years before the growth reduction period (1966–1975). Moreover, a high rate of mortality was observed in standing balsam fir and white spruce in 2003. This mortality, which can reasonably be attributed to the budworm outbreak of the 1970s, accounted for 10% of the total basal area (standing dead plus live) of white spruce and 9% of the total basal area of balsam fir. In addition, in some sample plots, a number of fallen dead balsam fir trees that had probably been attacked by spruce budworm were still recognizable on the ground. However, it should be noted that assumptions about mortality caused by the spruce budworm cannot be verified because no mortality survey was available for the sample plots.

The situation was different in the sample plots logged during summer (1958–1960). The spruce budworm outbreak in the 1950s had a clear impact on balsam fir and white spruce before the 1958–1960 logging operations (Fig. 3). These stands were more affected by spruce budworm defoliation than stands harvested in winter logging operations, which were probably too young to be defoliated at the time of the outbreak. Between 1952 and 1958, the period of growth reduction before harvesting, the mean annual radial growth of balsam fir and white spruce was reduced by 59% and 37%, respectively, as compared with the 10 years before this period (1942–1951). Because of the severity of the 1950s outbreak, an important number of balsam fir and spruce saplings could have been killed, resulting in the low density of advance softwood regeneration (790 stems/ha) observed before logging (1950). The advance regeneration decreased further to 506 stems/ha after logging (1960). Basal area and merchantable stem density of balsam fir and spruces steadily increased between 1970 and 2003, even though the spruce budworm outbreak of the 1970s seriously affected the area, as it did in stands harvested during the winter logging operations. Between 1976 and 1992, the mean annual radial growth at breast height was reduced by 28% for balsam fir, 21% for white spruce, and 27% for red spruce as compared with the 10 years before the growth reduction period (1966–1975). However, mortality observed in 2003 was lower than mortality observed in plots used to assess the impact of the winter logging operations of 1942–1949. There was no mortality among spruces and balsam fir mortality was only 4%.

Discussion

Even though data on the degree and extent to which soil disturbances occurred following harvesting were not available, it is generally recognized that summer logging generally disturbs the soil surface more than winter logging (Webber et al. 1969; Harvey and Bergeron 1989). Therefore, white birch and yellow birch, which regenerate better on disturbed soil or decomposing wood (Erdmann 1990; Bell 1991), could have been favoured in summer cutovers. The increase in the proportion of hardwood species may also be explained by brighter conditions on the soil surface because of the low density of advance softwood regeneration (saplings) after logging (1960), which accounted for only 506 stems/ha. Better light conditions favour the establishment and development of intolerant hardwood species. Interactions between these two hypotheses could also explain the increase in the proportion of hardwood species. The increase in the proportion of hardwood species relative to softwood species that occurs following mechanized clearcuts in softwood stands (Frisque et al. 1978; Ruel 1992) and mixedwood stands (Archambault et al. 1998; Laflèche et al. 2000) is a well-known problem. The findings of this study confirm that this situation can even occur in a context of high-intensity diameter-limit cuttings conducted in the summer, where horse skidding was used instead of mechanical skidders. Several studies have indicated that summer logging usually causes more damage to advance regeneration than winter logging (Tubbs and Reid 1984; Bella 1986; Stone and Elioff 2000; Stone 2002).

Balsam fir – yellow birch forest ecosystems are very productive stands, but they are difficult to manage because of their multispecies composition. To minimize potential problems, such as an undesirable change in forest cover composition or invasion by competing species, it is important to keep developing new silvicultural practices that will ensure ecosystem sustainability and minimize the negative impacts of logging operations. Shelterwood cutting, group-selection cutting, seed-tree cutting, and cutting with protection of advance regeneration and soils are among the silvicultural approaches to be considered. Research efforts should be devoted to determining the optimal stand openings that will prevent invasion by competing species and maximize the development of newly established regeneration. Therefore, historical or permanent experimental sites, such as those examined in the present study, are of paramount importance to allow the evaluation of the actual impact of logging operations on the long-term vegetation dynamics and the development of more effective silvicultural strategies. Not only existing sites should be maintained, but also new ones should be established. However, the design of new long-term experimental sites must be based on the use of appropriate statistical designs, and control plots in which there is no treatment must be implemented. A weakness of the present study is the lack of control plots. For the present study, the availability of control plots would have allowed us to evaluate more precisely the effect of disturbances following the harvest operations. We assumed to a certain extent that the forests under study were in steady-state conditions, that is, very little change in composition for the time frame of the study. As a consequence, the evaluation of the extent to which disturbances were important was based on the comparison with preharvest stand information. Nevertheless, changes in species composition can occur.

Conclusion

This study confirms that a major short-term change associated with high-intensity diameter-limit cutting in balsam fir – yellow birch ecosystems, i.e., an increase in the proportion of hardwood species, can last for more than 50 years. Future stands, especially those harvested in the summer, could see their proportion of birch species increase substantially. These changes to species composition were the result of interactions between the types of logging operations, the density and composition of advance regeneration, the microsite conditions after logging, and the length and severity of spruce budworm outbreaks.

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