Forest Structure and Small-Mammal Responses to Variable-Retention Timber Harvest in the Cape Breton Highlands of Nova Scotia

By

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ABSTRACT

American marten (*Martes americana* Turton) are endangered in Nova Scotia and the population on Cape Breton Island is critically low. A marten recovery strategy is in place but there are large gaps in information regarding future prey abundance and forest structure in managed stands across the Cape Breton Highland Plateau. The Crowdis Mountain study area was established in 2002 by StoraEnso, now New Page, Port Hawkesbury Limited, to study the effects of variable-retention harvesting techniques on habitat requirements of marten. The goal of this study was to determine the response of small mammals, standing and downed dead wood (SDDW), and understory ground vegetation to these alternative harvesting techniques. Sampling occurred from May to September pre-treatment in 2002 and post-treatment 2003 and 2005. It was concluded that treatments did not have a significant effect on small mammals or density of standing and volume of downed dead wood. Small mammals displayed an increasing trend over the entire study area and these increases were correlated with percent cover of fern, slash/fine debris, and CWD volume. Increases in SDDW were found when snag density and CWD volume data were combined for all treatment units but treatment effect was found to be non-significant. This study confirmed that because of past silvicultural practices, stands in the Crowdis Mountain study area had low small-mammal abundances, understory cover, and SDDW. Experimental harvesting treatments implemented were economical and maintained minimum coarse stand-type requirements of marten and did not negatively affect small-mammal abundances. However, at 50 years old, stands were showing signs of wind damage and increasing trends in small-mammal abundance and SDDW recruitment, independent of silvicultural intervention.

Key words: marten, small mammals, coarse woody debris, silviculture, balsam fir.

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1.0 INTRODUCTION

American marten (*Martes americana* Turton) are listed as endangered in Nova Scotia and the population on Cape Breton Island is estimated to be between 15 and 30 individuals (Nocera *et al.* 1999, Scott 2001, Scott and Hebda 2004). The decline in numbers was a result of forest fragmentation and large reductions of mature coniferous forest habitat by natural and anthropogenic disturbances (NSAMRT 2002). The majority of forest stands across the highland plateau are < 30 years old for two reasons. First of all, spruce budworm (*Choristoneura fumiferana* Clem) infestations began in the late 1970s and continued through the 1980s, killing over 80% of all balsam fir (*Abies balsamea* (L.) Mill) stands. Secondly, salvage operations removed 90% of merchantable timber, which further reduced the average stand age and removed standing dead trees.

In 2004, the Nova Scotia Department of Natural Resources (NSDNR) accepted interim planning, developed by StoraEnso Port Hawkesbury Limited, the largest forest licensee of public lands in the area, to retain and increase marten habitat over the Cape Breton Highlands license area (StoraEnso 2004). Using parameters identified by the Nova Scotia American Marten Recovery Team (NSAMRT 2002), a spatial analysis was conducted and a marten habitat management zone (MHMZ) was identified. The MHMZ was based on continuity and connectivity of forest stands, meeting coarse cover-type criteria for marten habitat (height ≥ 6 m, basal area ≥ 18 m^2, and conifer species $\geq 30\%$ of total basal area) (StoraEnso 2004). The MHMZ maintains 20 000 ha of marten habitat, projected to increase to 50 000 ha by 2030. However, it is not known if meeting the minimum spatial and coarse cover-type criteria will maintain sufficient structure and prey-base required by marten to maintain reproductive success (NSAMRT 2002).

The Crowdis Mountain study area, included within the MHMZ, was identified by the woodlands division of StoraEnso to study the effects of alternative harvesting techniques on habitat attributes required by marten (StoraEnso 2004). The study area encompasses 50-year-old, naturally regenerated, pre-commercially thinned, coniferous stands and represents future forest conditions of the managed forest covering a large portion of the Cape Breton highland plateau (StoraEnso 2004). It has been suggested that because of past silvicultural activity, stands within the study area have lower small-mammal abundances and structural complexity than stands initiated by fire or insects (NSAMRT 2002). Stands in the study area meet stand-cover criteria for marten habitat, but key elements affecting small-mammal abundances and habitat quality, such as horizontal structure and vertical heterogeneity, have not been assessed. The goal of the harvesting treatments was to remove timber while retaining minimum amounts of cover required by marten and small mammals. It was hypothesized that harvesting would increase the amount of forest edge and reduce canopy closure, eventually increasing understory-vegetation cover and occurrences of wind-thrown trees.

A detailed literature review on marten, small mammals, and coarse woody debris (CWD) was completed (Appendix I). To summarize, research on eastern populations of marten suggests that complex physical structure, overhead cover, and high small-mammal densities are key characteristics of high-quality marten habitat (Sturtevant *et al.* 1996, 1997, Payer and Harrison 2000a, 2000b, 2003). Complex physical structure in the form of CWD has been positively associated with small-mammal habitat (Bowman *et al.* 1999, Chambers 2002, Fuller *et al.* 2004, Poole *et al.* 2004), particularly red-backed voles (*Myodes gapperi* Vigors) and deer mice (*Peromyscus maniculatus* Wagner) that use it for nests, burrows, cover, foraging substrates, and travel (Stevens 1997, Manning *et al.* 2001, Payer and Harrison 2003). Brown *et al.* (2003) and Pasitschiak-Arts and Messier (1998) suggested that 15-20% coverage of CWD is required for healthy populations of small mammals. Bellhouse and Naylor (1996) stated that > 20% ground cover of downed woody debris was required to maintain both marten and small-mammal populations. Bowman *et al.* (2001) found a significant relationship between abundance of small mammals and CWD in later stages of decay. Keisker (2000) found a positive association of small mammals with the lower range of CWD diameter classes but stated that this relationship was most likely due to an increased percentage of ground cover provided by more-abundant small CWD. Many studies about small-mammal assemblages have been conducted where CWD volume was above limiting thresholds for small mammals and characterization of CWD size and decay-class distributions is lacking (Stevens 1997). Without such information, determining species-specific requirements of CWD characteristics is problematic.

Silviculture treatments applied in the Crowdis Mountain study area were designed to increase complex physical structure while maintaining varying amounts of overstory cover, but it is not known how these treatments will affect assemblages of small mammals. Small mammals respond differently to forest canopy removal depending on geographic location, harvesting method, and original stand condition (Parker 1989, Orrock *et al.* 2000). Clough (1987) and Potvin *et al.* (1999) reported that small-mammal abundances were not affected by silviculture when advanced regeneration was protected. Ream (1981) and Von Trebra *et al.* (1998) stated that red-backed voles respond negatively to canopy removal. Ream (1981) and Fox (1983) found deer mice to respond positively to canopy removal. This is further supported by Potvin *et al.* (1999) who described abundances of deer mice doubling in response to timber harvesting. Merritt (1981) stated that red-backed voles are positively associated with ground cover but that canopy cover was less important. Studies by Thompson and Curran (1995) and Poole *et al.* (2004) suggest that understory vegetation and volume of CWD determine habitat use by marten and small mammals more strongly than age-class and species composition.

The objective of this study was to test the effects of variable-retention silvilculture on standing and downed dead wood (SDDW) and small-mammal abundances in mature, pre-commercially thinned, balsam-fir forest of Cape Breton. If small mammals in the Crowdis Mountain study area respond similarly to overstory removal as described in other studies, it is expected that strip-cut treatments (partial removal) will have no negative effect on small mammals. However, patch-retention treatments (clear-cut patches) will have a negative effect on red-backed voles and a positive effect on deer mice. The study was designed to test the null hypothesis that variable-retention harvesting will not have a significant effect on capture rate of small mammals, understory and overstory vegetation characteristics, or density/volume of SDDW.

Testing the responses of forest structure and small mammals to variable-retention timber harvesting will provide baseline information important to the development of harvesting techniques to maintain landscape and site-level requirements of marten on Cape Breton Island, NS. If silviculture treatments increase small-mammal abundance, dense understory vegetation, and SDDW, while maintaining coarse cover-type criteria for marten, then treatments may be used to extract valuable timber resources while maintaining marten habitat in mature, pre-commercially thinned, balsam-fir stands across the highland plateau.

2.0 METHODS

2.1 STUDY AREA

The study area encompasses 1225 ha of forest north of Crowdis Mountain (46.160 N, 60.820 E), in the southern reaches of the highland plateau on Cape Breton Island, Nova Scotia (Fig. 1).

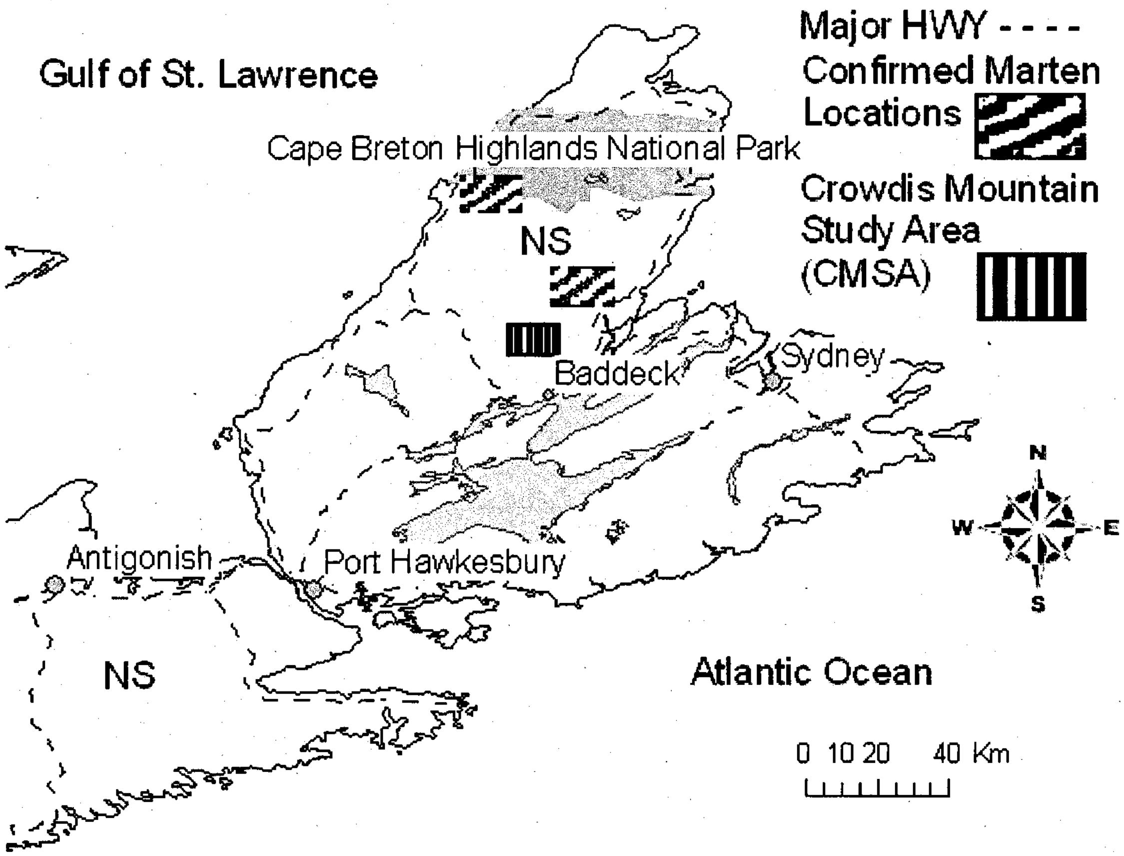

Figure 1. Cape Breton Island, Nova Scotia, showing the Crowdis Mountain study area location and confirmed marten (*Martes americana*) locations.

The mean summer and winter temperatures are 14.5 °C and -2.5 °C, respectively, annual precipitation ranges from 1000 mm to 1600 mm, and soils are primarily shallow, loamy podzols (fertile soil, mix of clay, sand, silt and minor organic) (Environment Canada 2005). The study area is surrounded by the Cape Breton Highlands Ecoregion in the Altlantic Maritime Ecozone (Rowe 1972). Although it is included within the Acadian Forest Region, it resembles stand composition and characteristics more closely related to boreal forest (Thompson *et al.* 2003). Forest composition in the study area is typical of the highlands plateau, composed almost entirely of sparse, low-growing balsam-fir-dominated conifer stands. Much of the forest was clear-cut in the mid-1950s, and regenerated naturally with balsam fir. Stands were motor-manually pre-commercially thinned to increase yield, and later sprayed with Btk (*Bacillus thuringiensis* var. kurstaki) to protect against damage by eastern spruce budworm (*Choristoneura fumiferana* Clem) (Brander 1994).

The study area has been classified as class 2, or fair, marten habitat, defined as being at least 33% dominated by softwood, > 6 m tall, with minimum basal area > 18 m^2/ha (Naylor *et al.* 1994, NSDNR 2003). It has been suggested that the study area lacks the necessary structure and prey base to be more than marginal marten habitat throughout the mature stage because of past harvesting and pre-commercial thinning (NSAMRT 2002).

2.2 EXPERIMENTAL DESIGN

Twenty 12.6 ha circular treatment areas were identified within the study area (Fig. 2).

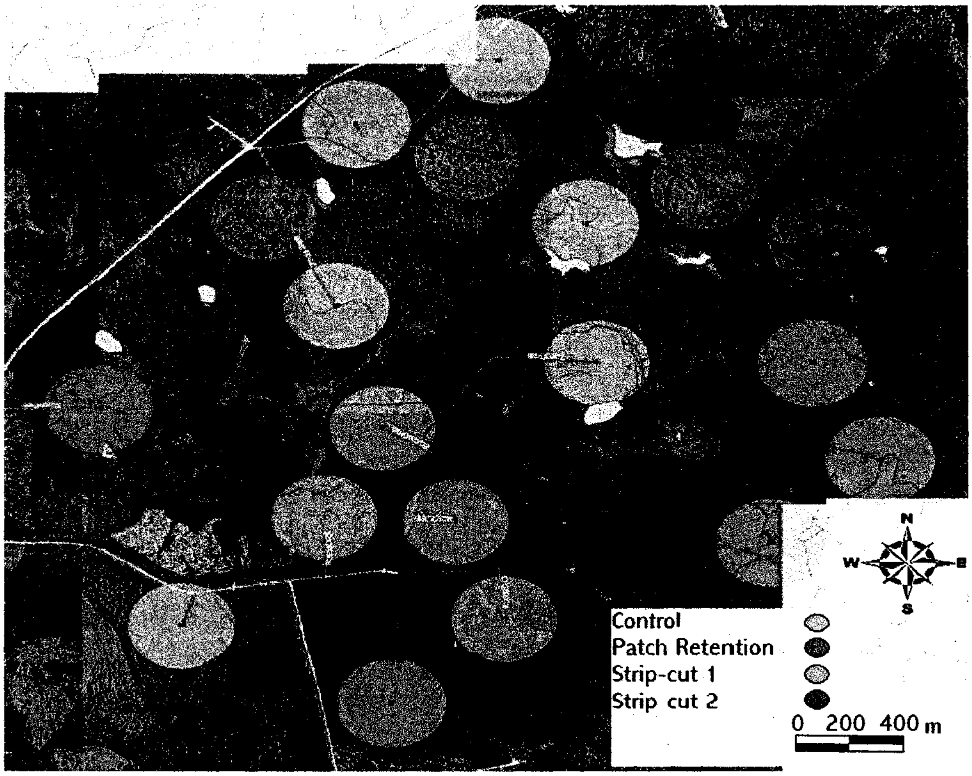

Figure 2. Crowdis Mountain study area in 2002 showing 12.6 ha experimental treatment units.

Circular shapes were selected to reduce area-to-edge ratio and to isolate treatment areas from one another. A total of 200 permanent sample points (PSPs) were established: ten per treatment area, spaced 100 m apart along three parallel transects oriented north and south. At each PSP, during the summers of 2002, 2003, and 2005, overstory and understory vegetation and SDDW characteristics were measured using fixed-area plots. Small-mammal trapping grids were centred in each of the 12.6 ha circular treatment areas and small mammals were trapped between August and September pre-treatment 2002, and post-treatment in 2003 and 2005.

2.3 Silvicultural Treatments

StoraEnso woodlands management team developed three silviculture treatments designed to increase understory vegetation, stratify age-class structure, and increase SDDW. Treatments (strip-cut 1, strip-cut 2, and patch retention) were each randomly assigned to five of the twenty, 12.6 ha circular test areas. The remaining five areas were used as control plots to enable comparisons of treated with non-treated test areas. Treatments were implemented and completed between October and November 2002, using an ENVIRO® single-grip processor and forwarder combination.

2.3.1 Strip-cut 1

The strip-cut-1 silviculture treatment was designed to remove 40% of the overstory, while retaining 22-25 m^2/ha basal area. Adjacent extraction trails, 6 m wide and spaced 20 m apart, were clear-cut and 15-40% of the basal area was removed within 5 m of each side of the extraction trail. A 4-m-wide undisturbed strip was left between each harvested pass (Fig. 3).

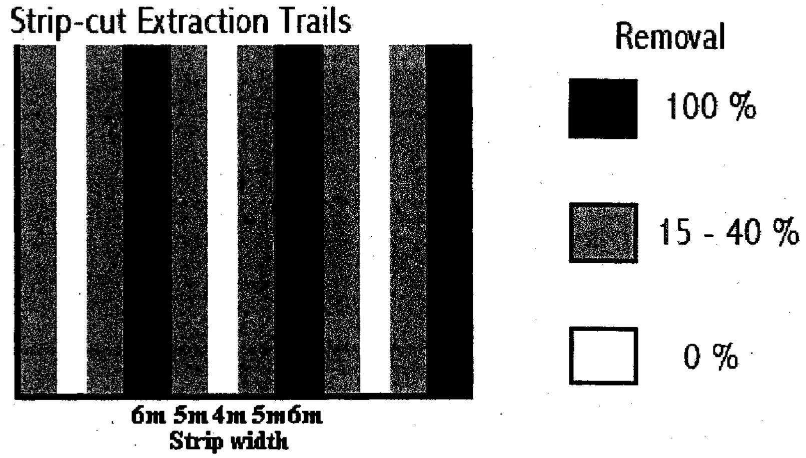

Figure 3. Strip-cut harvest treatment trail dimensions and removal percentages implemented in the Crowdis Mountain study area.

2.3.2 Strip-cut 2

The strip-cut 2 silviculture treatment was implemented as described for the strip-cut 1 treatment, but 12-14 trees/ha were cut and positioned on the ground to increase elevated CWD and structural features believed to be important to marten. These culled trees were selected based on having diameters (dbh ≥ 15 cm), crowns (width ≥ 2 m), and branch sizes that were larger than non-selected trees. Culled trees were cut at the base, between 0.5 m and 1.5 m up the trunk, and the butt-end was placed on top of the cut stump, to resemble naturally broken or up-rooted stems. It was suggested that these trees would be important to small mammals and marten during winter months by creating subnivean space beneath the snow.

2.3.3 Patch Retention

The patch-retention treatment removed 50% of the overstory in a checkerboard pattern of 0.75 ha clear-cut and undisturbed patches. The goal of this treatment was to increase understory vegetation through direct canopy removal, decrease the amount of harvested edge compared to the strip-cut treatments, and create openings large enough to increase wind-throw effects along the forested edges.

2.4 DATA COLLECTION

2.4.1 Vegetation

Basal area of balsam fir was measured using a 2x basal-area-factor prism at each permanent sample point (Husch *et al.* 1982). Average height, diameter breast height outside bark (dbh), and stump age were calculated using a sub-set of four sample trees, identified as "in" trees. Height was measured using a Suunto® clinometer, dbh was measured using a diameter tape, and age at stump height was estimated from cores taken with an increment borer.

Ground vegetation was sampled in two 10-m^2 circular fixed-area plots, 5 m east and 5 m west of the PSP centre. Plots were separated into quarters, and the percent cover of trees, shrubs, ferns, grasses, herbs, mosses/lichens, slash and fine debris, mineral soil, and leaf litter of each quadrat was estimated and totaled. Tree species were recorded in three height classes (< 0.5 m, 0.5 m to 1.3 m, and > 1.3 m) and the number of trees in each height class was recorded.

2.4.2 Standing and Downed Dead Wood (SDDW)

SDDW > 7.5 cm diameter (Graham *et al.* 1994, Stevens 1997) was inventoried within a 5-m-radius circular plot (Taylor 1997) centred at the staked PSP centre in each treatment area. Dead wood was separated into two groups: logs (inclining > 45° from vertical) and snags (inclining < 45° from vertical, height > 2 m). Midpoint diameter and length of each piece in the circular plot was measured with calipers and a measuring tape. Decay class of each piece was estimated according to Bartels *et al.* (1985).

2.4.3 Small-Mammal Trapping

Small-mammal trapping grids were centred within each 12.6 ha circular treatment area and consisted of 25 sampling points spaced equidistantly 25 m apart. A Sherman single-capture box trap and an Ugglan multiple-capture cage trap was set at each sampling point within 5 m east and west of the sampling point centre. Traps were placed in best available microsite locations near logs, stumps, or cover (Bole 1939) and as described by Von Trebra *et al.* (1998) and Krohne and Hoch (1999), a total effective trapping area of 1.56 ha was estimated. Sherman traps are reported to have greater capture success than the Victor Tin Cat (Belant and Windels 2007) or other multiple-capture cage traps (Sone and Tojo 1993). However, two trap types were used to reduce any species bias that may result if lighter small mammals (e.g. masked shrews) were not heavy enough to trigger the closure mechanism of the Sherman trap (Nicolas and Colyn 2006). Also, if small-mammal densities are high, multiple-capture cage traps might increase capture or recapture potential. Live traps were chosen because the small-mammal density was suspected to be low and repetitive kill trapping and removal over field seasons would have been detrimental to the population and thus the data analysis. Traps contained cotton wadding and a water source (piece of apple) and were baited with a mixture of peanut butter, bacon fat, and rolled oats (Sekgororoane and Dilworth 1995) and set for three consecutive nights (Bole 1939) for a total of 150 possible trap-nights for each plot. Pre-baiting was not used in this study but capture rates may have been higher if it had been included in sampling (Delany 1975).

Trapping was conducted during August and September in 2002, 2003, and 2005. It was thought that small-mammal abundance would be greatest during this period as found in other studies by Poole *et al.* (2004) and Peirce and Peirce (2005). Small-mammal trapping in late-summer/early-fall is believed to be less affected by differential survival rates caused by resource limitations during winter and early-spring conditions. Traps were set before noon each morning and species caught, recaptures, and traps that were sprung or failed were recorded.

To address recapture rate and movement between traps and home range, voles and mice were ear-tagged, using a number 1005-1 Monel ear tag from National Band and Tag

Co., and then released (Delany 1975, Bowman *et al.* 2001). This effort proved ineffective for calculating a population index described by Hayne (1949) or Jolly-Seber population estimates described by Manning *et al.* (2001) due to the lack of recaptures. Burge and Jorgensen (1973) and Delany (1975) suggested that 15-100 recaptures were required to estimate individual small-mammal home range. Direct enumeration of small-mammal captures per unit area (Parker 1989) and capture rate per 100 trap nights (Potvin *et al.* 1999) were calculated and used as an estimate of relative abundance. Hannon *et al.* (2002) subtracted 0.5 trap-nights from the total possible for each failed trap. However, in this study one trap-night was subtracted from the total number for each sprung or failed trap that did not yield a capture because of the lack of consensus regarding this practice. Live-animal capture and handling guidelines were followed (BC Ministry of Environment, Lands and Parks 1998) and activities were approved by the Animal Care Committee at Lakehead University. All mortalities were recorded, collected, and donated to the University of New Brunswick.

2.5 DATA ANALYSIS

This study was a completely randomized design and all analyses were conducted using Microsoft® Office Excel 2003, Statistica 7.1© (StatSoft Inc. 2005) and Datadesk 6.01™ (Data Descriptions Inc. 1996). Three linear models, listed below, were developed to compare measured variables among circular treatment areas.

2.5.1 Model 1

Stand conditions in 2002 (basal area, height, diameter, age, SDDW, and total small-mammal captures per 100 trap-nights) were analysed using a single-factor analysis of variance (ANOVA) to detect if there were differences between randomly assigned treatment groups (C, SC1, SC2, and PR). The output was also used to describe the initial stand conditions of the Crowdis Mountain study area.

The data set for each response variable was tested for homogeneity using Bartlett's Test and was transformed if the calculated critical value exceeded the value given in a Chi-squared distribution table for $k - 1$ degrees of freedom at $\alpha = 0.001$ (Lorenzen and Anderson 1993). In cases where statistical significance did not change between transformed and untransformed data, reported results were based on untransformed data (Conover and Iman 1981).

2002 Data Set
One-factor Analysis of Variance (ANOVA)
General Linear Model: Eq. 1

$$Y_{ijk} = \mu + T_i + \varepsilon_{(i)j} + \rho_{(ij)k}$$

$$i = 1, 2, 3, 4; j = 1, 2, 3, 4, 5; k = 1, 2, 3, \ldots, r$$

where:
Y_{ijk} = the measured response of the j^{th} replicate of the i^{th} silviculture treatment level for the k^{th} sample point.
μ = the overall mean
T_i = the fixed effect of the i^{th} silviculture treatment
$\varepsilon_{(i)j}$ = experimental error
$\rho_{(ij)k}$ = sampling error (sample points nested in treatments)

Equation 1 shows k taking the values 1 through r. Here r represents the number of samples in the treatment group which varies from one response variable to another.

2.5.2 Model 2

Silviculture treatments were implemented in the fall of 2002. Resulting stand conditions in 2003 and 2005 were measured and mean basal area, height, diameter, age, SDDW, and captures per 100 trap-nights were analyzed using a two-factor analysis of covariance (ANCOVA). Model 2 was developed to determine whether the application of silviculture treatments had an effect on response variables in 2003 and 2005 (post-treatment) across the range of the covariate measured in 2002 (pre-treatment). Each data set was tested for homogeneity as for Model 1.

2003-2005 Data Set (using 2002 data as a covariate)
Two-factor Analysis of Covariance (ANCOVA)
General Linear Model: Eq. 2

$$Y_{ijk\ell} = \mu + T_i + V_{(x)} + A_j + TA_{ij} + \varepsilon_{k(ij)} + \rho_{((ij)k)\ell}$$

$$i = 1, 2, 3, 4; j = 1, 2; k = 1, \ldots, 20; l = 1, 2, 3, \ldots, r$$

where:

$Y_{ijk\ell}$ =the measured response of the k^{th} replicate of the i^{th} silviculture treatment level and the j^{th} age class for the l^{th} sample point.
μ = the overall mean
T_i = the fixed effect of the i^{th} silviculture treatment
$V_{(x)}$ = the random effect of the covariate on the i^{th} silviculture treatment
A_j = the fixed effect of the j^{th} year of measurement
TA_{ij} =the fixed effect of the interaction between treatment and year of sampling
$\varepsilon_{k(ij)}$ = experimental error
$\rho_{((ij)k)\ell}$ = sampling error (sample points nested in treatments)

Equation 2 shows l taking the values 1 through r. Here r represents the number of samples in the treatment group which varies from one response variable to another.

2.5.3 Model 3

A step-wise regression model was used to check for a relationship between small-mammal capture rates and measured habitat features as predictors. Capture rate was tested against 18 habitat features: height, diameter, age, basal area, moss/lichen, low shrubs, fern, sedges/grasses, club moss, slash/fine debris, mineral soil, leaf litter, CWD length, CWD diameter, CWD volume, snag density, treatment, and year. The model was generated forward and then backward, sequentially removing variables to find the best possible relationship at a tolerance level of $\alpha = 0.05$.

2002, 2003 and 2005 Data Set
Step-Wise Regression
General Linear Model: Eq. 3

$$y = b_0 + b_1x + \varepsilon$$

where:
y = small-mammal captures/100TN ($N = 60$)
x = habitat features ($x = 1,\ldots 18$)
b_0 = set of intercepts (value of each y when each $x = 0$)
b_1= set of coefficients, one each for each x
ε = error

3.0 RESULTS

3.1 STAND CONDITIONS 2002

Allocating treatment type to the 12.6 ha treatment units in 2002 did not have a significant effect ($P \geq 0.05$) on measured stand descriptors and confirmed that there was no bias associated with treatment-type delineation pre-treatment (Appendix II). Treatment areas encompassed relatively uniform, 50 year-old, balsam-fir-dominated stands (Table 1).

Table 1. Crowdis Mountain study area pre-treatment stand conditions (2002).

	Total Age (yr)	Density (stems/ha)	Height (m)	DBH (cm)	Basal Area (m^2/ha)	Standing Dead Wood (stems/ha)	Downed Dead Wood (m^3/ha)	Total Small Mammals (captures/100 trap nights)
Mean	48.60	878.00	11.60	21.20	31.00	299.90	34.40	1.52
SE	9.30	240.00	1.60	4.90	6.40	183.20	16.30	0.20
CV	0.11	0.27	0.14	0.23	0.21	0.61	0.47	0.14

Mean tree diameter was larger (approximately 6 cm) than reported in other pre-commercially thinned, balsam-fir stands at 50 years of age (Penner *et al.* 2006) and SDDW was highly variable and unevenly distributed. Prior to treatment in 2002, red-backed voles were captured more frequently than other small-mammal species (Table 2).

Table 2. Summary of small-mammal captures (100 trap-nights^{-1}) for each treatment group and species (red-backed vole (RBV), masked shrew (MS), deer mouse (DM), short-tailed shrew (STS), short-tailed weasel (STW), and northern flying squirrel (NFS)) in the Crowdis Mountain study area between August and September 2002.

Treatment Group	Species					
	RBV	MS	DM	STS	STW	NFS
Control	2.06	0.14	0.00	0.27	0.00	0.00
Strip-cut1	0.96	0.14	0.27	0.00	0.14	0.00
Strip-cut 2	1.50	0.14	0.00	0.00	0.00	0.00
Patch Retention	0.27	0.00	0.13	0.00	0.00	0.13
Total Individuals	35.00	3.00	3.00	2.00	1.00	1.00
100TN^{-1}	1.19	0.10	0.10	0.07	0.03	0.03

Six species were captured but only three species (red-backed vole, masked shrew, and deer mouse) were caught in more than one treatment area.

Small mammals were completely absent in 5 of the 20 trapping grids and there was high variation between delineated treatment units (Fig. 4).

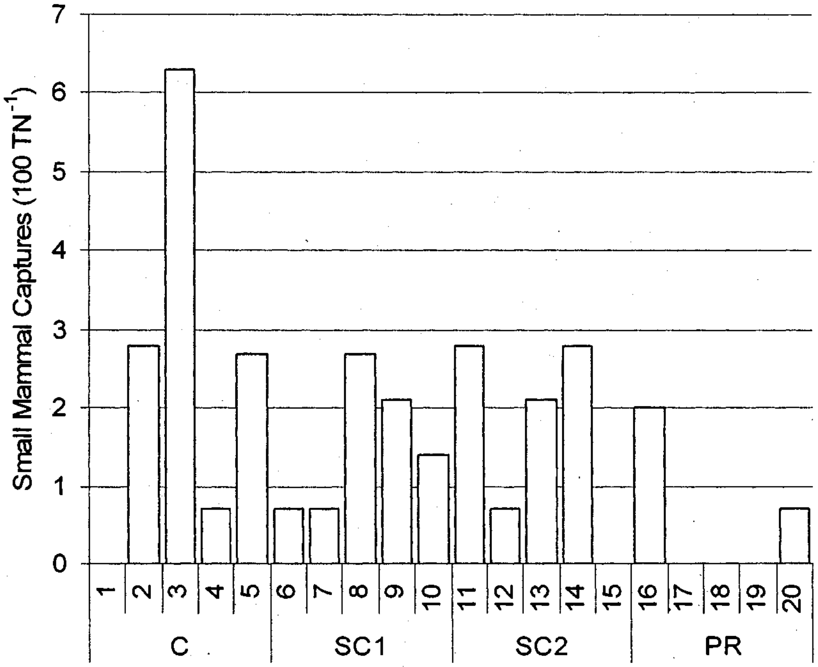

Figure 4. Small-mammal capture rates in the 12.6 ha treatment units numbered 1-20 in the Crowdis Mountain study area fall 2002 (control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR)).

Mean small-mammal capture rates for each treatment group were not significantly different (P = 0.2759, F (3,16) = 1.4116) and there was high variability within and among treatment groups (Fig. 5).

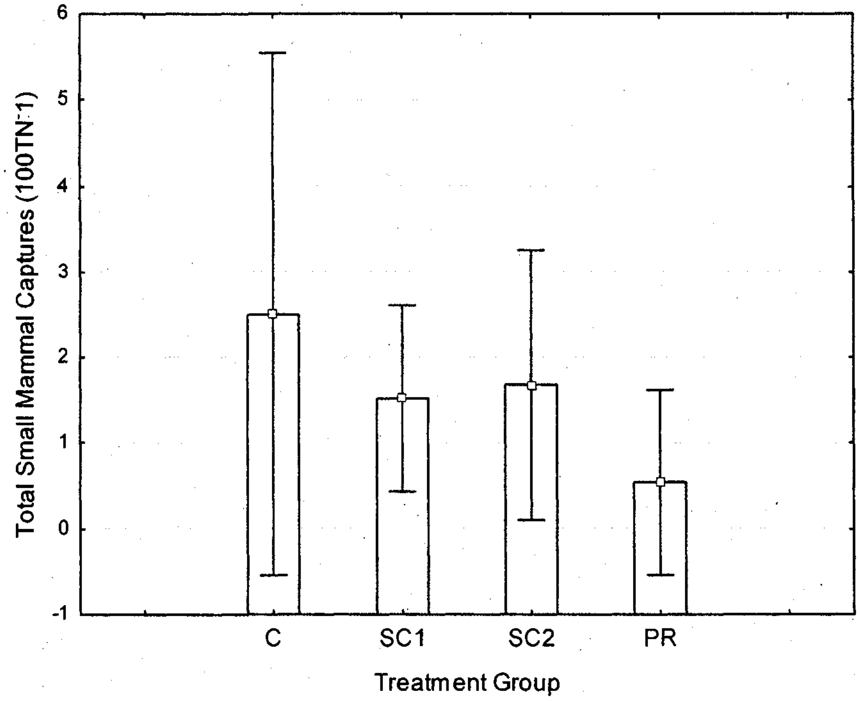

Figure 5. Fall-2002 mean small-mammal capture rates stratified by treatment (control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR)) in the Crowdis Mountain study area (vertical bars denote 95% confidence intervals).

3.2 EFFECTS OF TREATMENTS

Treatment (T_i) was found to have a significant effect on age, height, DBH, and basal area variables (Table 3).

Table 3. Crowdis Mountain study area Analysis of Covariance (ANCOVA) results showing significance (*P*) of silviculture treatment (T_i), pre-treatment covariate (V_x), and year (A_j) on mean tree age, height, diameter breast height (DBH), basal area (BA), standing and downed dead-wood (SDDW), and small-mammal captures (SM).

Source[a]	d.f.[b]	Age (yr)	Height (m)	DBH (cm)	BA (m^2/ha)	SDDW (m^3/ha)	SM (100 TN^{-1})
T_i	3	**<0.0001**	**<0.0001**	**0.0061**	**<0.0001**	0.8221	0.2825
V_x	1	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**<0.0001**	**N/A**
A_j	1	0.1387	0.3322	0.4850	0.1305	0.1696	0.7105
TA_{ij}	3	0.5210	0.4182	0.6804	0.4456	0.4107	**0.0415**
$\varepsilon_{k(ij)}$	240						

[a] T_i (i = 1, 2, 3, 4) = silviculture treatments; V_x (x= 1, 2) = 2002 covariate; A_j (j = 1, 2) = year of sampling.
[b] Degrees of freedom for $\varepsilon_{k(ij)}$ for age, height, DBH, basal area, SDDW was 240 and for SM captures was 40. Bolded values are significant (α=0.05).

A significant effect of treatment on SDDW and small-mammal capture rate was not detected. Bonferroni post-hoc testing showed that for age, height, and DBH variables, the patch-retention treatment was the only treatment with significantly different values than the control treatment (Appendix III).

In 2002, the covariate data (V_x) measuring age, height, diameter, basal area, and SDDW (pre-treatment) was significantly different from the 2003 and 2005 data (post-treatment). Bonferroni post-hoc testing showed that only the patch-retention treatments had significantly lower mean values for all measured variables after harvest (Appendix III).

Basal areas for all treatments were significantly different from the control plots and the patch retention treatment had significantly lower basal area than the strip-cut-1 treatment but not the strip-cut-2 treatment (Fig. 6).

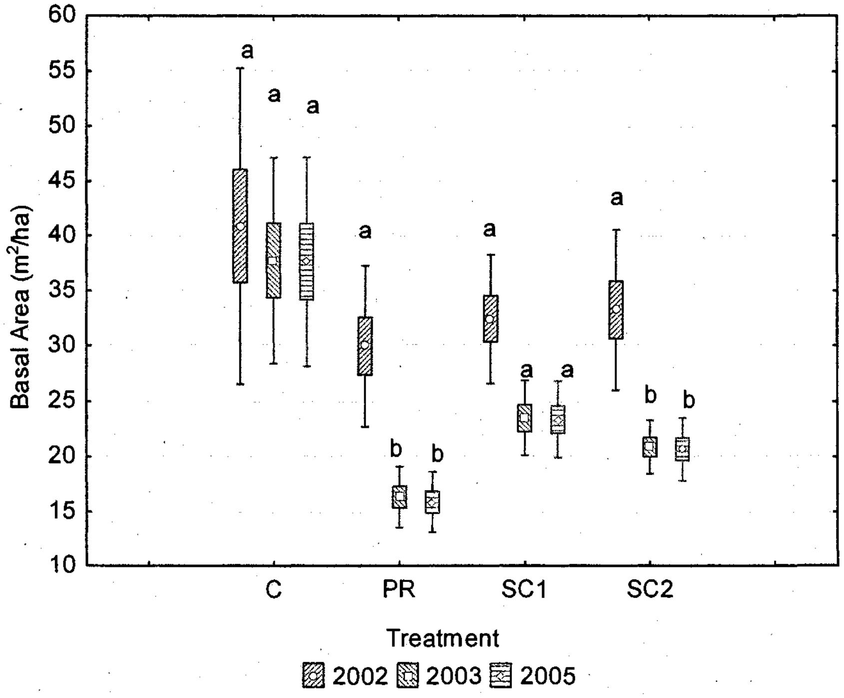

Figure 6. Response of basal area (m^2/ha) to treatment type (control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR)) in the Crowdis Mountain study area (Boxes indicate standard error and whiskers indicate 95% confidence intervals; within treatment, means with the same letter are not significantly different according to post-hoc comparison ($P > 0.05$) ($n = 20$)).

Within treatments, basal area after harvesting in 2003 and 2005 was significantly lower in the patch retention and the strip-cut 2 treatments but was not significantly lower in the strip-cut 1 treatment.

3.3 STANDING AND DOWNED DEAD WOOD (SDDW)

3.3.1 Snags

Density of snags was not significantly different among years or treatment types and ranged between 220 and 390 stems/ha (Fig. 7).

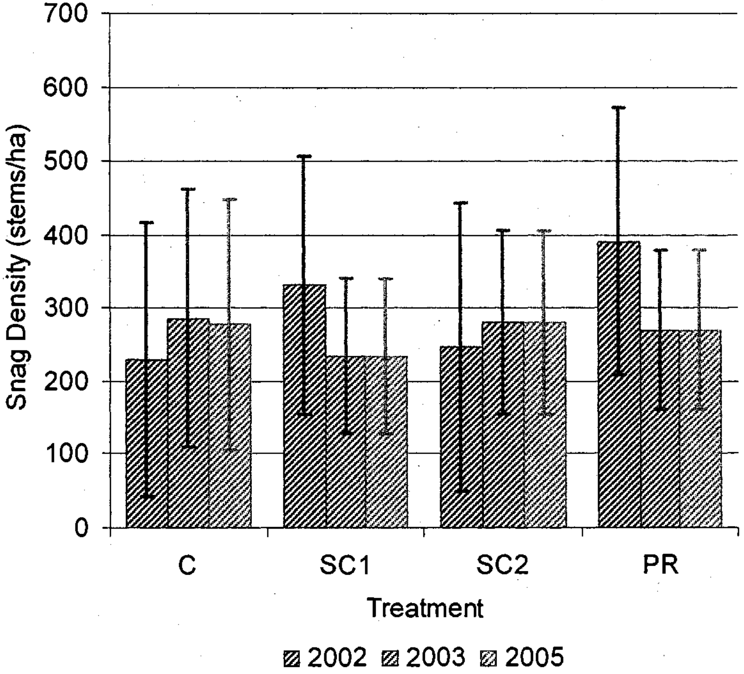

Figure 7. Snag density for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (whiskers indicate 95% confidence intervals).

The diameter-class distribution of snags followed similar patterns in the three years of sampling (Fig. 8).

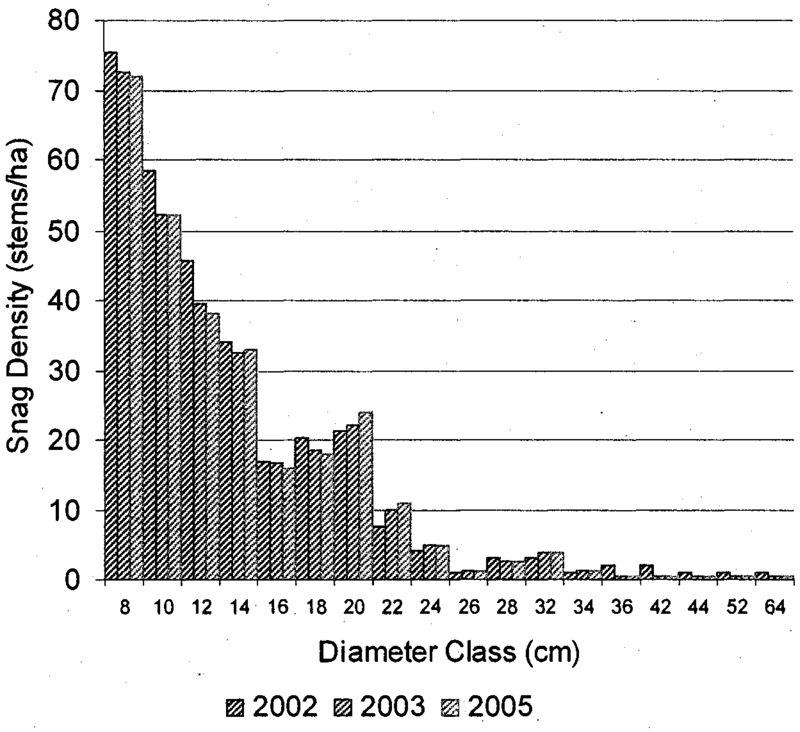

Figure 8. Crowdis Mountain study area snag diameter-class distribution (all 20 sites combined).

Number of snags decreased steadily as diameter-class increased from 8 cm to 16 cm. Snag abundance sharply increased between diameter classes 16 cm and 20 cm and then decreased again to fewer than 5 snags/ha until a maximum diameter-class of 64 cm was reached.

Snag decay-class distribution was bell-shaped but skewed to the right of the median (decay-class 5) pre- and even more so post-treatment (Fig. 9).

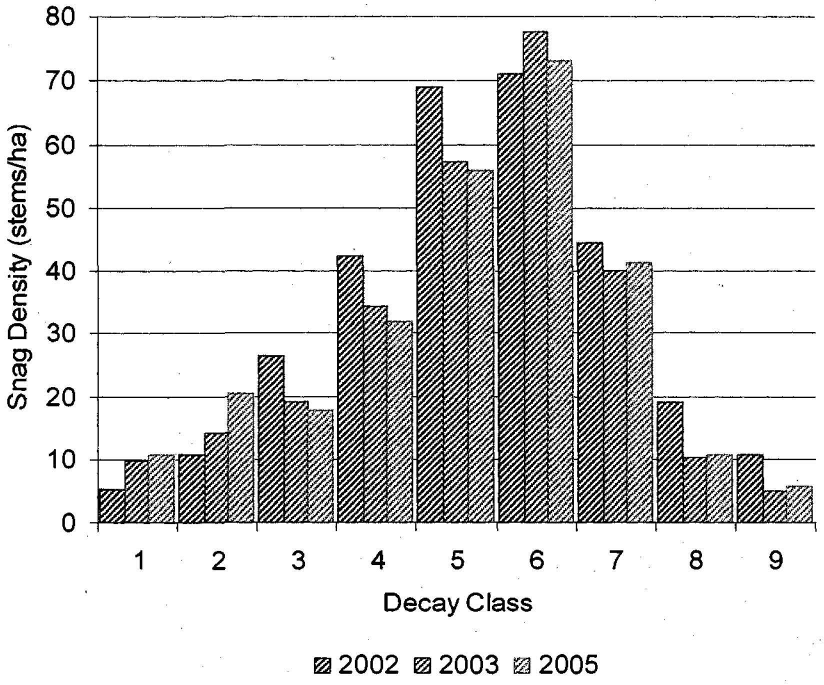

Figure 9. Crowdis Mountain study area snag decay-class distribution (all 20 sites combined).

Roughly 50% of the total snag density was classified as decay-class 5 and 6.

3.3.2 CWD

The volume of CWD was not significantly different among treatment units or years and ranged between 23 and 52 m^3/ha (Fig. 10).

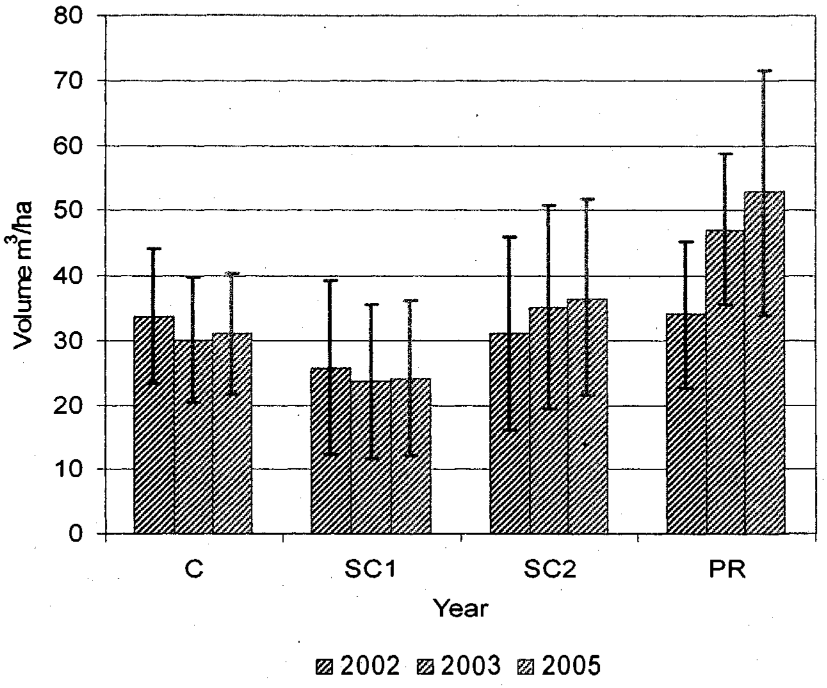

Figure 10. Volume of downed CWD for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (whiskers indicate 95% confidence intervals).

Mean volume of CWD in 2003 and 2005 was higher in the strip-cut-2 and patch-retention treatments.

In 2003 and 2005, the volume of CWD between diameters 8 and 16 cm decreased and the volume of CWD between diameters 18 to 48 cm increased (Fig. 11).

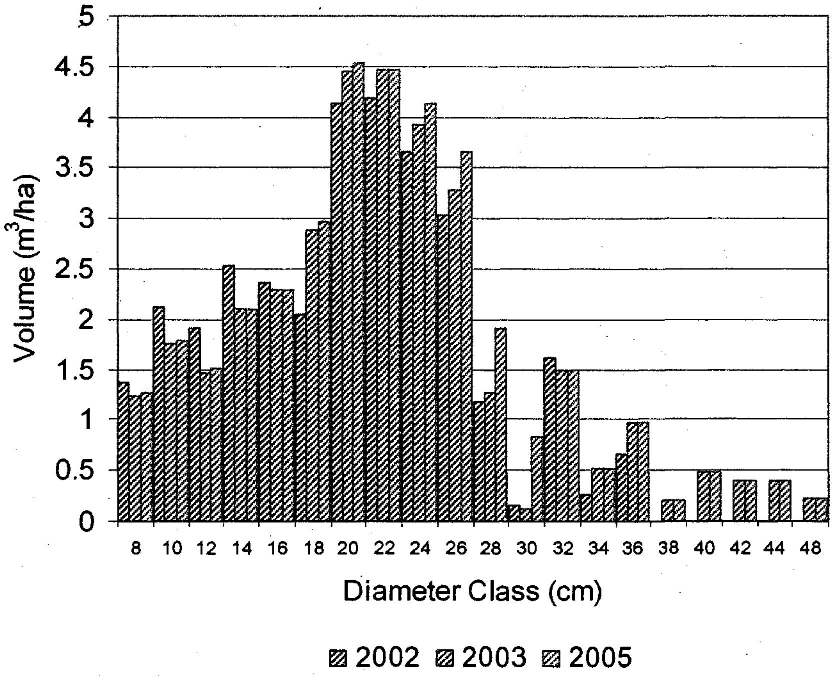

Figure 11. Crowdis Mountain study area CWD diameter-class distribution (all 20 sites combined).

Volume of CWD increased in all decay classes post-treatment except decay-class 5 and remained relatively constant in 2005 except for decay-classes 1 and 2, which continued to increase (Fig. 12).

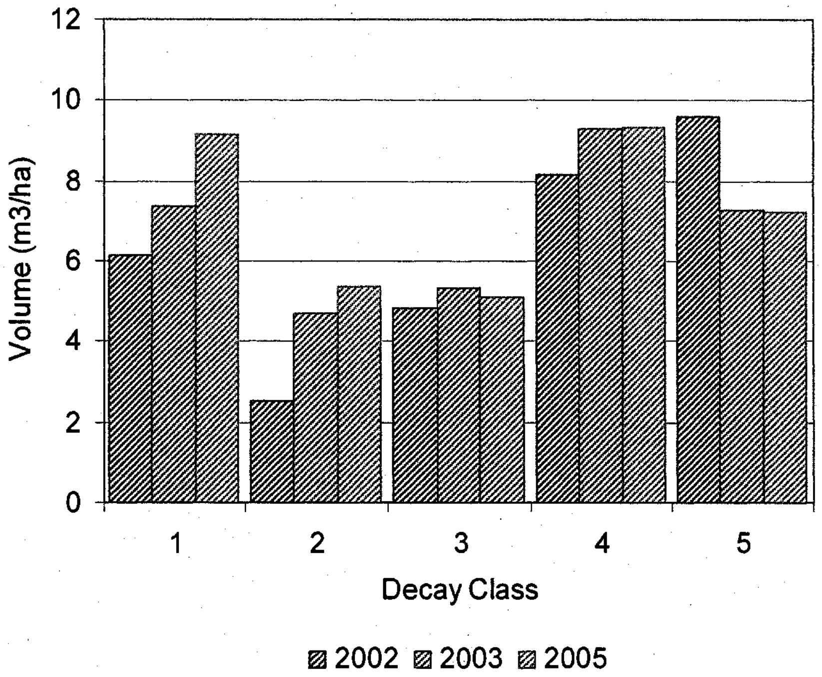

Figure 12. CWD volume distribution for decay classes 1 - 5 in the Crowdis Mountain study area.

3.4 SMALL MAMMALS

Seven small-mammal species and 370 individuals were captured during the three trapping sessions in 2002, 2003, and 2005. In 2002, 45 individuals were captured during 2940 trap-nights; in 2003, 177 individuals were captured during 2980 trap-nights; and in 2005, 148 individuals were captured during 2779 trap-nights. Percentage of total captures by species in 2002, 2003, and 2005 respectively were: red-backed voles (*Myodes gapperi* Vigors) (77.8, 50.3, 78.4); masked shrews (*Sorex cinereus* Kerr) (6.7, 33.9, 4.1); deer mice (*Peromyscus maniculatus* Wagner) (6.7, 5.6, 5.4); short-tailed shrews (*Blarina brevicauda* Say) (4.4, 6.8, 0.7); short-tailed weasel (*Mustela erminea* Linnaeus) (2.2, 1.7, 8.1); red squirrel (*Tamiasciurus hudsonicus* Erxleben) (0.0, 1.7, 3.4); and northern-flying squirrel (*Glaucomys sabrinus* Shaw) (2.2, 0.0, 0.0).

Uggland multiple capture traps yielded 47.5% of total captures and Sherman single capture box traps yielded 52.5% of the total captures. Uggland traps were responsible for 98.6% of total masked-shrew captures and 36.9% of total red-backed-vole captures. The failure rate of the Sherman box traps was 6.2 times that of the Uggland traps.

The effect of treatment (T_i) or year (A_j) on capture rate was not significant but an interaction effect between these variables was detected (P = 0.045) (Appendix III). Capture rate of small mammals in the control plots was not significantly different between trapping years but displayed an increasing trend (Fig. 13).

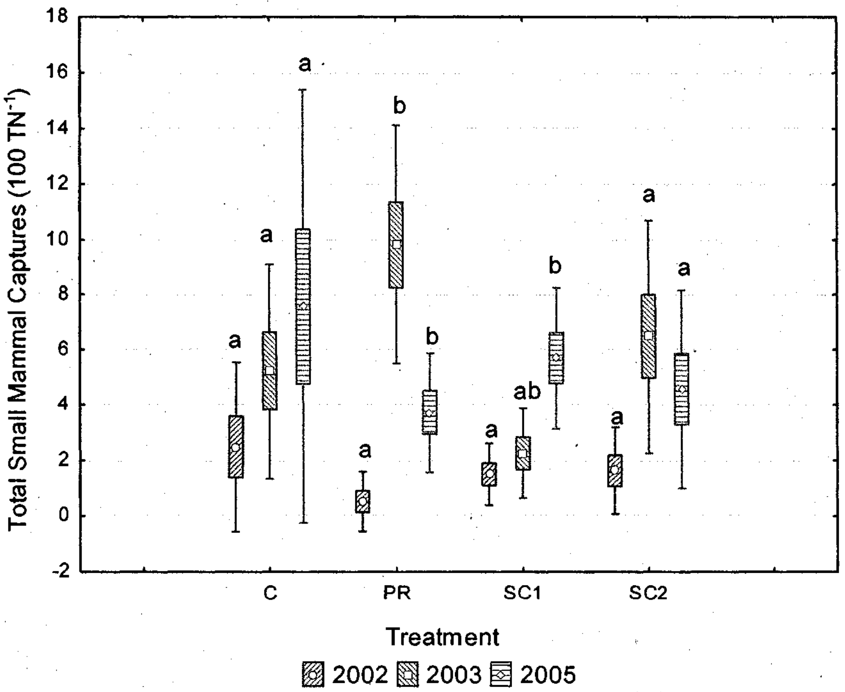

Figure 13. Crowdis Mountain study area small-mammal capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals; within treatment, means with the same letter are not significantly different according to post-hoc comparison (P > 0.05) (n = 20)).

Small-mammal captures across all the harvest treatments were not significantly different from captures in the control. However, within treatments, capture rate increased significantly in the patch retention treatment between 2002 and 2003 and in the strip-cut 1 treatment between years 2002 and 2005. Similar to the control, capture rate in the strip-cut 1 treatment showed an increasing trend over the three years of sampling. Patch retention and strip-cut 2 capture rate trends were also similar, increasing the year after harvest in 2003 and then decreasing three years post-harvest in 2005. Although treatment and year were not found to have a significant effect, mean capture rate across all treatments was lowest pre-harvest.

Red-backed vole captures comprised 65% of total small-mammal captures over the three years of sampling and capture rates between treatments were not significantly different (Fig. 14).

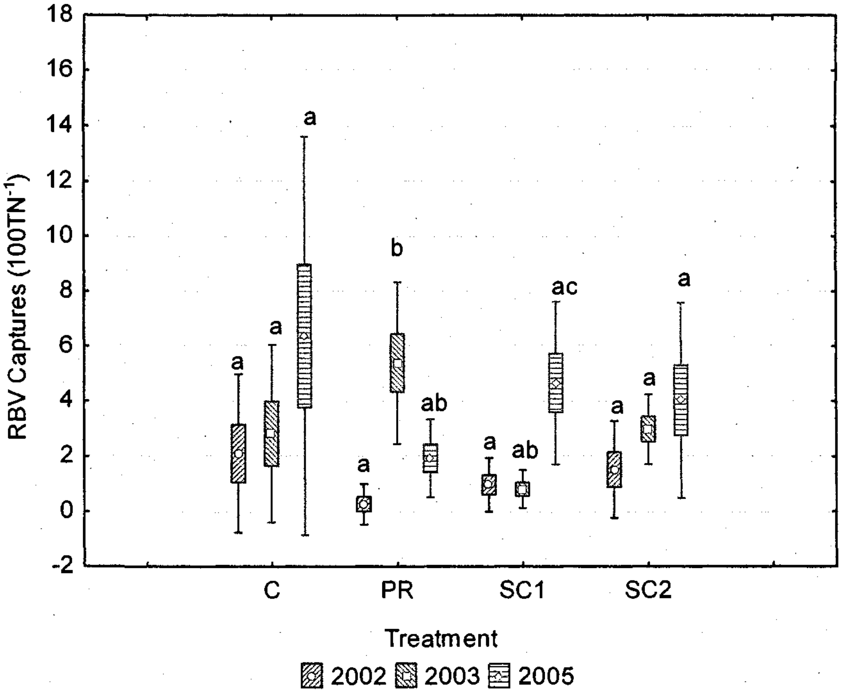

Figure 14. Crowdis Mountain study area red-backed vole (*Myodes gapperi* Vigors) capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals; within treatment, means with the same letter are not significantly different according to post-hoc comparison ($P > 0.05$) ($n = 20$)).

Control and patch-retention red-backed vole capture rates expressed trends similar to total small-mammal captures. Within treatments, vole captures in the strip-cut-1 treatment increased significantly between years 2003 and 2005 and strip-cut-2 vole captures showed an increasing trend similar to vole captures in the control.

Masked-shrew captures were very low in 2002 and 2005 and were completely absent in the patch-retention and the strip-cut-2 treatments respectively (Fig. 15).

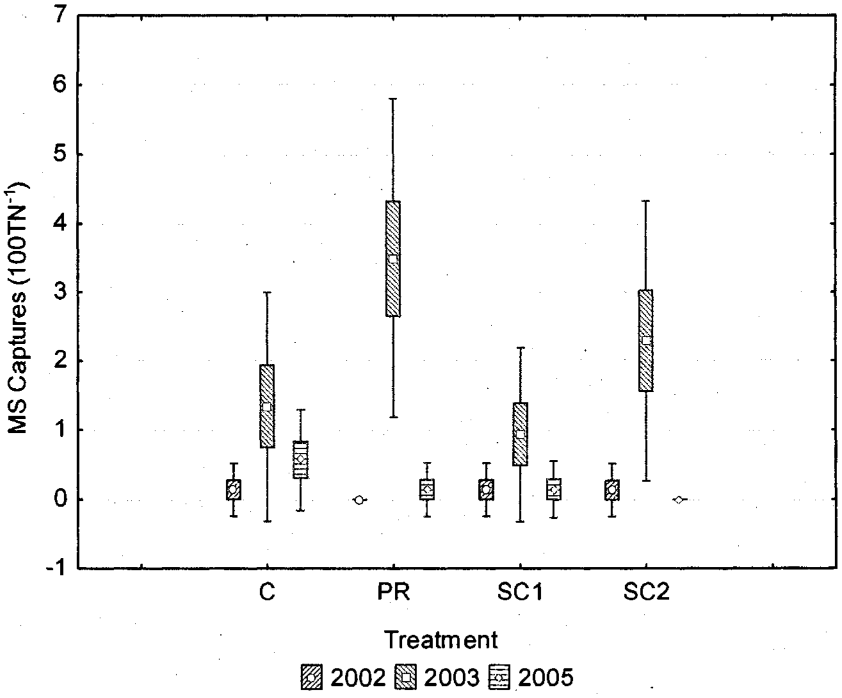

Figure 15. Crowdis Mountain study area masked shrew (*Sorex cinereus* Kerr) capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals).

Although a significant difference between treatments was not found, masked-shrew capture rate, expressed as a percentage of total small-mammal captures, increased to 33.9% of total captures in 2003, compared to 6.7% and 4.1% in 2002 and 2005, respectively.

Deer-mouse captures were also very low, especially during the first year of sampling when one mouse was caught in the patch-retention treatment and two were caught in the strip-cut-1 treatment (Fig. 16).

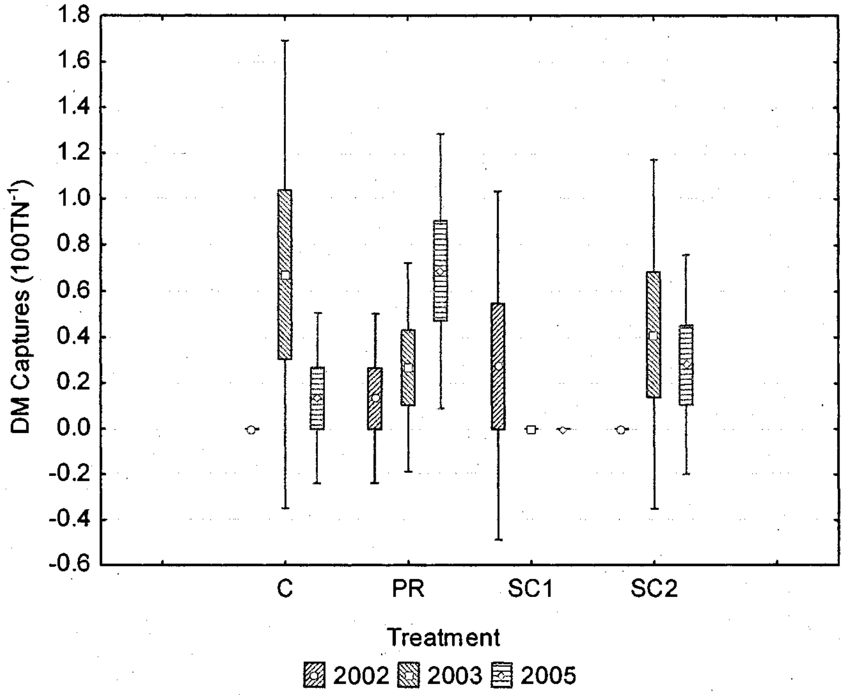

Figure 16. Crowdis Mountain study area deer mice (*Peromyscus maniculatus* Wagner) capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals).

After harvest in 2003 and 2005, deer-mouse captures continued to be low at 10 and 9 individuals, respectively. The patch-retention treatment was the only treatment to have captures of deer mice in all three sampling years.

Short-tailed shrews were captured only in the control plots in 2002 and in the patch-retention plots in 2005 (Fig. 17).

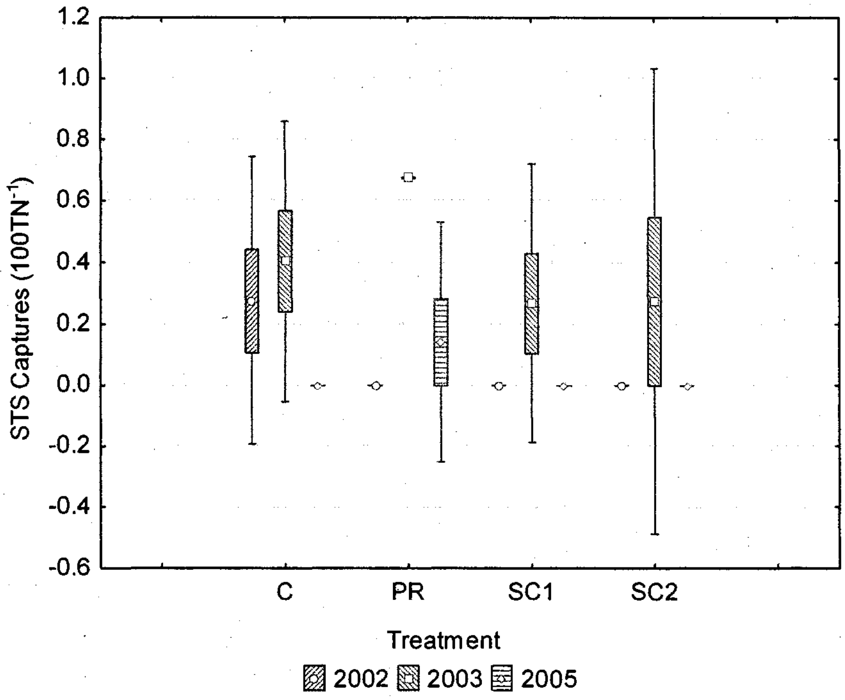

Figure 17. Crowdis Mountain study area short-tailed shrew (*Blarina brevicauda* Say.) capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals).

In 2003, all the treatment types had captures of short-tailed shrews but no treatment had more than one capture in either sampling grid.

Short-tailed weasels were captured in only the strip-cut 1 treatment in 2002 and were absent in 2003 but found in all treatments in 2005 (Fig. 18).

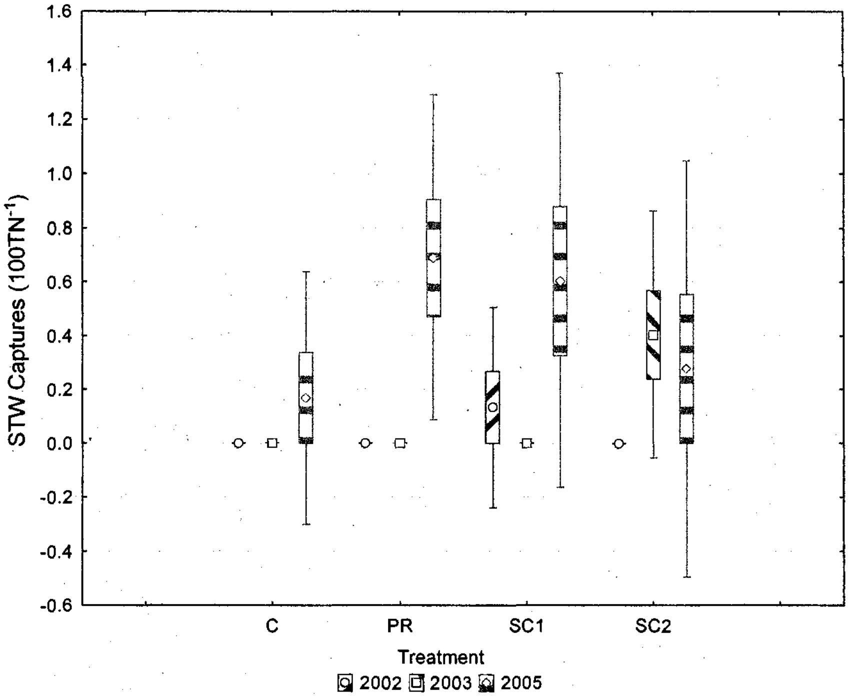

Figure 18. Crowdis Mountain study area short-tailed weasel (*Mustela erminea* Linnaeus) capture rate for treatments control (C), strip-cut 1 (SC1), strip-cut 2 (SC2), and patch retention (PR) (Boxes indicate standard error and whiskers indicate 95% confidence intervals).

The regression model used to find correlations between total small-mammal captures and habitat variables was significant ($R^2 = 0.35$; $P = 0.02$) (Appendix IV). The resulting correlation matrix of habitat variables identified capture rate as having a positive relationship with year ($p < 0.01$), fern ($p < 0.01$), slash/fine debris ($p < 0.01$), CWD length ($p < 0.01$), and CWD volume ($p < 0.01$).

4.0 DISCUSSION

Pre-treatment coarse cover-type characteristics (e.g., stand age, tree height, average DBH, and basal area) in the Crowdis Mountain study area were the result of previous pre-commercial thinning of naturally regenerated balsam fir on a site-class 2, wet-boreal forest type (Ker and Bowling 1991, Carmean 1996, Sturtevant *et al.* 1997, Thompson *et al.* 2003, McCarthy and Weetman 2006, 2007). Past pre-commercial thinning reduced tree mortality caused by intra-specific competition and increased diameter growth and stand volume by increasing available growing space (Raulier *et al.* 2003). Trees expressed a high degree of taper and were evenly spaced, and large crowns greatly reduced the amount of understory vegetation.

Although stands were sprayed with insecticide, an unevenly distributed, moderate-to-low density of snags was produced from previous budworm attacks. The data showed a high proportion of small snags (< 16 cm DBH) in later stages of decay. Spraying insecticide greatly reduced tree mortality, when compared with other areas of the highland plateau, but did not eliminate tree mortality completely.

Volume of downed CWD was low owing to high decomposition rates in wet boreal forests (Thompson *et al.* 2003) and low CWD volume left on site after the clear-cut harvest that initiated the stand. However, CWD decay-class and diameter distributions showed downed dead wood in the earliest stage of decay, similar in diameter to the mean diameter of live balsam-fir stems. This indicated that in the Crowdis Mountain study area, balsam fir stands were beginning to accumulate CWD. At 50 years old these stands showed occurrences of butt-rot and wind-throw susceptibility increased as a result (Burns and Honkala 1990). Severe weather events (e.g. heavy wind and snow-loading) may have also contributed to the increases in CWD.

Pre-treatment in 2002, small-mammal capture rates were low and highly variable among experimental units, and there were no captures in five of the 20 2002 trapping grids. Total captures of small mammals per 100 trap-nights were low when compared with other studies (Bayne and Hobson 1998, Pasitschniak-Arts and Messier 1998, Simon *et al.* 1998, Potvin *et al.* 1999, Gliwicz and Glowacka 2000, Manning *et al.* 2001, and Bowman *et al.* 2001). Coincidentally, CWD volume was also low and because of high crown closure and stocking, understory vegetation was comprised of vegetation types (i.e., mosses and lichens) that provide limited cover to small mammals (Boonstra and Krebs 2006). The Crowdis Mountain study area was considered habitat for marten because it met coarse cover-type criteria. However, the data in 2002 confirmed suspicions of low small-mammal abundance and reduced structural complexity in the form of CWD (NSAMRT 2006).

Statistical testing showed that the harvesting treatments implemented in the fall of 2002 significantly reduced the mean basal area in each treatment unit. As expected, mean age, canopy height, and DBH in the strip-cut-1and strip-cut-2 treatments were not significantly altered by harvesting and basal area remained above 20 m^2/ha in these treatments for the duration of the study. The patch-retention treatment had significantly lower mean ages, canopy height, and DBH because approximately half of the fixed-area plots were in clear-cut patches and half were in the residual patches. The mean produced by combining measured variables in clear-cut and residual patches did not accurately describe the overall stand condition in the 12.6 ha patch retention treatment units.

These findings suggest that the strip-cut treatments applied in the Crowdis Mountain study area maintained minimum coarse cover-type attributes of marten habitat (Naylor *et al.* 1994, Sturtevant *et al.* 1996, Payer and Harrison 1999(a), 1999(b), NSAMRT 2002, Poole *et al.* 2004) but the patch retention treatments did not. However, once the vegetation in clear-cut areas of the patch retention treatments reach a height > 6 m, cover and subsequently habitat value may be improved if prey (e.g., hare) are also available and structural requirements (e.g., CWD) are sufficient. Responses were measured one and three years after treatment implementation and so the coarse cover-type data simply produced short-term-response habitat features. Marten habitat is determined by a range of factors that can be but is not always well represented by coarse cover-type delineation. To identify productive marten habitat accurately, fine-filter criteria (understory cover, size and distribution of CWD, and prey base) must be included in the selection process (Sturtevant *et al.* 1996, Payer and Harrison 1999a, 1999b, Poole *et al.* 2004).

No differences in the density of snags among treatments or among measurement years were found. However, when the snag data from all the permanent sample plots were combined, decreases in smaller snags (< 16 cm diameter) in late stages of decay and increases in diameter-classes 20-24 cm in early stages of decay were found post-treatment. Smaller snags in later stages of decay were decreasing because they were falling to the ground and snags between 20 and 24 cm diameter in early stages of decay were more recent wind-thrown trees, often affected by butt rot, and/or were along harvest-block edges. Little change in the snag diameter and decay-class relationship was observed two years later in 2005. The rate of snag recruitment between 2002 and 2003 was much greater than measured two years later in 2005 and this suggest that snag recruitment was influenced by the edge created by the harvest treatments.

A significant difference in the volume of CWD between treatments and between measurement years was not detected. However, combining the permanent sample plots revealed increases in wind-felled trees between diameter-classes 18-28 cm and decay-classes 1-4 in 2003 and 2005. Natural dynamics of CWD decay and accumulation patterns in stands initiated by fire or severe insect infestations follow a general "U-shaped" temporal pattern between stand-initiating events (Lang 1985). CWD inputs from pre-disturbance stand conditions decrease with time since disturbance (Sturtevant *et al.* 1997) and as stands senesce, post-disturbance CWD inputs increase from non-stand-replacing events (e.g., insect, wind-throw, and disease) (Brassard and Chen 2006).

In the Crowdis Mountain study area, residual CWD produced by stand initiation was almost completely absent and the inventoried downed dead wood was comprised of fallen snags, created during budworm outbreaks, and more recent wind-throw. The total CWD volume was much lower than reported by Sturtevant *et al.* (1997) for stands of similar age, initiated by fire or budworm. Bissonette *et al.* (1997) and Payer and Harrison (1999a) reported that 29.7 m^2/ha of downed CWD was above limiting thresholds for marten. CWD in the Crowdis Mountain study area was 23-52 m^3/ha which is less than the volumes reported in balsam-fir stands of similar age in Newfoundland (Sturtevant *et al.* 1997, Thompson *et al.* 2003) and other forests in Nova Scotia (McCurdy and Stewart 2005). CWD volume was also < 18-20% of the total stand volume reported by Linder *et al.* (1997), Sippola *et al.* (1998), Kuuluvainen *et al.* (1998), Siitonen *et al.* (2000), and Karjalainen and Kuuluvainen (2002) to be necessary for maintaining ecological function.

In the absence of fire and insect disturbances in wet boreal forests comprised of balsam fir, wind is generally the dominant disturbance (Taylor and MacLean 2005, McCarthy and Weetman 2006). A combination of wind and butt rot is driving gap dynamics in the study area and stands are beginning to accumulate large CWD (> 20 cm diameter). Burns and Honkala (1990) reported that balsam fir is highly susceptible to a variety of root and butt rots and can show greater than 50% infection rate by approximately 70 years, greatly increasing susceptibility to wind damage. The data in this study showed that the increases in post-disturbance or post-stand-initiation CWD volume in decay classes 1 through 4 were due to wind. Wind-felled trees in early stages of decay were detected pre-treatment and although it was not directly measured, observations suggested that the rate of recruitment was accelerated by the increases in forest edge created by silviculture treatments (Raulier *et al.* 2003).

Small-mammal capture rates were low when compared to other studies (Von Trebra *et al.* 1998, NSDNR 2001, Fuller *et al.* 2004) and this was attributed to poor understory cover and food availability due to low CWD volume and low herbaceous plant cover (Bowman *et al.* 1999, 2000, Chambers 2002, Fuller *et al.* 2004, Boonstra and Krebs 2006). Previous trapping studies conducted by the NSDNR also reported low capture success and suggested that low small-mammal capture rates were due to low amounts of understory structure in the form of CWD and herbaceous plant cover (NSDNR 2001).

Capture rates did not significantly differ among treatments or among measurement years. The replicate trapping grids in each treatment type showed a high amount of variation, especially in the control grids, masking any direct treatment effect and indicating that small mammals were unevenly distributed over the study area. The control and strip-cut-1 treatments had consecutively higher total small-mammal captures from 2002 to 2005. Captures in the strip-cut-1 treatment were significantly higher in 2005 than pre-treatment in 2002. However, because they were not higher than found in the control plots, it could not be confirmed that this increase was a direct result of the treatment. As found in the control plots, increases in capture rate could have been caused by natural fluctuations in small-mammal populations.

The patch-retention and the strip-cut-2 treatments showed similar patterns of small-mammal captures: increasing initially after treatment and decreasing two years later in 2005. Coincidentally, these areas had the greatest volumes of CWD and the highest level of basal area removal. In the patch-retention treatment, small-mammal captures increased significantly after harvest in 2003, to the highest mean value of any treatment in all years of study. In 2005, the capture rate decreased but continued to be significantly higher than found pre-treatment. It is plausible that small mammals were responding to increases in cover, provided by slash and fine debris and food availability, provided by balsam fir seeds now closer to the ground. In 2005, vegetation < 2 m and CWD was more abundant but capture rates were lower. In 2005, cover by slash and fine debris would have decreased because of needle drop and the additional seed source found initially after harvesting would have diminished. Regardless, capture rate responded positively to the patch-retention treatments, contrary to what has been reported by West *et al.* (1980) and Cockle and Richardson (2003).

Red-backed voles are positively correlated with dense understory cover, windthrow (Pauli *et al.* 2006) and large CWD (Christian and Hanowski 1996, Bowman *et* *al*. 2000). Red-backed voles were the most abundant small-mammal species captured in the Crowdis Mountain study area but understory cover and CWD volumes were low. The highest capture rates were found in the controls in 2005 (6.3 captures 100 TN^{-1}) and the patch-retention treatment in 2003 (5.2 captures 100 TN^{-1}). These findings suggest that red-backed vole abundance was on the rise from initial measurements in 2002, but higher capture rates could not be attributed to treatments and were possibly a function of natural variability in localized populations within treatment units. The low capture rate expressed across all sampling areas, was comparable to reports by the NSDNR (2001) and was attributed to the lack of understory vegetation, cover, and possibly predation from other carnivores.

The increasing trends from 2002 to 2005 may be explained by observed increases of windthrow, influenced by tree age, diameter, and occurrences of butt rot. Red-backed voles have been reported to show resilience to timber harvesting once ground vegetation is established (Kirkland 1990, Witt 2001, Etcheverry *et al.* 2005) and was reported to respond positively to partial canopy removal (Simon *et al.* 2002, Fuller *et al.* 2004). Similar to what was reported by Scott *et al.* (1982), Medin and Booth (1989), Monthey and Soutiere (1985), and Christian and Hanowski (1996), mean red-backed vole captures in the Crowdis Mountain study area did not show a positive or negative response to canopy removal. These findings are supported by Sekgororoane and Dilworth (1995) who reported that small mammals did not respond to overstory removal 0-5 years after harvesting.

Captures were greatest in 2005 for all treatments except the patch-retention, but a direct correlation with specific treatment type was not possible. As the capture rate increased the variation also increased, indicating that higher occurrences were site-specific and not related to the treatment implemented. Therefore, it was determined that red-backed voles were resilient to the timber harvest treatments implemented in the Crowdis Mountain study area.

Masked-shrew captures were also very low and differences among treatments were not detected except for the patch-retention treatment in 2003. Because mean captures were consistently higher in 2003 across all treatment types, this pulse was attributed to increased food availability, possibly a pulse of insect abundance during the summer and fall of 2003. Although not as abundant as masked shrews, short-tailed shrews also displayed a similar pulse of abundance in 2003. Shrews specifically require humid microenvironments (Miller and Getz 1977) and understory cover for protection from predators (Nordyke and Buskirk 1991). Insect density is positively associated with dense herbaceous vegetation and stands in the Crowdis Mountain study area may be lacking sufficient cover, and thus insects to support high shrew populations.

Captures of deer mice and short-tailed shrews were very low ($< 1\ 100\ \mathrm{TN}^{-1}$) for the duration of the study and no treatment response was detected. Deer-mouse capture rates were most consistent in the patch-retention treatment, the only treatment to have captures of deer mice in all three sampling years. Deer mice are described as habitat generalists and may have been present in the patch-retention treatments more consistently because of the added food availability and cover produced from timber harvesting. In 2005, the regenerating clear-cuts had greater than 60% coverage of wild red raspberry (*Rubus idaeus* Var. *strigosus*) and the added cover from logging residue may have increased invertebrate populations (Monthey and Soutiere 1985). Because of trap dimensions, short-tailed weasels were not a targeted small-mammal species. Catches were incidental and were most likely the result of the weasels chasing a targeted species into the trap. Regardless, it was interesting to see that in 2005, weasels were present in all treatment types, possibly a response to increases in prey availability.

This study does not support the suggestions by Sullivan *et al.* (1999), Cockle and Richardson (2003) and Simard and Fryxell (2003) that partial harvesting has a negative effect on small-mammal assemblages. Significant increases were not detected even though capture rates were greatest post-treatment. The data suggest that in the Crowdis Mountain study area, canopy removal did not negatively affect small-mammal capture rates, supported by West *et al.* (1980), Scott *et al.* (1982), and Von Trebra *et al.* (1998) who also found no change in small-mammal assemblages after partial or complete canopy removal.

Prior to this study, the balsam-fir stands in the Crowdis Mountain study area had been clear-cut, pre-commercially thinned, and sprayed with insecticide. When compared with what has been reported in other studies, they have low levels of downed CWD (Thompson *et al.* 2003, McCurdy and Stewart 2005) and low abundances of small mammals (Von Trebra *et al.* 1998, NSDNR 2003, Fuller *et al.* 2004). Pre-commercial thinning removed overhead cover, important to snowshoe hare during early stages of succession (Beaudoin *et al.* 2004), but I doubt that it greatly reduced the volume of CWD. Natural stem exclusion prevented by pre-commercial thinning is not a significant source of CWD until stem diameters exceed 7.5 cm (Stevens 1997) which can take over 30 years in un-thinned balsam fir stands (Penner *et al.* 2006). Also, because of pre-commercial thinning, post-disturbance wind-thrown logs in decay classes 1 and 2 were larger than would have been present if pre-commercial thinning had not occurred. Spraying insecticide should have greatly reduced the level of mortality caused by budworm and SDDW was detected only at moderate to low levels, in unevenly distributed clumps. However, after approximately 50 years since stand initiation or when tree diameters exceed 20 cm in balsam-fir forests (Burns and Honkala 1990), non-stand-replacing disturbances (i.e., wind and pathogens) increase CWD volume as stands senesce (Sturtevant *et al.* 1997). From these observations, I conclude that the Crowdis Mountain study area did not require special management practices to enhance future habitat availability for small mammals or marten, if excluded from future timber harvesting.

Unfortunately, solving the problem of not having enough marten habitat available in the Cape Breton Highlands is not as simple as just terminating timber harvesting from the landscape. The stands contain merchantable timber that is included in overall wood-supply analysis and expected cost-recovery of past silviculture (i.e., pre-commercial thinning and spraying) is integral to the success of the local forest industry. Therefore, putting a timber-harvesting moratorium on stands that resemble conditions of this study area would not only have negative effects on the forest industry, but would also do little in maintaining healthy populations of marten if not incorporated into a larger, long-term, landscape-level strategy of habitat retention and restoration.

The primary concern with respect to stand conditions resulting from management practices of clear-cut harvesting, pre-commercial thinning, and insecticide application is the lack of or reduction of CWD volume and understory cover in younger stages of stand development. To reiterate, Potvin *et al.* (1999) stated that, with regard to maintaining habitat for marten and small mammals, "what is done is less important than what is left or untouched". Maintaining stand characteristics that provide productive habitat for marten (e.g., large diameter snags, > 30 m^3/ha CWD, > 18 m^2/ha basal area) starts at the stand-initiation stage (i.e., harvesting). Clear-cut harvesting produces stand conditions that have lower coarse cover-type values than the minimum required by marten. However, when stands reach 6 m in height, cover requirements are restored. If the harvesting process decreases SDDW and consequently small mammals, then viable marten habitat is still not available until stands began to accumulate SDDW as a result of natural gap dynamics (approximately 50 years in the Crowdis Mountain study area). Short rotations (< 60 years) of clear-cut harvesting have the potential to reduce SDDW and prey abundances below minimum productive levels for marten if adequate amounts of SDDW (> 30 m^3/ha) are not left after each harvesting event.

The study was designed to look at the effects of variable-retention harvesting on habitat requirements of marten, specifically SDDW and small mammals. Essentially, the study initiators wanted to know if the treatments designed could be implemented economically while maintaining the critical habitat features required by marten. The study showed that the tested strip-cut treatments can achieve this goal. Strip-cut treatments were economical yet maintained minimum levels of coarse cover-type criteria required by marten and small mammals. Cost analysis conducted by the Forest Engineering Research Institute of Canada (FERIC) calculated that the ENVIRO® single-grip processor cost $96/productive machine hour in the strip-cut treatments (Meek and Pentassuglia 2003). Felling costs were 8.88 $/$m^3$ and production was 10.8 m^3 per productive machine hour. Though more expensive than conventional clear-cutting, this harvesting option can cost-effectively remove timber while maintaining basal area and promoting understory development and SDDW accumulation.

Based on the information gathered in this study, I believe that the strip-cut harvesting systems designed by StoraEnso are a cost-effective way to harvest while maintaining minimum requirements of marten habitat. Treatments can be implemented in two stages using a processor and forwarder combination. The first stage of harvesting should be conducted as soon as trees are merchantable for pulp or saw-log production, which will vary depending on stand composition and whether pre-commercial thinning was implemented. Raulier *et al.* (2003) reported that balsam fir does not significantly increase in annual increment as a result of commercial-thinning operations. Balsam fir is also highly shade-tolerant, and natural thinning of 7-m-tall, 60-year-old stands was reported by McCarthy and Weetman (2007). More information is required on the relationship between increased annual increments and reductions in fibre quality, but to date, given the length of time it takes for balsam fir to thin naturally, I recommend pre-commercial thinning as a viable silviculture option preceding strip-cut implementation. The initial harvest should maintain a minimum of 20 m^2/ha basal area and slash and logging debris should be left on site and manipulated accordingly to reduce soil disturbances in extraction trials.

The second stage of harvesting would be determined by the growth rate and species composition in the clear-cut harvested "strips". Once this vegetation exceeds minimum height requirements for marten (6 m), the leave "strips" from the initial harvest can then be clear-cut. Depending on how the stand responds to the initial harvest (e.g., excessive wind-throw) it may be required to implement the second stage of harvesting to offset significant volume losses before vegetation in the initial harvest strips reach 6 m. The timing of this two-stage harvesting system and inclusion in an overall landscape-level habitat matrix will be critical for its effectiveness in maintaining marten habitat characteristics. The average stand DBH during the initial harvest will determine wind-throw susceptibility (i.e., SDDW recruitment) and the length of time between harvesting stages will determine overall marten habitat suitability.

In this study, stand composition, SDDW, and small mammals were quantified and conclusions were made regarding habitat responses and resulting habitat quality without monitoring any actual marten. The next logical step would be to test the assumptions of this study by monitoring the response of reintroduced marten to confirm the effectiveness of habitat manipulation. Further investigation into SDDW spatial dynamics (degree of clumping) and specific habitat function of small mammals is also required. Marten and small mammals both require SDDW but each may require it at different spatial scales to maintain specific ecological function. More abundant, highly clumped, small diameter, late decay-class, CWD, found close to the ground, may prove more important to small mammals than marten that require less abundant, large, elevated, early decay-class, CWD for subnivean access to prey. The relationship between strip-cut widths and edge effects (blow-down and understory vegetation) also should be investigated to address wind-throw susceptibility and consequent volume losses resulting from modified harvesting treatments. Further prey-base analyses should include more than just small mammals to describe overall food availability for marten.

5.0 CONCLUSIONS

Forest management practices focused on increased fibre production and yield were responsible for the stand conditions measured in the Crowdis Mountain study area. Stands contained high basal area because of past forest management. However, understory vegetation and SDDW were low and consequently so was small mammal abundance.

Snag density was highly variable, indicating that snags were not evenly distributed over the sampling area. Decayed snags resulting from past budworm outbreaks were detected in congregated groups and more recent wind-felled trees were detected throughout the study sites pre-treatment. Although not specific to treatment type, sampling from 2002 to 2005 showed increases in large snags in early stages of decay and decreases in snags in late stages of decay because of wind action. Statistical testing confirmed that harvesting treatments did not have a significant effect on density of snags but may have increased stand susceptibility to wind-throw by increasing edge.

CWD was also highly variable across the study site and was low when compared with balsam fir stands initiated by fire or insects. Volumes of CWD were not significantly different among treatments pre- or post-harvest but increases in large diameter CWD (> 18 cm) were detected over the entire study area post-treatment and these increases were attributed to wind damage. Treatments increased the amount of forest edge, which is known to cause increases in windthrow but a significant relationship between treatments and CWD volume could not be established. Therefore, it was concluded that treatments did not have a significant effect on CWD volume. However, observed increases in wind-thrown trees along harvested edges were not captured by the sampling design which may have changed the final results.

Small-mammal capture rates continued to be low post-harvest and were highly variable among treatment types and years of sampling. Treatments did not have a significant effect on small-mammal capture rates when compared to the control plots. However, higher abundances post-treatment in 2003 and 2005 were found and these increases in small mammals were positively correlated with percent coverage of fern, slash/fine debris and CWD volume found across the entire study area.

This study showed that variable-retention harvesting did not have a negative effect on small mammals, snag abundance, CWD, or understory vegetation, and can be used in pre-commercially thinned balsam-fir forests to extract timber economically while maintaining minimum habitat requirements of marten.

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7.0 APPENDICES

7.1 APPENDIX I: LITERATURE REVIEW

MARTEN ECOLOGY GENERAL

Description

American marten are small to medium-sized carnivorous mammals weighing between 500 g and 1400 g as adults (Banfield 1974, Buskirk and Ruggiero 1994). The average weight across North American range for adult males and females is 900 g and 600 g respectively. Males are larger than females but are similar in appearance. They have a fox-like pointed face, short limbs, a slender body and torso between 35 cm and 43 cm, and a long bushy tail between 18 cm and 23 cm (Banfield 1974). Fur is medium length, glossy, solid tan to dark brown in colour with an irregular pale cream to bright amber throat patch (Buskirk and Powell 1994). Marten do not depend on subcutaneous fat reserves for maintaining endothermy: rather, highly active behaviour and continuous feeding are their adaptations to cold temperatures. They also require well-insulated denning sites to prevent elevated rates of heat loss in sub-zero conditions (Katnik 1992).

A member of the weasel family Mustelidae, the marten is one of seven species in the genus *Martes* with fourteen subspecies distributed globally (Buskirk and Powell 1994, Car and Hicks 1997). Marten communicate intra-specifically by producing chemical signals from anal sacs, abdominal and plantar glands, but have relatively weak scent glands compared to other Mustelidae (Strickland and Douglas 1987). Skull features are distinguishable from other North American mustelids by the presence of four upper and lower premolars (Burt 1980). Marten are agile climbers with semi-retractable claws, using vertical and horizontal structure to travel and hunt in the forest canopy and along the ground. Studies have reported wild animals as old as 12 years and farmed marten of 22 years (Buskirk and Powell 1994, Buskirk and Ruggiero 1994).

Distribution

Possibly arriving to North America during glaciation some 10000 BP via the Bering Land Bridge, marten are widely distributed and range west from the southern Sierra Nevadas of California and Vancouver Island to the east on the island of Newfoundland (Scott 2001). Found south in northern New Mexico to the tree-line in arctic Alaska and Canada, the majority of their distribution is now in temperate to sub-arctic boreal and taiga zones (Burt 1980, Buskirk and Ruggiero 1994, Thompson 1994). It has been suggested that in North America, two distinct species of marten exist, *Martes caurina* Merriam 1890, and *Martes americana* (Car and Hicks 1997). Taxonomic status of *Martes* is not fully resolved because of issues regarding sub-species designation made possible by an increase in genetic assessment capabilities. *M. caurina* in southwestern ranges is believed to be a sub-species of more commonly found *M. americana* but distinct maternal lineages have not been confirmed (NSAMRT 2002).

The distributions of North American populations have undergone expansions and contractions and have increased in insularity. Once common in coastal ranges of northern California, they are now scarce and limited to western mountain ranges that provide remaining habitat. Marten are considered endangered in New Mexico and are under special management and protection measures in various states and provinces across North America. Extirpated from seven U.S. states where healthy populations once existed, interest in using marten as indicator species in mature forests has increased (Fuller and Harrison 2005).

Concern for marten populations in boreal and Acadian forest regions of Canada is growing, especially throughout the Atlantic Provinces where the greatest declines have occurred (Scott 2001). The Newfoundland population of *M. americana*, believed to be the sub-species *M. americana atrata* (Forsey *et al.* 1995, Kyle and Strobeck 2003) was listed as endangered in 1998. Marten have been listed as endangered in Nova Scotia since 2001 (Scott 2001).

CAPE BRETON MARTEN POPULATION

General

Marten have been detected in the southeastern highlands of Victoria County and in the northwestern highlands, near the southern boundary of Cape Breton Highlands National Park (Banks 1992, Scott 2001). Historical timber-harvest activities combined with recent insect infestations and subsequent salvage logging have fragmented the landscape between these populations (Brander 1994, NSAMRT 2002). Detection efforts by Nocera *et al.* (1999) reported low capture rates and the absence of marten in areas where there is suitable habitat, indicating that populations are critically low. However, due to the limitations of detection methods, especially when population numbers are so low, density estimates can be highly variable (Zielinski and Kucera 1995, Scott and Hebda 2004).

Genetics and Population Recovery

Genetic analysis has determined that the Cape Breton Island marten (*M. americana americana*) is the same sub-species as mainland populations and re-introductions may be a possible recovery option (NSAMRT 2006). Relocations have been successful as a conservation tool in mainland Nova Scotia and in Maine (Scott and Hebda 2004). Research regarding sufficient translocation numbers and release sites with adequate quantity and quality of vacant habitat is ongoing (Nocera *et al.* 1999, NSAMRT 2006).

MARTEN HABITAT REQUIREMENTS

Marten are associated with continuous, conifer-dominated, late- successional, mesic forests (Banfield 1974, Soutiere 1979, Buskirk and Ruggiero 1994, Thompson and Colgan 1994). Marten select older, conifer-dominated stand types because they have greater prey abundance, basal area, tree diameter, and volume of CWD than non-selected stand types (NSAMRT 2002, Poole *et al.* 2004). These characteristics provide maternal denning sites (cavities greater than 40 cm), elevated resting and winter thermoregulation sites, subnivean access to prey, and cover from predators in the form of tall trees and high canopy closures (Naylor *et al.* 1994).

Sturtevant *et al.* (1996) suggested that complex physical structure and overhead cover are more important to marten than stand age. Payer and Harrison (1999a) and Poole *et al.* (2004) concluded that complex physical structure is more important to marten than overstory coverage, age, or species composition. Studies by Chapin *et al.* (1997), Sturtevant *et al.* (1997), Potvin *et al.* (2000) and Payer and Harrison (2000a) reported that marten use mixedwood stands and younger age classes when structural characteristics of late-succession forests were present.

Payer and Harrison (1999a, 1999b, 2000b) found that marten selected mature softwood, hardwood, and mixedwood stands > 9 m tall. The highest marten habitat indices are recorded in forests containing downed dead wood with a mean diameter of 22 cm with some 40 cm pieces (Naylor *et al.* 1994, Brown *et al.* 2003). Fuller and Harrison (2005) suggested retaining 18 m^2/ha basal area and canopy closure greater than 30%. Potvin *et al.* (1999) stated that "what is done is less important than what is left or untouched". Thus, if dense herbaceous or shrubby regenerative vegetation is established and if key structural characteristics, like CWD, are maintained, younger age classes may provide marten with cover and maintain prey populations (Payer and Harrison 2000b, 2003).

PREY BASE

Prey base is a useful indicator of ecological condition and directly influences marten population dynamics (age structure, ovulation rates, pregnancy rates, and home range size) (Buskirk and Ruggiero 1994). Marten are dietary generalists and feed on small mammals (mice, shrews, squirrels and voles), hare, fish, birds, eggs, carrion, insects, berries, and fruits (Buskirk and Ruggiero 1994). Diet varies depending on geographic location, seasonal availability, abundance, and caloric requirements (Buskirk and Ruggiero 1994, Poole *et al.* 2004).

Small mammals are a basic component in many food chains and their abundance can regulate breeding success of terrestrial and avian predators (Bayne and Hobson 1998, Parker 1989). Marten abundance often follows small-mammal abundance, with population changes sometimes offset, as in typical predator-prey relationships (Thompson and Colgan 1987, Hodges *et al.* 1999). The coincidence of their populations suggests that small mammals are important determinants of marten fitness (Buskirk and Powell 1994, Payer and Harrison 2003).

Several mammal species are consumed by marten as they forage along the ground or surface of snow, investigating access points to subnivean space created by CWD (snags, logs, stumps, broken tree tops, root masses) and other forms of structure (rocky slopes, ground vegetation, live branches and limbs) (Naylor *et al.* 1994, NSAMRT 2002). Forage species include: snowshoe hare (*Lepus americanus* Erxleben), red-backed voles (*Myodes gapperi* Vigors), deer mice (*Peromyscus maniculatus* Wagner), meadow voles (*Microtus pennsylvanicus* Ord), woodland jumping mice (*Napaeozapus insignis* Miller), masked shrews (*Sorex cinereus* Kerr), short-tailed shrews (*Blarina brevicauda* Say), ground squirrels (*Spermophilus* F. Cuvier), red squirrels (*Tamiasciurus hudsonicus* Erxleben), and northern flying squirrels (*Glaucomys sabrinus* Shaw)

Diet variety and analysis by Poole *et al.* (2004) suggest that marten have an increased reliance on Cricetidae (voles, lemmings and mice), Soricidae (shrews), and Sciuridae (squirrels) during seasonal and cyclic variations in hare populations. Snowshoe hare is a major prey species, especially in winter, and are selected in proportion to availability (Fuller and Harrison 2005). Hare inhabit a wide variety of sites but generally prefer 10-20 year-old dense coniferous forests (Hodges *et al.* 1999, Poole *et al.* 2004). Hare have been reported absent in clear-cut areas until sufficient cover is established and increase home range and movement in response to increased overstory removal (Payer and Harrison 1999b, 2000a, Potvin *et al.* 2000).

Squirrels are abundant in mid- to late-successional conifer and mixedwood cone-producing forests and have an important ecological relationship with marten. Squirrels are hunted in the canopy and their nests and middens are used by marten as resting sites (Buskirk and Ruggiero 1994). Deer mice have been reported to be a minor diet item in areas where a high selection of prey is available but are the primary prey source in areas such as Vancouver Island, where species variety is limited (Buskirk and Ruggiero 1994). Red-backed voles are a key diet species of marten and are selected in relation to availability or local abundance (Buskirk and Ruggiero 1994, Poole *et al.* 2004).

COARSE WOODY DEBRIS (CWD)

CWD cycles contribute significantly to the capital pool of nutrients found in forested ecosystems and is one of the primary energy sources for complex food webs (Bartels *et al.* 1985, Butler *et al.* 2002). Essential to wildlife habitat and nutrient acquisition influencing long term site productivity, CWD dynamics is an issue of growing importance to foresters, biologists, and land managers (Bunnell *et al.* 2002a). Defined as SDDW in various stages of decay, including snags, root masses, stumps, limbs, and logs (Bartels *et al.* 1985, Neitro *et al.* 1985, Stevens 1997), CWD contributes to slope stabilization, mineral cycling, and nitrogen fixation (Koenigs *et al.* 2002, Machmer 2002). CWD also provides food and habitat for small mammals, birds, amphibians, worms, insects, asymbiotic mychorrhizal fungi, and nitrogen-fixing bacteria (Naylor *et al.* 1994, Bellhouse and Naylor 1996, Lindgren and MacIsaac 2002). Standing and downed CWD is used for building nests, dens, and burrows and provides hiding cover for predators, protective cover for prey, and travel corridors between the canopy and the forest floor (Chambers 2002).

CWD cycles derived from chronosequence studies display classic u-shaped pattern of accumulation between stand-replacing events in natural fire regimes (Brassard and Chen 2006). Volume of CWD in a stand over time is correlated with two important phases of forest development; building (living) and deconstruction (decaying) (Stevens

1997). As stands mature from the initiation stage, CWD volume, comprised of pre-disturbance and post-disturbance stem mortality, transform from standing dead wood to downed logs and stumps in various stages of decay. Decay causes CWD volume to decrease over time until density-dependent intra-specific competition (self-thinning) occurs between larger diameter stems (Sturtevant *et al.* 1997).

Self-thinning occurs once stands reach maximum carrying capacity but does not contribute significantly to CWD volume until tree diameters exceed 7.5 cm (Stevens 1997). Influenced by moisture and temperature, small downed woody debris (< 7.5 cm) decays rapidly, adding organic matter and providing a short-term influx of nutrients. Stevens (1997) reported that CWD > 7.5 cm diameter will decay more gradually and provide longer term nutrient release and greater carbon storage. Stevens (1997) also stated that a wide range of diameters and decay-classes are required not only for long-term site productivity but for various ecological functions.

Suppression-induced stem-exclusion rates for snags > 9 cm in douglas fir (*Pseudotsuga menziesii* (Mirb) Franco) stands in western Oregon were greatest between age 20 and 40 years (Neitro *et al.* 1985). According to density management diagrams of balsam fir stands in New Brunswick, it can take over 40 years for competition-induced mortality to produce snags greater than 7.5 cm diameter (Penner *et al.* 2006). As stands reach maturity, competition-induced mortality rate decreases (Neitro *et al.* 1985) yet volume of CWD continues to increases. Stem exclusion rates become influenced by stand age and increased susceptibility to non-stand- replacing events including insects, pathogens, and weather (wind, ice, snow). Diameter and decay-class distributions of snags and logs broaden through repeated cycles of disturbance and succession (Brassard and Chen 2006).

Minimum diameter designation for standing and downed woody debris within a forest stand is based on species composition, diameter class, and specific ecological function (Sturtevant *et al.* 1997). Snags have been defined by Watt and Caceres (1999) as standing dead, dying, or defective trees, greater than 10 cm in diameter, greater than 3 m tall, with cavities or potential to develop cavities. Downed woody debris includes coarse roots, stumps, limbs and logs (inclining > 45° off vertical) and is considered > 7.5 cm in diameter (Stevens 1997). Influenced by a wide variety of site specific conditions (geography, stand composition, age, disturbance regime and natural events), SDDW provides continuity of habitat and structural linkages to previous stand conditions in natural disturbance regimes (Stevens 1997). Harvesting and silvicultural treatments, when combined with landscape management strategies of fire and insect suppression, have the potential to produce CWD-deficient, second- and third-rotation forest if appropriate CWD management considerations are not defined (Sturtevant *et al.* 1996).

Weight and volume calculations quantifying CWD have been used to describe local site conditions. Maine forests were reported to have sufficient levels of CWD to support marten and small-mammal communities in stands > 12 m tall. Payer and Harrison (2000a) suggested maintaining snag volume > 10 m^3/ha for marten. Pedlar *et al.* (2002) described CWD volume in mixedwood stands and deciduous stands in northern Ontario as 160 m^3/ha and 105 m^3/ha respectively, much higher than in pure conifer stands with average volumes of 18 m^3/ha. Stevens (1997) reported a range of weights between 50 and 113 Mg (10^6) of CWD with input rates ranging between 2.4 and 7.0 m^3/ha/year. Average volume of 29.7 m^3/ha were reported across a number of study areas where CWD was above suggested minimum thresholds for marten and small mammals (NSAMRT 2002). These values are useful when the focus is on broader landscape variables such as carbon or fuel loading but because of influence from a multitude of site-specific conditions (species, type (snag, log or stump), stage of decay, diameter, length, orientation, degree of clumping, etc.), characterization is imperative to understanding specific ecological function (Bunnell *et al.* 2002b, McCay *et al.* 2002).

SMALL MAMMALS

Small mammals respond positively to increases in plant diversity, food, moisture, and cover associated with decaying logs (Bellhouse and Naylor 1996). Brown *et al.* (2003) and Pasitschiak-Arts and Messier (1998) suggested that 15-20% ground coverage of CWD is required for healthy populations of small mammals. Bellhouse and Naylor (1996) stated that > 20% ground cover of downed woody debris was required to maintain marten and small-mammal populations. Bowman *et al.* (2001) found a significant relationship between abundance of small mammals and CWD in later stages of decay. The process of decomposition increases nutrient release to plants through increases in mycorrhizal associations and increases the log's ability to retain water. Plants use decomposing CWD as sites for germination and downed dead wood attracts insects (Thomas 2002) that break down the log and increase access for roots and fungi (Bellhouse and Naylor 1996). Keisker (2000) found a positive association with small mammals and the lower range of CWD diameter classes but stated that this relationship was most likely due to an increased percentage of ground cover provided by more-abundant small CWD. Many studies about small-mammal assemblages have been conducted where CWD volume was above limiting thresholds for small mammals and characterization of CWD size and decay distributions is lacking (Stevens 1997).

Small mammals are important to forested ecosystems because they are a primary prey source for various other mammals and birds (Ream 1981); they help control invertebrates (Gaines and McClenaghan 1980) and densities of other small mammals (Lautenschlager *et al.* 1997), and disperse seeds and mycorrhizal fungal spores (Stevens 1997, Brassard and Chen 2006). Small mammals co-occur and interact through a myriad of direct and indirect pathways and the presence and abundance of some species influence those of others (Brooks *et al.* 1998). However, often the short duration of studies limit conclusions regarding species-specific interactions (Morris 2005).

Small mammals inhabit a wide variety of habitats over the range of individual species (Bowman *et al.* 2001) and have been described as coarse-grained foragers and fine-scale descriptors of vegetative cover (Morris 1987, Simon *et al.* 2002). Home range, a function of habitat quality (Bondrup-Nielsen 1986) is generally < 1 ha (Hansson 1996, Potvin *et al.* 1999) and population density, which can fluctuate considerably (Delany 1975, Fryxell *et al.* 1997), influences dispersal (Gaines and McClenaghan 1980) and is best determined by macrohabitat variables (Orrock *et al.* 2000, Morris 2005).

Clough (1987) described the importance of litter, ground vegetation, and shrub layers to small mammals. Parker (1989) found that small-mammal abundance was positively correlated with invertebrate abundance, which was also positively correlated with ground cover from slash, fine debris, and low ground vegetation such as raspberry (*Rubus idaeus* Var. *strigosus* Michx.). Short-tailed shrews inhabit a wide variety of habitats but are more common in dense forests with deep litter that is not dry (Peterson

1966). They feed on invertebrates and some plant material, are non-cyclic, have a home range of 0.2-0.4 ha, and have densities as high as 62 ha^{-1} (Bole 1939). Masked shrews also inhabit a wide variety of habitats but require a mat of vegetation for cover and high abundance of moisture near the ground (Bole 1939, Fox 1983). It has been suggested that masked shrews are positively associated with CWD because of the added moisture retention it provides (Ream 1981). Deer mice have been described as habitat generalists (Chambers 2002) and are reported to respond positively to early seral stages after fire or clear-cutting (Ream 1981, Fox 1983). They feed on seeds and vegetation, are cyclic, have a home range of 0.2 - 1.2 ha, and densities have been reported as high as 25 -37 ha^{-1}.

Red-backed voles normally inhabit later successional conifer, deciduous, and mixedwood forests at moderate elevations (Fisher 1968, Merritt 1981) and are positively associated with dense herbaceous understory plant cover, CWD (Peterson 1966, Pauli *et al.* 2006, Orrock *et al.* 2000) and other forms of structural heterogeneity (Naylor *et al.* 1994, Sullivan *et al.* 1999). Red-backed voles feed on the fruiting bodies of hypogenous ectomycorrhiza-forming fungi (DeGraf and Rudis 1987), lichen, foliage, seeds, roots, bark, and insects (Bartels *et al.* 1985, Sekgororoane and Dilworth 1995, Simon *et al.* 1998, Sullivan *et al.* 1998). Populations are non-cyclic (Finerty 1945, Bondrup-Nielsen 1986, Fryxell *et al.* 1997) and have been reported to fluctuate wildly (Bayne and Hobson 1998, Hannon *et al.* 2002, Boonstra and Krebs 2006) at an average density of 25 ha^{-1} (Bondrup-Neilsen 1986, Pauli *et al.* 2006).

The response of small mammals to canopy removal is dependent on various factors. Small mammals may be temporarily displaced by natural disturbances such as fire but populations re-colonize once understory ground cover and complex physical structure close to the ground is restored (West *et al.* 1980, Parker 1989). This statement is supported by findings by Simon *et al.* (2002) who reported that CWD and grass/herb vegetative cover explained > 50% of the variation between small-mammal abundances in clear-cut and naturally burned areas. Payer and Harrison (2000a) studied small mammals in budworm-affected areas and clear-cut harvested stands and found similar assemblages in each disturbance type. However, Cockle and Richardson (2003) found canopy removal by clear-cutting to reduce species diversity and have a negative effect on small-mammal fitness by increasing disease and parasites.

West *et al.* (1980) studied the northern red-backed vole (*Myodes rutilus* Pallas) and found no difference in abundance between harvested and unharvested treatment areas but captures were lower in clear-cuts. Scott *et al.* (1982) found no change in small-mammal assemblages after timber harvesting and stated that slash left on site was responsible for the lack of response. Potvin *et al.* (1999) reported the relative abundance of deer mice doubling and red-backed voles showing no response after canopy removal from clear-cutting. Sullivan *et al.* (1999) described Kirkland's (1990) review of 21 studies on small mammals and reported two studies stating increases and four studies reporting decreases in red-backed voles after clear-cutting.

West *et al.* (1980) found similar populations of small mammals between shelterwood and unharvested areas. Poole *et al.* (2004) reported red-backed voles decreasing in areas less than five years after overstory removal through shearing while deer mice and meadow voles increased. Medin and Booth (1989) found no difference in red-backed vole abundance between partially harvested and unharvested areas. Monthey and Soutiere (1985) reported greater abundance of red-backed voles in both partially cut stands (50% removal) and regenerating clear-cuts than in uncut stands. Medin and Booth (1989) stated that red-backed voles decreased when basal area was reduced from 20.7 to 5.1 m^3/ha but that no response was found when 1/3 of stand basal area was removed. Von Trebra *et al.* (1998) found that 30-50% removal of the forest canopy did not negatively affect red-backed-vole or deer-mouse populations. Steventon *et al.* (1998) reported a higher abundance of red-backed voles in areas with light timber extraction than in un-cut forests in west central British Columbia. Christian and Hanowski (1996) found that strip-cutting had no effect on small mammals yet Simard and Fryxell (2003) found a negative response of small mammals to selective logging because of the reduction in food availability (seed production).

Pasitschiak-Arts and Messier (1998) stated that vegetation along harvested edges supports a richer diversity of small mammals. Sekgororoane and Dilworth (1995) found that small mammals did not respond to forest edge 0-5 years after disturbance but there was an effect 6-10 years after harvest. Bayne and Hobson (1998) found that deer mice responded to edge but that red-back voles did not. Hannon *et al.* (2002) did not find a difference in populations between cut and riparian buffers. A study by Kingston and Morris (2000) also found no effect of edge on small-mammal populations, contrary to Cockle and Richardson (2003) who found a positive response.

Limited data exist regarding small-mammal assemblages and abundances in the Cape Breton Highlands of Nova Scotia (NSAMRT 2002). A trapping survey conducted by the Nova Scotia Department of Natural Resources (NSDNR) in conifer, deciduous, and mixedwood stands across Cape Breton Island in fall of 2001 resulted in the capture of seven species. Red-backed voles, masked shrews, deer mice, and short-tailed shrews (listed in order of decreasing abundance) were trapped and collected. Single captures of a woodland jumping mouse, meadow vole, and northern flying squirrel were also recorded.

The greatest relative abundance found by the NSDNR trapping session was in conifer and mixedwood stands (3.5-8.33 captures/100 trap nights). Captures of red-backed voles and deer mice were higher than reported in northern boreal forests of Quebec (Potvin *et al.* 1999) and Labrador (Simon *et al.* 1998) but were lower than reported in the southern boreal forest of Saskatchewan (Bayne and Hobson 1998, Pasitschniak-Arts and Messier 1998) and summarised by Gliwicz and Glowacka (2000) for various areas in North America, including northern BC (Manning *et al.* 2001) and northern NB (Bowman *et al.* 2001). High variability among sampling areas, sampling methods, and experimental designs limit comparability of small-mammal assemblages and abundances among different studies (Boonstra and Krebs 2006). However, comparing captures of small mammals in Cape Breton with those experienced by other studies illustrates that small-mammal assemblages in Cape Breton are within an expected range by species and habitat type.

7.2 APPENDIX II: PRE-TREATMENT ANOVA TABLES

Crowdis Mountain study area analysis of variance (ANOVA) for age measured in 2002.

Source	Df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	583.225	194.408	1.9718	**0.1589**
Plot Number (PN)	16	1577.53	98.5958	1.2265	0.2617
Expt. Error	100	100	8038.83		
(Corrected) Total	119	10199.6			
Overall mean	1	282949			
Raw total	150	293148.6			

Crowdis Mountain study area analysis of variance (ANOVA) for height measured in 2002.

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	71.9925	23.9975	2.6776	**0.0821**
Plot Number (PN)	16	143.398	8.96241	4.5979	≤ 0.0001
Expt. Error	100	194.925	1.94925		
(Corrected) Total	119	410.316			
Overall mean	1	16213			
Raw total	150	16623.316			

Crowdis Mountain study area analysis of variance (ANOVA) for diameter breast height (dbh) measured in 2002.

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	106.029	35.3431	2.0454	**0.1480**
Plot Number (PN)	16	276.474	17.2796	1.9724	0.0221
Expt. Error	100	876.066	8.76066		
(Corrected) Total	119	1258.57			
Overall mean	1	50803.1			
Raw total	150	52061.67			

Crowdis Mountain study area analysis of variance (ANOVA) for total basal area measured in 2002.

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	1979.87	659.956	2.051	**0.1472**
Plot Number (PN)	16	5148.27	321.767	2.8301	0.0008
Expt. Error	100	11369.3	113.693		
(Corrected) Total	119	18497.5			
Overall mean	1	140631			
Raw total	150	159128.5			

Crowdis Mountain study area analysis of variance (ANOVA) for standing and downed dead wood (SDDW) measured in 2002.

Source	Df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	256.833	85.6112	2.8271	**0.0717**
Plot Number (PN)	16	484.523	30.2827	2.4496	0.0036
Expt. Error	100	1236.23	12.3623		
(Corrected) Total	119	1977.59			
Overall mean	1	3570.66			
Raw total	150	5548.25			

Crowdis Mountain study area analysis of variance (ANOVA) for small-mammal capture measured in 2002.

Source	Df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	9.70013	3.23338	1.4116	**0.2759**
Plot Number (PN)	16	36.6481	2.2905		No test
(Corrected) Total	19	46.3482			
Overall mean	1	47.933			
Raw total	20	94.2814			

7.3 APPENDIX III: POST-TREATMENT ANCOVA TABLES

Crowdis Mountain study area analysis of covariance (ANCOVA) for basal area in 2003 and 2005 using 2002 data as a covariate (* significant at the 0.05 level).

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	10271.6	3423.85	21.421	***≤ 0.0001**
Plot Number (PN)	16	2557.41	159.838	2.6357	0.0009
2002 Covariate (BC)	1	11407.1	11407.1	188.1	***≤ 0.0001**
Year (Yr)	1	2.81667	2.81667	2.5414	**0.1305**
T * Yr	3	3.11667	1.03889	0.93734	**0.4456**
PN * Yr	16	17.7333	1.10833	0.018276	1.0000
Expt. Error	199	12068.3	60.6445		
(Corrected) Total	239	55585.2			
Overall mean	1	153723			
Raw Total	240	209308.2			

Bonferroni post hoc test results showing differences in basal area between treatment-types in the Crowdis Mountain study area (* significant at the 0.05 level).

Treatments	Difference	Standard Error	Prob (>F)
SC1 – C	-9.79211	2.408	*0.0053813
SC2 – C	-15.9032	2.396	*≤ 0.0001
SC2 – SC1	-6.11108	2.309	0.100928
PR – C	-18.0895	2.484	*≤ 0.0001
PR – SC1	-8.29737	2.32	*0.0150142
PR – SC2	-2.18629	2.324	0.931864

Crowdis Mountain study area analysis of covariance (ANCOVA) for height in 2003 and 2005 using 2002 data as a covariate (* significant at the 0.05 level).

Source	Df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	743.976	247.992	13.936	*≤ **0.0001**
Plot Number (PN)	16	284.711	17.7944	2.7535	0.0005
2002 Covariate (HC)	1	222.222	222.222	34.387	*≤ **0.0001**
Year (Yr)	1	0.000462963	0.000462963	1	**0.3322**
T * Yr	3	0.00138889	0.000462963	1	**0.4182**
PN * Yr	16	0.00740741	0.000462963	0.000071639	1.0000
Expt. error	199	1286.02	6.46243		
(Corrected) Total	239	3237.2			
Overall mean	1	24769.4			
Raw Total	240	28006.6			

Bonferroni post hoc test results showing differences in height between treatment-types in the Crowdis Mountain study area (* significant at the 0.05 level).

Treatments	Difference	Standard Error	Prob (>F)
SC1 – C	-0.887667	0.8519	0.894775
SC2 – C	-1.04601	0.8582	0.808167
SC2 – SC1	-0.158341	0.7703	0.999983
PR – C	-4.81321	0.868	*0.000266062
PR – SC1	-3.92554	0.771	*0.000653126
PR – SC2	-3.7672	0.7705	*0.000981479

Crowdis Mountain study area analysis of covariance (ANCOVA) for diameter in 2003 and 2005 using 2002 data as a covariate (* significant at the 0.05 level).

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	2123.21	707.738	6.0019	***0.0061**
Plot Number (PN)	16	1886.72	117.92	3.9529	≤ 0.0001
2002 Covariate (DC)	1	881.013	881.013	29.534	***≤ 0.0001**
Year (Yr)	1	0.0560185	0.0560185	0.51102	**0.4850**
T * Yr	3	0.168056	0.0560185	0.51102	**0.6804**
PN * Yr	16	1.75394	0.109621	0.0036747	1.0000
Expt. error	199	5936.36	29.8309		
(Corrected) Total	239	11320.1			
Overall mean	1	89745.3			
Raw Total	240	101065.4			

Bonferroni post hoc test results showing differences in diameter height between treatment-types in the Crowdis Mountain study area (* significant at the 0.05 level).

Treatments	Difference	Standard Error	Prob (>F)
SC1 – C	1.16576	1.983	0.993206
SC2 – C	1.01221	2.033	0.997237
SC2 – SC1	-0.153551	2.026	1
PR – C	-6.21724	2.057	*0.0475238
PR – SC1	-7.383	2.048	*0.0141553
PR – SC2	-7.22945	1.985	*0.0130933

Crowdis Mountain study area analysis of covariance (ANCOVA) for age in 2003 and 2005 using 2002 data as a covariate (* significant at the 0.05 level).

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	17415.8	5805.28	16.892	***≤ 0.0001**
Plot Number (PN)	16	5498.78	343.674	1.7285	0.0439
2002 Covariate (AC)	1	9791.41	9791.41	49.246	***≤ 0.0001**
Year (Yr)	1	54.15	54.15	2.428	**0.1387**
T * Yr	3	52.35	17.45	0.78244	**0.5210**
PN * Yr	16	356.833	22.3021	0.11217	1.0000
Expt. error	199	39566.3	198.825		
(Corrected) Total	239	71420.3	239		
Overall mean	1	484202			
Raw Total	240	555622.3			

Bonferroni post hoc test results showing differences in age between treatment-types in the Crowdis Mountain study area (* significant at the 0.05 level).

Treatments	Difference	Standard Error	Prob (>F)
SC1 – C	3.32712	3.389	0.917969
SC2 – C	1.66086	3.412	0.997556
SC2 – SC1	-1.66626	3.437	0.997613
PR – C	-18.215	3.443	*0.000439743
PR – SC1	-21.5421	3.479	*≤ 0.0001
PR – SC2	-19.8759	3.391	*0.000144519

Crowdis Mountain study area analysis of covariance (ANCOVA) of small-mammal captures in 2003 and 2005 using 2002 data as a covariate (* significant at the 0.05 level).

Source	df	SS	MS	MS-ratio	Prob (>F)
Treatment (T)	3	36.2845	12.0948	1.3883	**0.2825**
Plot Number (PN)	16	139.391	8.71197		No test
Year (Yr)	1	2.04734	2.04734	0.14277	**0.7105**
T * Yr	3	148.796	49.5987	3.4587	***0.0415**
PN * Yr	16	229.441	14.3401		
(Corrected) Total	39	555.961			
Overall mean	1	1327.94			
Raw Total	40	1883.901			

7.4 Appendix IV: Regression Tables

Test of sum of squares whole model versus sum of squares residual (SM treatment response) in the Crowdis Mountain study area.

Dependent Variable	Adjusted R^2	SS Model	Df Model	MS Model	SS Residual	Df Res	MS Res	F	P
Caps/100TN	0.353	549.17	28	19.613	282.953	31	9.13	2.148	0.02

Univariate Tests of Significance for Caps/100TN (SMtreatmentResponse) in the Crowdis Mountain study area (Sigma-restricted parameterization Effective hypothesis decomposition).

Source	Df	SS	MS	MS-ratio	Prob (>F)
Intercept	0.577	1	0.577	0.063	0.803
Height	1.121	1	1.121	0.122	0.728
Diameter	0.044	1	0.043	0.004	0.945
Age	11.079	1	11.079	1.213	0.279
Basal Area	1.639	1	1.639	0.179	0.674
Moss/lichen	0.118	1	0.118	0.012	0.910
Low Shrubs	0.308	1	0.308	0.033	0.855
Fern	1.049	1	1.049	0.114	0.362
Sedges/ grasses	0.064	1	0.064	0.007	0.934
ClubMoss	12.625	1	12.625	1.383	0.248
Herbs	0.054	1	0.054	0.005	0.939
Slash/fine debris	36.615	1	36.615	4.011	**0.045**
Mineral soil	5.937	1	5.937	0.650	0.426
Leaf Litter	4.688	1	4.688	0.513	0.478
CWD Length	16.667	1	16.667	1.826	**0.186**
CWD Diameter	5.679	1	5.679	0.622	0.436
CWD Volume	0.178	1	0.178	0.019	0.889
Snag Density	11.489	1	11.489	1.258	0.270
Treatment	18.304	3	6.101	0.668	0.577
Year	3.493	2	1.746	0.191	0.826
Treatment* Year	171.059	6	28.509	3.123	**0.016**
Error	282.954	31	9.127		

Correlations between small-mammal captures and measured habitat variables (significant at $p < 0.05$, N=60 (Casewise deletion of missing data)).

Variable	Year	Fern	Slash/ fine debris	CWD Length	CWD Volume
Captures /100TN	0.4305	0.3615	0.453	0.4022	0.4234
	P=0.001	P=0.005	P=0.0009	P=0.001	P=0.001