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Woody debris structures on large clearcut openings: Oases for small mustelids and prey species?

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ABSTRACT

Restoration structures such as post-harvest woody debris piles on large clearcut openings may provide habitat for many mammal species. Mustelids such as the American marten (*Martes americana*) and several weasel (*Mustela* spp.) species are important furbearers in temperate and boreal forests and are negatively affected by clearcutting. We ask if constructed piles of woody debris on large (e.g., 30–50 ha) openings will be used by these mustelids and their small mammal prey species? We tested the hypotheses (H) that (H_1) the presence (index of activity patterns) of small mustelids, (H_2) abundance, species richness, and species diversity of the forest-floor small mammal community, and (H_3) reproduction of the major species: southern red-backed vole (*Clethrionomys gapperi*), long-tailed vole (*Microtus longicaudus*), and deer mouse (*Peromyscus maniculatus*), would be greater on sites with woody debris piles than on sites with dispersed debris. A fourth hypothesis (H_4) predicted that the above response variables would be similar across a linear configuration of debris piles on the large openings. Mustelid presence and populations of forest-floor small mammals were sampled on four replicated ($n = 4$) large (30–50 ha) clearcut sites from 2017 to 2022 in south-central British Columbia, Canada.

Mean presence of mustelids was significantly higher (4.5 to 8.0 times) in piles than dispersed sites during the first four post-harvest years, thereby supporting H_1. Mean index of mustelid activity per line was similar among pile index-lines across these clearcut openings. Mean abundance of *C. gapperi* was similar statistically between treatment sites but with numbers 3 to 10 times higher, on average, in the piles than dispersed sites and was likely biologically important. Mean abundance of *M. longicaudus* was significantly higher (1.2–4.4 times) in the piles than dispersed sites. Mean abundance of *P. maniculatus* and *Sorex* spp. were similar between treatment sites. Mean abundance of total small mammals, including the less common species, was significantly higher (1.4–1.9 times) in piles than dispersed sites. Similarly, mean species richness and diversity were both significantly higher in piles than dispersed sites. Mean number of total recruits was higher in piles than dispersed sites for *M. longicaudus* and *P. maniculatus*, but most measures of reproductive attributes were similar, thereby providing only partial support for H_3. As per support for H_4, response variables were similar across a linear array of debris piles on large openings. Our study is the first to measure responses of small mustelids and forest-floor small mammals to constructed piles of woody debris as a means of habitat restoration on relatively large (mean area 40.5 ha) openings created by conventional clearcutting. Piles of debris seem to act as oases for mustelids and their prey species in ecological restoration of cutover forest land.

1. Introduction

Forest biomass is a major component of the renewable energy sector designed to help reduce greenhouse gas (GHG) emissions from fossil fuel use and modulate climate change (IPCC, 2014; Titus et al., 2021). Bioenergy feedstocks from forestry will likely be a driving factor in meeting requirements for greater proportions of energy from renewable sources over the next several decades (Thiffault et al., 2015; Bacovsky et al., 2016; Petrokofsky et al., 2021). The primary feedstocks removed from forests as biomass include residues such as tops, branches, and foliage left after harvesting merchantable trees, sub-merchantable stems or whole trees left after commercial harvesting, and salvage wood from natural disturbances of wildfire, insects, diseases, and drought (Lindenmayer et al., 2008; Titus et al., 2021).

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Although wood residues may be used as renewable biomass feedstocks, this use needs to be balanced with the major roles of woody debris in forest ecosystem function, long-term site productivity, and biodiversity. These functions include nutrient cycling, contribution of organic matter to soil structure, and modification of micro-climate (Harmon et al., 1986; McComb and Lindenmayer, 1999; Laiho and Prescott, 2004). In particular, woody debris provides crucial habitat for a wide variety of forest mammal species in terms of foraging, resting, reproduction, and various forms of structural cover (security and thermal) (McComb, 2003; Maser et al., 2008; Bunnell and Houde, 2010).

A recent *meta*-analysis on the effects of harvesting forest-based biomass on terrestrial wildlife concluded that there were no consistent negative effects on biodiversity (Homyack and Verschuyl, 2019). However, in general, there have been few experimental studies on management of woody debris, either as biomass removals, or as restoration structures for mammals in harvested forest ecosystems (Seibold et al., 2015). A major exception has been the construction of woody debris structures (e.g., piles and windrows) built from post-harvest residues. This practice has generated some vital mammalian biodiversity networks for small mustelids, small mammal prey species, and other mammalian carnivores, at least on relatively small clearcut openings (Sullivan et al., 2012, 2017a; Seip et al., 2018).

Restoration structures are particularly relevant on clearcut openings where many mammal species have declined in abundance or disappeared owing to a loss of various components of forest stand structure (Fisher and Wilkinson, 2005). Furbearers such as the American marten (*Martes americana*), short-tailed weasel (*Mustela richardsonii*) (Colella et al. 2021), and long-tailed weasel (*Neogale frenata*) (Patterson et al. 2021) are negatively affected by the large openings created by clearcutting as they may be prey for other carnivores (Hargis et al., 1999; Buskirk and Zielinski, 2003; Linnell et al., 2017a; Lavoie et al., 2019). Marten occupy forested landscapes of older (>80 years) conifer-dominated stands with relatively high levels of canopy closure, substantial levels of coarse woody debris on the forest-floor, and riparian ecosystems (Buskirk and Powell, 1994; Roloff et al., 2020; Farnell et al., 2020). The two weasel species are broadly distributed in various forest successional stages, edge habitats, and riparian woodlands where dense understory vegetation provides cover for them and habitats for small mammal prey (Simms, 1979; King et al., 2007; Evans and Mortelliti, 2022). Both weasel species also occur in open habitats such as clearcuts, assuming that there is sufficient structural cover from avian predators and available food (Buskirk and Zielinski, 2003; Evans and Mortelliti, 2022).

Major prey species for small mustelids are the southern red-backed vole (*Clethrionomys gapperi*) that occupies old forests and disappears for many decades after clearcutting (Martin, 1994; Zwolak, 2009; Linnell et al., 2017b) and *Microtus* voles that may be common on some clearcuts and other successional stages post-harvest (Simms, 1979; Buskirk and Zielinski, 2003; Wilk and Raphael, 2017). Both *C. gapperi* and *Microtus* may undergo dramatic fluctuations in abundance over many years and decades (Krebs, 2013; Sullivan et al., 2017b). Other prey species include the generalist deer mouse (*Peromyscus maniculatus*), northwestern chipmunk (*Neotamias amoenus*), and *Sorex* shrews that also occur on recent clearcuts.

Regenerated forests take many decades to provide sufficient habitat for furbearer species and their prey. A continuing problem is the contiguous large openings that accumulate over time. These openings are typically 30 to 50 ha in size from conventional clearcut harvesting but may be 100 s to 1000 s of ha if salvage harvesting of insect- and wildfire-damaged stands occur (Lindenmayer et al., 2008). Habitat restoration is much needed in these large openings to provide food and cover during the many decades of forest recovery. As noted, although woody debris structures have provided habitat for small mustelids and prey species on relatively small (5 – 10 ha) clearcut openings, it is not known if debris structures placed across large (>30 ha) openings will be used by these species. Thus, we ask if constructed piles of woody debris on large (e.g., 30 – 50 ha) openings created by conventional clearcutting will be used by small mustelids and their small mammal prey species? We tested the hypotheses (H) that (H_1) the presence (index of activity patterns) of small mustelids, (H_2) abundance, species richness, and species diversity of the forest-floor small mammal community, and (H_3) reproduction of the major species: *C. gapperi*, long-tailed vole (*M. longicaudus*), and *P. maniculatus*, would be greater on sites with woody debris piles than on sites with dispersed debris. A fourth hypothesis (H_4) predicted that the above response variables would be similar across a linear configuration of debris piles on the large openings.

2. Methods

2.1. Study areas

Two study areas were located in south-central British Columbia (BC), Canada: (i) Donald ($51^0$29′07″N; $117^0$05′46″W) and Dart Creek ($51^0$18′42″N; $116^0$54′09″W) replicate sites at 35 km northwest and 7 km east, respectively, of Golden, and (ii) Munro West ($49^0$42′21″N; $119^0$57′01″W) and Munro East ($49^0$42′38″N; $119^0$53′53″W) replicate sites 32 and 35 km west of Summerland, respectively. The Donald site is in the Interior Cedar-Hemlock (ICH_{mk}; m,k = moderate precipitation regime, cool temperature regime) biogeoclimatic subzone with topography ranging from hilly to steep terrain at 1100–1200 m elevation in the lower ranges of the Rocky Mountains. Upland coniferous forests dominate the ICH landscape and comprise the highest diversity of tree species of any zone in BC. Western red cedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) dominate mature climax forests with Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), lodgepole pine (*Pinus contorta* var. *latifolia*), white spruce (*Picea glauca*), Engelmann spruce (*Picea engelmannii*), their hybrids, and subalpine fir (*Abies lasiocarpa*) common in these stands (Meidinger and Pojar, 1991).

The Dart Creek site is in the Montane Spruce (MS_{dk}; d,k = dry precipitation regime, cool temperature regime) and the two Munro sites in the (MS_{dm}; d,m = dry precipitation regime, mild temperature regime) biogeoclimatic subzones. Topography ranged from hilly to steep terrain at 1100–1210 m elevation at Dart Creek to rolling hills at 1450–1520 m elevation at the two Munro sites. The MS has a cool, continental climate with cold winters and moderately short, warm summers. Mean annual precipitation ranges from 30 to 90 cm. There are extensive even-aged post-fire lodgepole pine stands which have regenerated after wildfire. Hybrid interior spruce (*Picea glauca* × *P. engelmannii*) and subalpine fir are the dominant shade-tolerant climax trees. Trembling aspen (*Populus tremuloides*) is a common seral species and black cottonwood (*Populus trichocarpa*) occurs on some moist sites (Meidinger and Pojar, 1991). Average ages of lodgepole pine ranged from 80 to 120 years and for Douglas-fir and other conifers ranged from 120 to 220 years. Overstory mean tree heights ranged from 22 to 26 m at Summerland and from 25 to 32 m at Golden over all conifer species. Other than the conventional piling of post-harvest debris, there were no site preparation treatments on any of these harvested sites, prior to planting a mixture of lodgepole pine, Douglas-fir, and interior spruce seedlings in the year after harvest.

2.2. Experimental design

Four replicate blocks with a mean (±SE) area of 40.5 ± 5.6 ha (range 28.2 to 50.2 ha) were chosen from the areas clearcut in 2016–2018. Blocks were separated by a mean distance of 42 km at Golden and 3.3 km at Summerland. A randomized complete block design had the following two treatments: (a) dispersed post-harvest debris with no vertical structure > 0.3 m (Fig. 1a), and (b) a linear configuration of woody debris piles (Fig. 1b) installed across the long axis of each clearcut unit (Fig. 1c). The 8 sites (2 treatments × 4 replicates) were selected with respect to operational scale, harvest sites that were the size of current forestry operations, and reasonable proximity of sites to one

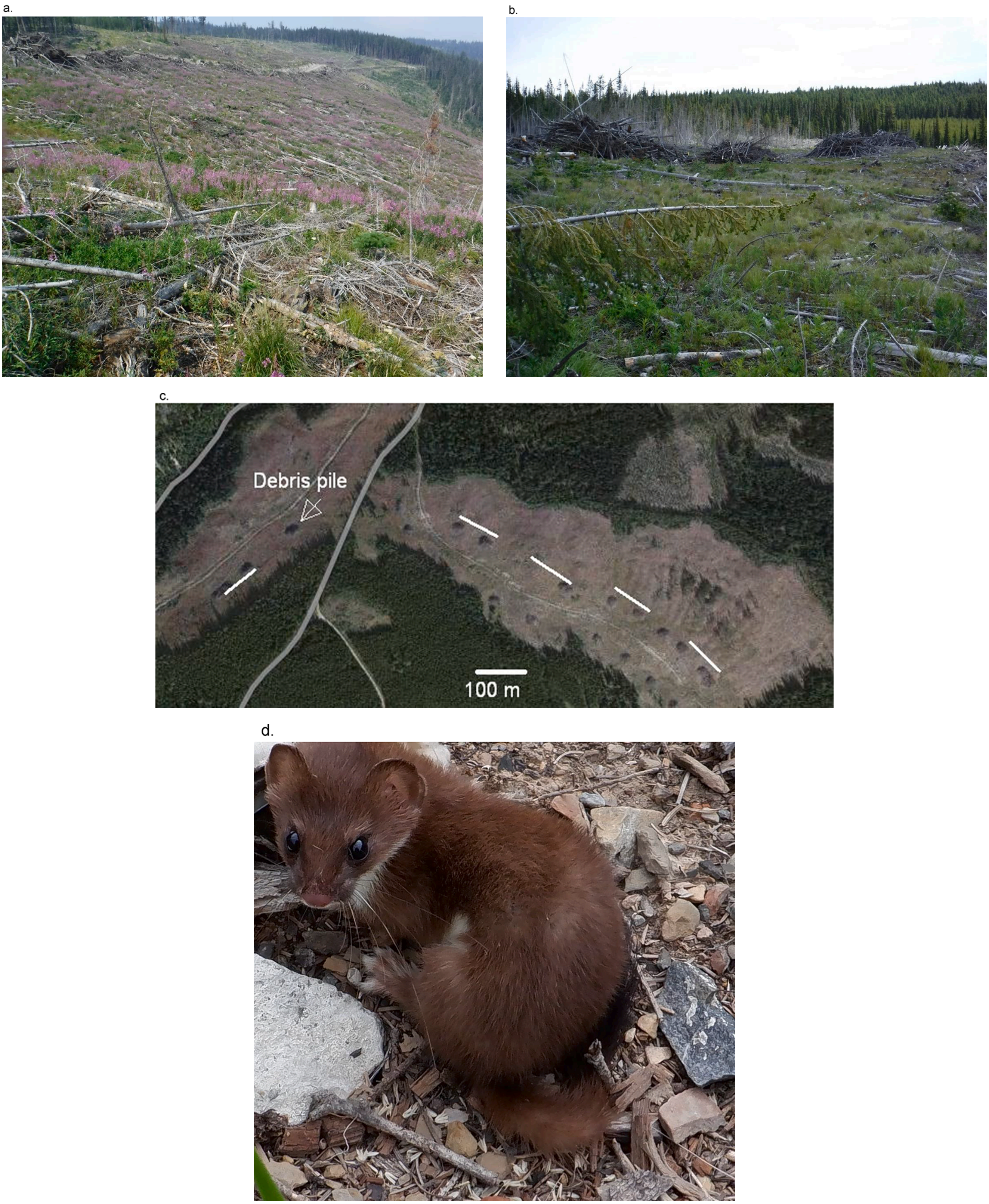

Fig. 1. Photographs of post-harvest woody debris treatment sites at the Summerland study area in south-central British Columbia, Canada: (a) dispersed, (b) piles, (c) linear configuration of piles (index lines as white bars) across a large clearcut unit (Google Imagery © 2023 CNES/Airbus, Maxar Technologies, Province of British Columbia, Map data © 2023), and (d) short-tailed weasel in August 2021, 4 years after harvest and construction of piles.

another within each of the Golden and Summerland areas. The Dart Creek site was terminated after two years (2019–2020) because of highway construction and was replaced by another nearby site for 2021–2022. This latter site was chosen for accessibility and continuity of mammal measurements after clearcut harvesting and was located 10 km northwest of Golden, thereby maintaining the independence of treatment blocks.

The dispersed debris and piles treatment sites within a block were reasonably separated to enhance statistical and biological independence (Hurlbert, 1984): a mean (±SE) of 0.14 ± 0.02 km (range 0.10 – 0.50 km). Movements of individual animals were calculated between and within the set of respective index-lines (see below) that comprised the dispersed and debris pile treatment sites. For the major small mammal species, a measure of this independence was that only 40 of 1036 (3.9%) individual *C. gapperi*, 15 of 1521 (1.0%) individual *M. longicaudus*, and 179 of 1513 (11.8%) individual *P. maniculatus* were captured on more than one index-line. Thus, index-lines that comprised the dispersed and debris pile treatment sites were also considered to be independent units. Treatment sites and index-lines were not considered independent for marten, weasels, or *N. amoenus*.

2.3. Woody debris treatments

Clearcut harvesting and subsequent woody debris treatments were installed in the harvest year where piles were created during processing of cut timber, followed by some specific site preparation work with an excavator (Table 1). Piles were composed of tops, branches, and bole ends of harvested trees, as well as trees knocked down during harvest, low-quality commercial trees, dead wood, and non-commercial trees left at the harvest site. The largest material retained in structures averaged 30 to 50 cm in diameter. Piles were located near landings or access roads where log processing occurred. Volumes of downed wood (≥1 cm in diameter) in the dispersed treatments were measured using the line-intersect method of Van Wagner (1968) in three plots, each of which was an equilateral triangle with 20 m sides. Location of plots was randomly chosen along the small mammal index-line on each site. Volumes of woody debris in piles were measured by the method of Hardy (1996), first estimating the volume of a pile and then using a biomass ratio (or bulk density factor) of 0.67 to estimate the net volume of woody biomass, after the method discussed in Sullivan et al. (2011).

Table 1
Mean ± SE dimensions of woody debris treatments at each of the four sites at the Golden and Summerland study areas, southern British Columbia, Canada. Each of the *dispersed* and *piles* treatments had five index-lines for measuring woody debris, presence of mustelids, and small mammal prey species.

Treatment and parameter	Donald	Dart Creek	West Munro	East Munro
Dispersed (n = 5 lines)				
Mean volume per line (m^3/ha)	171.3 ± 19.7	129.7 ± 7.8	222.3 ± 39.1	225.6 ± 24.0
Total volume (m^3) per site	8565	6511	6269	7603
Piles (n = 5 lines)				
Mean volume per line (m^3)	687.2 ± 167.3	1050.8 ± 487.1	1757.6 ± 128.8	971.5 ± 70.2
Total volume (m^3) per site	3436	5254	8788	4857
Number of piles per index-line	3.0 ± 0.3	4.2 ± 0.4	3.0 ± 0.5	6.4 ± 0.8
Mean height (m)	3.2 ± 0.1	2.8 ± 0.4	4.1 ± 0.3	3.0 ± 0.1
Mean diameter (m)	13.2 ± 2.5	12.9 ± 2.8	19.8 ± 1.6	10.0 ± 0.5
Total length of linear configuration of piles (m)	795.3	933.2	1075.2	962.6
Mean distance between piles in index-lines (m)	42.1 ± 1.9	51.8 ± 8.6	56.6 ± 4.1	24.6 ± 1.3
Mean distance between pile index-lines (m)	90.6 ± 41.3	64.5 ± 12.7	127.2 ± 52.4	62.1 ± 13.7

2.4. Forest-floor small mammals

Forest-floor small mammals were sampled at 4-week intervals from May or June to October 2016 to 2022 as per the sample years for each replicate block. Each of the two treatments within a given block had five 100-m index-lines for sampling small mammal prey species (Pearson and Ruggiero, 2003; Sullivan and Sullivan, 2010). Each index-line had seven trap stations at 14.3-m intervals with four Longworth live-traps at each station. Individual index-lines were installed in dispersed debris and in sets of debris piles and arranged linearly across the long axis of each clearcut block (Fig. 1c). Location of index-lines were at comparable distances (0.10 – 0.30 km, on average) to the edge of each respective clearcut block. Edge habitats were either mature/old-growth forest or 20- to 30-year-old second-growth forest.

Traps were supplied with whole oats, a slice of carrot, and cotton as bedding. Each trap had a 30-cm × 30-cm plywood cover for protection from sunlight (heat) and precipitation. Traps were set on the afternoon of day 1 and checked on the mornings of day 2 and day 3, and then locked open between trapping periods. All animals captured were ear-tagged with serially numbered tags, breeding condition noted, weighed on Pesola spring balances, and point of capture recorded. Breeding condition was noted by palpation of male testes and the condition of mammaries of the females (Krebs, 2013). Animals were released on the index-lines immediately after processing. Unfortunately, the overnight trapping technique resulted in a high mortality rate for shrews. Therefore, shrews were collected, frozen, and later identified according to tooth patterns (Nagorsen, 1996). All handling of animals followed guidelines approved by the American Society of Mammalogists (Sikes et al., 2016) and the Animal Care Committee, University of British Columbia.

2.5. Demographic analysis

Abundance estimates of the red-backed vole, long-tailed vole, and deer mouse were derived from the Jolly-Seber (J-S) stochastic model for open populations with small sample size corrections (Seber, 1982; Krebs, 1999). Minimum number alive was used to estimate populations of the meadow vole (*M. pennsylvanicus*) and heather vole (*Phenacomys intermedius*); number of individuals was used for the montane shrew (*S. monticolus*) and common shrew (S. *cinereus*). Number of individuals captured each trapping week was used for northwestern chipmunks owing to their occasional movements among lines. Population size for each species was based on mean number of animals per line for the five index-lines for each treatment site. Overall, we consider these estimates to be an index of population size per line (Krebs et al., 2011). Jolly trappability was calculated for the major species according to the estimate discussed by Krebs and Boonstra (1984). Species richness was the total number of species sampled for the mammal communities in each site (Krebs, 1999). Species diversity was based on the Shannon-Wiener index which is well represented in the ecological literature (Burton et al., 1992). Mean annual measurements of abundance, species richness, and species diversity of small mammals were calculated using the estimated parameter for each species or community for a given sampling period and then averaged over the number of sampling periods for each year.

For the assessment of reproduction, mass (g) at sexual maturity was used to determine age classes of *M. longicaudus*: juvenile = 1–24; adult ≥ 25 (Summerland) juvenile = 1–30; adult ≥ 31 (Golden), and *P. maniculatus*: juvenile = 1–20; adult ≥ 21. Juveniles were considered to be young animals recruited during the study. Measurements of recruitment (new animals that entered the population through reproduction and immigration), and number of successful pregnancies were derived from the sample of animals captured in each trapping session and then summed for each summer period. A pregnancy was considered successful if a female was lactating during the period following the estimated time of birth of a litter. Early juvenile productivity is an index relating recruitment of young into the trappable population to the number of lactating females (Krebs et al., 1969). A modified version of this index is number of juvenile animals at week *t* divided by the number of lactating females caught in week *t* – 4.

2.6. Presence of mustelids

The presence (index of activity) of mustelids was measured along the small mammal index-lines at each site by (a) observations, as well as live-trapping and release (marten and small weasels (Fig. 1d), (b) fecal scats on four 30 × 30 cm plywood boards used as covers at trap stations, and (c) predation disturbance of small mammals at trap sites. Captures and fecal scats were identified to marten or weasel according to (Murie, 1954; Zielinski and Kucera, 1995; British Columbia Ministry of Environment, 1998; Rezendes, 1999). One Tomahawk live-trap (Model 201, Tomahawk Live trap Company, Tomahawk, Wisconsin) equipped with a nest box (1-L plastic bottle with coarse brown cotton) was located at each of three stations. Traps were baited with strawberry jam and set in the evening on day 1 and checked in the mornings of days 2 and 3. Sampling periods and intervals were identical to those described for small mammal species. Mustelid observations and live captures (marten and weasels in Tomahawk traps and weasels in Longworth traps), fecal scats, and disturbance of live-traps were recorded during the 4–6 trapping periods each year. Thus, there were 12 to 18 cells for possible data entry each year: 4 to 6 trapping periods × 3 indicators of mustelid presence that yielded a proportional value divided by 12, 15, or 18 for each treatment site.

Predation disturbance of live-traps was readily identified to marten or weasels because of the very characteristic pattern of disturbance. Weasels disturbed an occupied Longworth trap by knocking the trap over, under the cover board, thereby opening the door and preying upon the occupant. Marten disturbed the occupied traps by breaking open the Longworth trap or rolling the Tomahawk trap and by moving the trap at least 1 m from the station. Other potential carnivores such as coyotes (*Canis latrans*) or lynx (*Lynx canadensis*) are uncommon in these habitats. Fecal scats may have been deposited at any time during the intervals between these trapping periods. All captured mustelids were identified to species and released.

2.7. Statistical analysis

A one-way analysis of variance (ANOVA) was used to detect differences among sites for dimensions of woody debris and distance between sampling lines (Zar, 1999). A repeated-measures analysis of variance (RM-ANOVA) (Statistical Programs for the Social Sciences, IBM Corp., 2022), over the first four post-harvest years where we had concurrent data for the four replicate blocks, was used to determine the effect of the two treatments on mean values for presence of mustelids, abundance of *C. gapperi*, *M. longicaudus*, *P. maniculatus*, total shrews, total small mammals, species richness, and species diversity, as well as the effects of time and treatment × time interactions. Meadow voles and heather voles had low and inconsistent sample sizes (<2/line), as did *N. amoenus* which occurred in the Summerland blocks only. These low samples precluded statistical analysis, but these species were included in the summaries of total abundance. This analysis was also conducted to detect differences in mean values for number of recruits, number of successful pregnancies, and juvenile productivity for *C. gapperi*, *M. longicaudus* and *P. maniculatus*. The measurement of mustelid presence was a combination of observations and captures, fecal scats, and predation disturbance of marten or small weasels calculated as a mean value for the five sample lines for each replicate site and year. In addition, the RM-ANOVA was used to test for differences in the presence of mustelids, mean abundance of the major species, total abundance, species richness, and species diversity of small mammals across the *linear sequence of index-lines* within the piles of woody debris treatments. The effects of time and treatment × time interactions were also tested in this analysis.

Homogeneity of variance was measured by the Levene statistic. Mauchly's *W*-test statistic was used to test for sphericity (independence of data among repeated measures) (Littel, 1989; Kuehl, 1994). For data found to be correlated among years, the Huynh-Feldt (H-F) correction was used to adjust the degrees of freedom of the within-subjects *F*-ratio (Huynh and Feldt, 1976). Proportional data were transformed by arcsin square root (Fowler et al., 1998). Overall mean values (n = 16, 4 replicate sites × 4 years) ± 95% confidence intervals (CIs) for the major species, totals, species richness, and species diversity were also calculated (Zar, 1999). In all analyses, the level of significance was at least P = 0.05.

3. Results

3.1. Woody debris treatments

The mean (±SE) distance between the five sampling lines was similar ($F_{1,3}$ = 8.08; P = 0.07) within the dispersed (164.3 ± 17.2 m) and piles (120.5 ± 11.9 m) treatment sites (Table 2). Mean volume of debris in piles was significantly ($F_{1,3}$ = 18.13; P = 0.02) higher (6.0 times) than that in the dispersed treatment. However, the mean overall volume of debris per replicate block was similar between dispersed and piles treatments, taking into account the overall number of ha of each block (Table 2). Mean (±SE) diameter and height of debris piles were 14.0 ± 2.1 m (range 10.0 – 19.8 m) and 3.3 ± 0.3 m (range 2.8 – 4.1 m), respectively. Mean volume of debris per pile and other dimensions are listed in Table 2. Mean (±SE) overall length of the linear configuration of piles across a clearcut block was 941.6 ± 57.6 m (range of 795 to 1075 m).

3.2. Presence of mustelids

Mean presence of mustelids was significantly ($F_{1,3}$ = 30.84; P = 0.01) higher (4.5 to 8.0 times) in piles than dispersed sites during the first four post-harvest years (Table 3; Fig. 2). This measure of mustelid activity was similar ($F_{3,3}$ = 2.45; P = 0.24) among the four blocks. This index of activity patterns included samples (e.g., total observations) of 184 weasels and 21 marten in piles and 34 weasels and 0 marten in dispersed debris at the four replicate sites. The composite parts of these measurements were: mustelid captures or observations (95 weasels and 3 marten), fecal scats (88 weasels and 9 marten), and disturbed traps (36

Table 2
Mean (n = 4 replicate sites) ± SE measurements for sampling lines and woody debris in dispersed and piles treatment sites, and results of univariate ANOVA. Within a row, columns of mean values with different letters are significantly different. Significant values are given in bold text.

	Treatment		Analysis	
Parameter	Dispersed	Piles	$F_{1,3}$	P
Distance between index-lines (m)	164.3 ± 17.2	120.5 ± 11.9	8.08	0.07
Volume of debris per ha or in piles (m^3)	187.2b ± 22.8	1116.8a ± 227.4	18.13	**0.02**
Overall volume of debris (m^3)	7237 ± 529	5584 ± 1137	1.06	0.38
Volume (m^3) of debris per pile	–	304.2 ± 96.2		
Number of piles per site	–	20.8 ± 4.0		
Piles per ha	–	0.6 ± 0.1		
Piles per index-line	–	4.2 ± 0.8		
Distance between piles (m)	–	39.1 ± 5.6		
Diameter (m)	–	14.0 ± 2.1		
Height (m)	–	3.3 ± 0.3		
Length of piles configuration (m)	–	941.6 ± 57.6		

Table 3
Mean annual ($n = 4$ replicate sites) ± SE presence of mustelids (based on live captures, fecal scats, and trap disturbance per sampling period and year) in the two treatment sites for the first four post-harvest years at the Summerland and Golden study areas, southern British Columbia, Canada, and results of RM-ANOVA. Significant values are given in bold text.

			RM-ANOVA					
Treatment			Treatment		Time		Treatment × time	
Year post-harvest	Dispersed	Piles	$F_{1,3}$	P	$F_{3,18}$	P	$F_{3,18}$	P
1	0.03 ± 0.03	0.24 ± 0.05	30.84	**0.01**	2.34	0.11	0.28	0.84
2	0.10 ± 0.03	0.45 ± 0.10						
3	0.08 ± 0.08	0.46 ± 0.12						
4	0.12 ± 0.04	0.56 ± 0.15						
Overall ($n = 16$)	0.08 ± 0.02	0.42 ± 0.06						

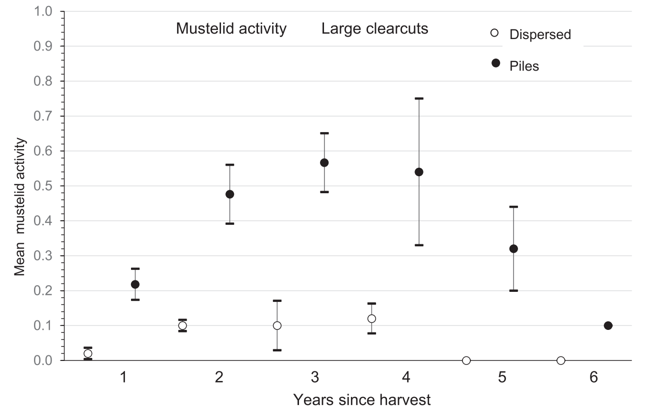

Fig. 2. Mean ($n = 4$ replicate sites) ± SE mustelid activity per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

weasels and 8 marten). Mean counts of mustelid presence per sampling period ranged from 0.24 to 0.56 in the sites with woody debris piles and from 0.03 to 0.12 in the dispersed sites (Table 3). Overall, mean (±95% C.I.s) number of observations of mustelid activity was 0.08 (0.03–0.13) in dispersed sites and 0.42 (0.30–0.55) in piles sites. Overall counts of mustelid activity for the two treatment sites per year were reasonably consistent within a treatment (dispersed, piles) for the four years post-harvest: 1st – (2, 22); 2nd – (10, 44); 3rd – (9, 50); and 4th – (11, 56). There were no significant ($P \geq 0.11$) time or treatment × time interaction effects.

Mean (±SE) index of mustelid activity per line was similar ($F_{4,12} = 2.16$; $P = 0.14$) among pile index-lines across these clearcut openings ranging from 0.12 ± 0.04 to 0.30 ± 0.04 (Table 4). Again, there were no significant ($P \geq 0.63$) time or treatment × time interaction effects. Similarly, there was no difference ($F_{3,12} = 0.89$; $P = 0.47$) in this measurement among the four blocks. Insufficient sample size of mustelid activity in the index-lines on the dispersed treatment sites precluded statistical analysis: mean (±SE) presence of mustelids ranged from 0.06 ± 0.04 to 0.17 ± 0.07.

Table 4
Overall mean ($n = 16$; 4 replicate sites × 4 years) ± SE presence of mustelids (based on the number of live captures, fecal scats, and trap disturbance per index-line as a proportion of the total observations on the five lines) among the *linear arrays of index-lines in the piles treatment sites* at the Summerland and Golden study areas, southern British Columbia, Canada, and results of RM-ANOVA.

		RM-ANOVA					
Treatment		Treatment		Time		Treatment × time	
Piles index- line	Mustelids	$F_{4,12}$	P	$F_{3,45}$	P	$F_{12,45}$	P
		2.16	0.14	0.18	0.91	0.82	0.63
1	0.19 ± 0.04						
2	0.24 ± 0.04						
3	0.30 ± 0.04						
4	0.14 ± 0.03						
5	0.12 ± 0.04						
Overall ($n = 80$)	0.20 ± 0.02						

3.3. *Abundance and diversity of small mammals*

A total of eight species of forest-floor small mammals, composed of 5566 individuals, were captured in 27 and 24 trapping periods at the Summerland and Golden study areas, respectively. *M. longicaudus* was the most common species captured with 1521 individuals followed by *P. maniculatus* (1513), *C. gapperi* (1036), montane shrew (*Sorex monticolus*) (782), *N. amoenus* (397), masked shrew (*S. cinereus*) (203), heather vole (61), and meadow vole (53). Susceptibility to capture was measured by Jolly trappability estimates with mean (±SE) values of 73.5 ± 6.0% for *C. gapperi*; 75.7 ± 2.6% for *M. longicaudus*; and 77.0 ± 2.1% for *P. maniculatus* in those sites where these species were common.

Mean abundance of *C. gapperi* was similar ($F_{1,3} = 5.09$; $P = 0.11$) statistically between treatment sites but with numbers 3 to 10 times higher, on average, in the piles than dispersed sites (Table 5; Fig. 3). Red-backed vole numbers declined significantly ($P < 0.01$) during the four post-harvest years. Mean abundance of *M. longicaudus* was significantly ($F_{1,3} = 9.38$; $P = 0.05$) different with the piles at consistently higher numbers (1.2 to 4.4 times on average) than the dispersed sites (Table 5; Fig. 4). Mean abundance of *P. maniculatus* and *Sorex* spp. were similar ($P \geq 0.25$) between treatment sites (Table 5; Fig. 5). Mean (±SE) abundance of *N. amoenus* was 3.6 ± 0.8 and 3.4 ± 1.0 animals per line in the dispersed and piles sites, respectively, at West Munro. These mean (±SE) estimates for northwestern chipmunks in dispersed and piles sites were 5.6 ± 1.3 and 7.1 ± 2.0, respectively, at East Munro.

Mean abundance of total small mammals, including the less common species, was significantly ($F_{1,3} = 9.52$; $P = 0.05$) different between treatment sites with the piles at consistently higher numbers (1.4 to 1.9 times on average) than the dispersed sites (Table 5; Fig. 6). Similarly, mean species richness and diversity were both significantly ($P \leq 0.03$) different between dispersed and piles sites (Fig. 7 + 8; Table 5). Mean richness was 1.2 to 1.6 times and mean diversity was 1.2 to 1.7 times higher in piles than dispersed sites. Other than the significant time effect for mean abundance of red-backed voles, there were no other significant ($P \geq 0.15$) time or treatment × time interaction effects. These comparisons were further clarified by mean values and non-overlapping 95% CIs, whereby overall mean abundance of *C. gapperi* was higher in piles than dispersed sites. Similarly, overall means for total small mammals, species richness, and diversity also followed this pattern (Fig. 9).

Mean (±SE) annual abundance of the three major species, total small mammals, species richness, and diversity per line were all similar ($P \geq 0.12$) among pile index-lines (1 to 5) across these clearcut openings (Table 6). Again, except for a significant ($P < 0.01$) time effect for *C. gapperi*, there were no significant ($P \geq 0.15$) time or treatment × time

Table 5
Mean ($n = 4$ replicate sites) ± SE annual abundance for each species, total abundance, species richness, and species diversity per index-line within the forest-floor small mammal community for the first four post-harvest years in the dispersed and piles treatments, at the Summerland and Golden study areas, southern British Columbia, Canada, and results of RM-ANOVA. Within a row, columns of mean values with different letters are significantly different. F-values identified by * were calculated using an H-F correction factor, which decreased the stated degrees of freedom due to correlation among repeated measures. Significant values are given in bold text.

RM-ANOVA								
			Treatment		Time		Treatment × time	
Parameter and year post-harvest	Dispersed	Piles	$F_{1,3}$	P	$F_{3,18}$	P	$F_{3,18}$	P
Mean abundance								
C. gapperi	–	–	5.09	0.11	5.32*	**<0.01**	1.16*	0.35
1	2.7 ± 1.0	8.2 ± 2.5						
2	0.4 ± 0.2	4.2 ± 2.0						
3	0.4 ± 0.3	2.1 ± 0.5						
4	0.1 ± 0.1	0.9 ± 0.3						
M. longicaudus	B	A	9.38	0.054	2.79*	0.10	0.73*	0.50
1	3.7 ± 2.1	5.0 ± 1.2						
2	5.1 ± 2.8	6.0 ± 3.0						
3	1.8 ± 1.0	3.6 ± 0.7						
4	0.8 ± 0.6	3.5 ± 0.8						
P maniculatus	–	–	2.02	0.25	1.23	0.33	0.11	0.95
1	2.3 ± 1.3	4.7 ± 2.3						
2	3.0 ± 1.0	3.9 ± 0.9						
3	4.7 ± 2.2	5.8 ± 1.7						
4	1.7 ± 0.8	2.4 ± 0.7						
Total *Sorex* spp.	–	–	0.39	0.58	0.98	0.42	0.41	0.75
1	0.7 ± 0.6	0.7 ± 0.3						
2	0.9 ± 0.2	1.0 ± 0.1						
3	1.3 ± 0.6	1.4 ± 0.3						
4	0.8 ± 0.2	1.5 ± 0.3						
Total small mammals	B	A	9.52	0.054	1.70*	0.20	0.09*	0.97
1	11.4 ± 3.6	19.5 ± 1.7						
2	12.8 ± 2.6	17.8 ± 2.0						
3	12.0 ± 2.8	17.3 ± 3.0						
4	6.1 ± 1.6	11.3 ± 1.7						
Species richness	B	A	14.29	**0.03**	1.45*	0.26	0.60*	0.62
1	2.70 ± 0.40	3.22 ± 0.31						
2	3.04 ± 0.16	3.68 ± 0.06						
3	3.00 ± 0.31	3.78 ± 0.17						
4	2.15 ± 0.15	3.50 ± 0.26						
Species diversity	B	A	48.45	**<0.01**	0.68*	0.43	0.44*	0.52
1	1.05 ± 0.20	1.23 ± 0.17						
2	1.17 ± 0.02	1.44 ± 0.04						
3	1.16 ± 0.11	1.54 ± 0.07						
4	0.87 ± 0.08	1.49 ± 0.11						

Fig. 3. Mean ($n = 4$ replicate sites) ± SE number of *Clethrionomys gapperi* per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

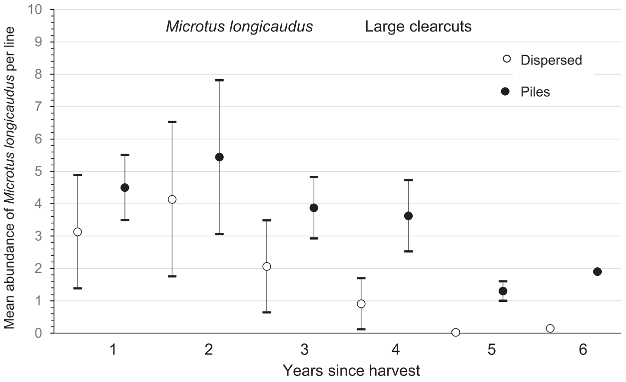

Fig. 4. Mean ($n = 4$ replicate sites) ± SE number of *Microtus longicaudus* per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

interaction effects. Low sample sizes on index-lines in the dispersed sites precluded statistical comparison.

3.4. Reproduction of the major species

Mean values for recruitment, number of successful pregnancies, and index of juvenile productivity of *C. gapperi* were all similar ($P \geq 0.10$) between sites (Table 7). Pregnancy rates of *C. gapperi* declined significantly ($P = 0.02$) with time in accordance with the decline in abundance (Table 7). Mean number of total recruits of *M. longicaudus* was significantly ($F_{1,3} = 15.45$; $P = 0.03$) higher in the piles than dispersed sites but mean number of juvenile recruits was similar between sites ($P = 0.08$).

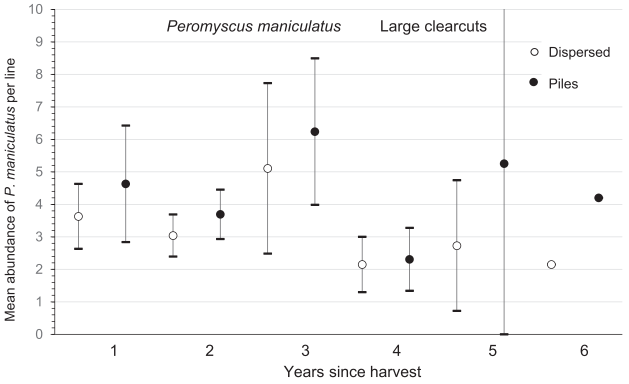

Fig. 5. Mean ($n = 4$ replicate sites) ± SE number of *Peromyscus maniculatus* per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

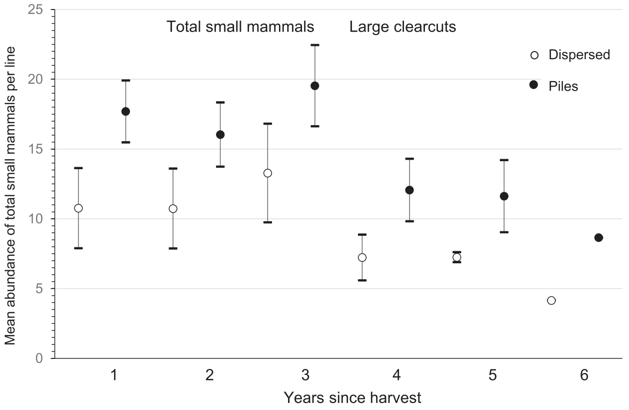

Fig. 6. Mean ($n = 4$ replicate sites) ± SE number of total forest-floor small mammals per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

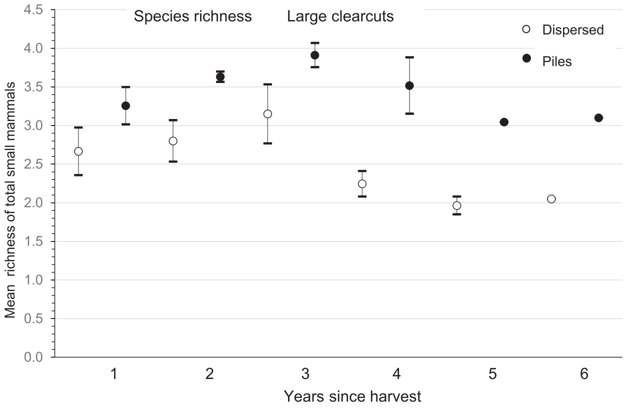

Fig. 7. Mean ($n = 4$ replicate sites) ± SE species richness of forest-floor small mammals per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

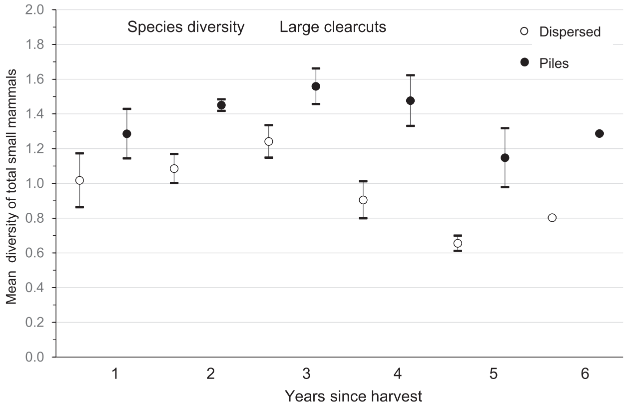

Fig. 8. Mean ($n = 4$ replicate sites) ± SE species diversity of forest-floor small mammals per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

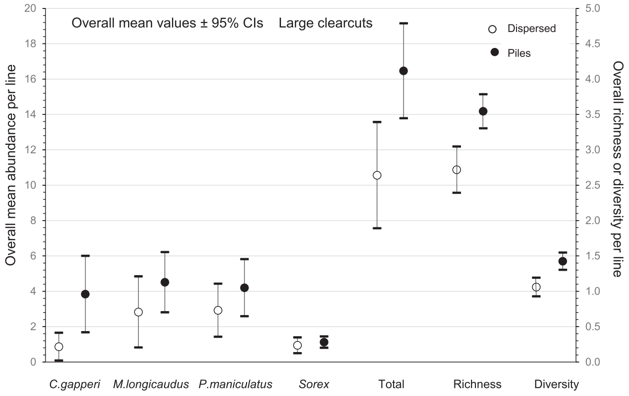

Fig. 9. Overall mean ($n = 16$; 4 replicate sites × 4 years) ± 95% CIs number and species richness and diversity of forest-floor small mammals per index-line in the dispersed and piles sites during the first six years after clearcut harvesting, southern British Columbia, Canada.

Mean number of successful pregnancies and mean index of juvenile productivity of *M. longicaudus* were similar ($P \geq 0.09$) between the two sites (Table 7). The pattern of reproductive attributes for *P. maniculatus* was similar to the other two major species (Table 7). Except for the significant time effect for *C. gapperi*, there were no significant ($P \geq 0.15$) time or treatment × time interaction effects for any of these analyses for the three species.

4. Discussion

4.1. Presence of mustelids

Our results for presence of small mustelids supported H_1 that these small carnivores would occur more frequently (4.5–8.0 times) on sites with piles of woody debris than on sites with dispersed debris. This result included short-tailed weasels, long-tailed weasels, and American marten and was recorded during the first four post-harvest years on “large” clearcuts (28 to 50 ha) created by conventional forest harvesting in two study areas. This index of mustelid activity patterns was similar among the four blocks that represented the two geographic zones and different forest ecosystems. Thus, considering that small mustelids are wide-ranging predators covering up to 100 s of ha (Powell, 1994; King et al., 2007), these carnivores used the woody debris piles in all four experimental blocks in a relatively consistent manner. In addition, the linear configuration of debris piles had substantial populations of small mammal prey. Similar results have been reported for mustelid use of constructed piles or windrows of woody debris on smaller clearcut units in montane spruce and Douglas-fir forests near Summerland, Elkhart, and Golden, BC (Sullivan et al., 2012; Sullivan et al., 2017a). Studies of single mustelid species reported significant use of woody debris structures by marten in northern sub-boreal spruce forest near Prince George,

Table 6
Overall mean ($n = 16$; 4 replicate sites × 4 years) ± SE annual abundance for each species, total abundance, species richness, and species diversity per index-line within the forest-floor small mammal community for the first four post-harvest years among the *linear arrays of index-lines in the piles treatment sites* at the Summerland and Golden study areas, southern British Columbia, Canada.

	Species or parameter						
Index-line	*C. gapperi*	*M. longicaud.*	*P. maniculat.*	*Sorex* spp.	Total small mammals	Species richness	Species diversity
1	3.3 ± 0.9	5.6 ± 0.9	5.2 ± 1.1	1.0 ± 0.2	17.8 ± 1.5	3.64 ± 0.15	1.49 ± 0.06
2	4.4 ± 1.1	3.3 ± 0.8	3.5 ± 0.9	0.9 ± 0.2	15.1 ± 1.6	3.41 ± 0.12	1.36 ± 0.05
3	2.6 ± 0.9	5.9 ± 1.2	3.5 ± 1.0	1.4 ± 0.3	17.0 ± 1.9	3.49 ± 0.19	1.38 ± 0.09
4	4.7 ± 1.4	4.1 ± 1.1	4.4 ± 1.1	1.2 ± 0.2	16.8 ± 1.5	3.57 ± 0.14	1.41 ± 0.08
5	4.2 ± 1.2	3.7 ± 0.8	3.9 ± 0.8	1.2 ± 0.2	15.6 ± 1.3	3.62 ± 0.18	1.49 ± 0.09
Overall ($n = 80$)	3.8 ± 0.5	4.5 ± 0.4	4.1 ± 0.4	1.1 ± 1.1	16.5 ± 0.7	3.55 ± 0.07	1.42 ± 0.03

Table 7
Overall mean ($n = 16$; 4 replicate sites × 4 years) ± SE demographic attributes for *C. gapperi*, *M. longicaudus*, and *P. maniculatus* per index-line within the forest-floor small mammal community for the first four post-harvest years in the dispersed and piles treatments, at the Summerland and Golden study areas, southern British Columbia, Canada, and results of RM-ANOVA. *F*-values identified by * were calculated using an H-F correction factor, which decreased the stated degrees of freedom due to correlation among repeated measures. Significant values are given in bold text.

RM-ANOVA								
Species + parameter	Dispersed	Piles	Treatment		Time		Treatment × time	
C. gapperi			$F_{1,3}$	*P*	$F_{3,18}$	*P*	$F_{3,18}$	*P*
Juvenile recruits	1.13 ± 0.65	4.93 ± 1.48	5.41	0.10	3.07*	0.12	0.80*	0.40
Total recruits	2.20 ± 1.07	9.15 ± 2.47	5.25	0.11	5.01*	0.056	1.26*	0.29
Pregnancies	0.73 ± 0.22	2.78 ± 0.76	3.75	0.15	5.46*	**0.02**	1.20*	0.33
Juvenile productivity	0.97 ± 0.46	1.74 ± 0.35	3.35	0.16	0.48*	0.70	0.39*	0.69
M. longicaudus			$F_{1,3}$	*P*	$F_{3,18}$	*P*	$F_{3,18}$	*P*
Juvenile recruits	4.64 ± 1.68	7.04 ± 1.10	7.06	0.08	1.04*	0.38	0.02*	0.98
Total recruits	6.44 ± 2.31	11.05 ± 1.78	15.45	**0.03**	1.05*	0.33	0.09*	0.77
Pregnancies	1.83 ± 0.63	3.55 ± 0.68	5.81	0.09	0.73*	0.50	0.05*	0.95
Juvenile productivity	1.36 ± 0.41	2.42 ± 0.27	2.08	0.24	0.10*	0.90	0.52*	0.61
P. maniculatus			$F_{1,3}$	*P*	$F_{3,18}$	*P*	$F_{3,18}$	*P*
Juvenile recruits	4.70 ± 1.37	7.45 ± 1.29	6.32	0.09	1.45	0.26	0.13	0.94
Total recruits	5.69 ± 1.52	10.43 ± 1.41	10.97	**0.04**	1.48	0.25	0.27	0.85
Pregnancies	0.74 ± 0.16	2.06 ± 0.40	4.39	0.13	0.13	0.94	0.11	0.95
Juvenile productivity	3.90 ± 1.15	5.10 ± 1.64	0.98	0.40	0.81*	0.39	0.33*	0.58

BC and by short-tailed weasels in boreal mixedwood forest in northeastern Alberta (Lisgo et al., 2002; Seip et al., 2018).

Den and rest locations of Pacific marten (*M. caurina*) followed a pattern of fine-scale forest heterogeneity of variable sizes of logs, snags, and live trees that offered increased structural complexity in Oregon and northern California (Delheimer et al., 2023). This observed pattern of use by marten may fit that offered by constructed piles and windrows of post-harvest woody debris on clearcuts.

In another study investigating various structural treatments post-harvest on new clearcuts, woody debris piles and riparian sites dominated the pattern of presence of small mustelids (Sullivan and Sullivan, 2021). The linear configuration of piles and nature of riparian zones provided pathways for dispersal movements of mustelids via cover and small mammal prey (Buskirk and Zielinski, 2003; Linnell et al., 2017a; Mougeot et al., 2020). Marten and weasels forage and select paths near or in downed wood (Buskirk and Zielinski, 2003; Andruskiw et al., 2008; Bunnell and Houde, 2010). Indeed, mustelid activity along our linear arrays of piles was also similar in these relatively large openings. Other linear habitats included upland and riparian strips on new clearcuts in Quebec where similar results were reported for marten, short-tailed weasels, and their small mammal prey (Darveau et al., 2001; Potvin and Bertrand, 2004).

In the very large openings occurring after wildfire, Pacific marten chose structurally complex sites with abundant coarse woody debris and residual trees, snags, and understory saplings and shrubs (Volkmann and Hodges, 2022). However, the secondary disturbance from salvage logging is substantially more harmful to marten, other forest specialists, and prey species than the original fire (Lindenmayer et al., 2008; Kelly and Hodges, 2022; Volkmann and Hodges, 2022). To this end, provision of woody debris piles (i.e., slash piles) may improve habitat and connectivity in salvage-logged areas for marten and prey species (Wilk and Raphael, 2018; Sullivan et al., 2021). A similar strategy could be adopted for large-scale insect outbreaks such as mountain pine beetle (*Dendroctonus ponderosae*) where salvage logging also occurs frequently and affects marten and other wildlife species (Steventon and Daust, 2009).

4.2. Forest-floor small mammal communities

Mean total numbers, species richness, and species diversity of small mammal prey species were all greater on sites with woody debris piles than on sites with dispersed debris, and hence supported the prediction of H_2. The composition of total mammals was dominated (73.1%) by three species: *C. gapperi*, *M. longicaudus*, and *P. maniculatus*. The major old forest species, *C. gapperi*, was higher in overall mean abundance in the piles than dispersed sites in all years but declined dramatically through the later post-harvest years. This initial response of *C. gapperi* to the debris piles was similar to other reports of constructed piles and windrows of post-harvest debris on new clearcuts (Lisgo et al., 2002; Fauteux et al., 2012; Seip et al., 2018; Sullivan and Sullivan, 2019). The dramatic population changes in *C. gapperi* over the four post-harvest years may have been related to the 6- to 7-year population fluctuation and peak abundance of this microtine in old forest sites in 2017–2018 before declining to low numbers in 2019–2020 (Sullivan and Sullivan, 2019). The disappearance of red-backed voles on new clearcut sites has been reported by several authors (Fisher and Wilkinson, 2005; Zwolak, 2009).

The similar overall mean abundance of *M. longicaudus* in piles and dispersed sites likely reflected the initial flush of *Microtus* species on at least some new clearcuts at 2 to 4 years post-harvest in both study areas (Sullivan and Sullivan, 2010, 2022). However, after this initial increase, long-tailed voles declined and there were fewer voles in dispersed sites than those with piles of woody debris. Thus, both *C. gapperi* and *M. longicaudus* did support H_2. The generalist *P. maniculatus* and *N. amoenus* were at similar mean abundance in piles and dispersed treatments of woody debris as reported in other investigations (Smith and Maguire, 2004; Craig et al., 2006; Waldien et al., 2006; Sullivan et al., 2017a), and hence along with *Sorex* spp. did not support H_2.

Conversely, *N. amoenus* was at significantly higher abundance in piles and windrows of debris than dispersed sites in an earlier study of small clearcut openings (Sullivan and Sullivan, 2019). Similarly, *S. monticolus* and soricids were at higher abundance in debris structures than dispersed sites in central BC (Sullivan et al., 2017a; Seip et al., 2018) and South Carolina, USA (Davis et al., 2010), respectively. Overall small mammal prey responses to construction of post-harvest woody debris habitats in North America are summarized in Sullivan et al. (2021).

4.3. Reproduction of the major species

The prediction of H_3 that reproductive attributes of the major species would be greater in piles of woody debris than dispersed debris was supported in part for total recruits of *M. longicaudus* and *P. maniculatus*. The other measures of juvenile recruits, pregnancies, and juvenile productivity also tended to follow this pattern at least in terms of biological importance but not statistical significance for all three species. Therefore, habitat quality in the woody debris piles did seem to be high enough to support these species in the first four years after harvest. Counts of small mammals as abundance or density per unit area may not be a sound measure of habitat quality, particularly where changes in forest succession or management occur (Van Horne, 1983; Lemaitre et al., 2010). Demographic parameters such as reproduction and survival also need to be investigated rather than just short-term measures of relative abundance and habitat associations. It seems likely that our woody debris habitats may be considered sources of small mammals as a means of balanced dispersal to maintain these species through succession of the regenerating forest (Diffendorfer, 1998; Sullivan and Sullivan, 2019).

4.4. Debris piles as part of forest restoration

The linear configuration of debris piles on our clearcut sites seemed to be colonized and used by small mammal prey species and small mustelids in a consistently uniform pattern that supported H_4: response variables would be similar across a linear array of debris piles on large openings. Although the retention of woody debris in piles across large (e.g., 30–50 ha) clearcut openings appears to be an effective way to provide habitat for small mustelids and their mammalian prey, post-harvest residues continue to be disposed of, usually by biomass removals or by burning (Homyack and Verschuyl, 2019; Sullivan et al., 2021). In cases where piles or accumulations are considered a fire-fuel hazard, current forest management policy requires forest operators to dispose of excess harvest residues as a means of hazard abatement in most provinces in Canada and in many jurisdictions in the US. However, except for situations with human intervention, there is no scientific evidence that post-harvest debris piles are ignition points for forest fires (Sullivan et al., 2021).

Conservation of debris piles as a baseline structure on clearcuts and other disturbed sites of resource extraction such as mining and petroleum would be applying trophic structure to forest restoration and land reclamation practices (Montoya et al., 2012; Fraser et al., 2015). Clearly, sufficient woody debris is necessary to achieve this objective and in the very large (e.g., 100 s to 1000 s of ha) wildfire areas, this task may be difficult if the burning was particularly severe and left little residual woody material (Lindenmayer et al., 2008; Volkmann and Hodges, 2022). However, in disturbed forest sites that are salvage logged, presumably some or enough post-harvest debris might be available to conserve piles of debris or other accumulations on the harvested site. Although our “large openings” were limited to 30–50 ha, results suggest that small mustelids and prey species may also use debris structures from salvage logging on much larger openings associated with wildfire and insect outbreaks.

5. Study limitations

We chose four replicate blocks that were the size of openings (30 to 50 ha) from conventional clearcutting and that each had a linear configuration of debris piles arranged along the long axis of the block. Ideally, the blocks should have all been harvested at the same time but this was not possible. Thus, we compared mustelid presence and small mammal prey species for four years post-harvest where data were available for the four replicates. There was some variability in the number and dimensions of piles across blocks but this provided for a broader set of inferences, and all were in the same degree of magnitude as previous studies. Overall amounts of debris and length of lines of debris piles across the blocks were reasonably similar. The interruption of the Dart Creek site was beyond our control and fortunately we were able to adopt a replacement block with similar attributes of woody debris piles and abundance patterns of mammal species to complete the data collection.

Overall, very few (234/4070 = 5.7%) individuals of the three major small mammal species were captured on more than one index-line thereby suggesting that our study design was composed of independent experimental units as per Hurlbert (1984). This premise was not accurate for the wide-ranging mustelids and *N. amoenus*, and hence we did not estimate population metrics for these species. Inferences are applicable to clearcut openings of 30 to 50 ha among forest ecosystems in southern BC and potentially to other regions of the Pacific Northwest of North America. In addition, these inferences represent responses of mustelids and forest-floor small mammals to piles and dispersed accumulations of woody debris on new clearcuts during summer and fall (May to October) only for the first four years post-harvest. Unfortunately, we do not have measurements of responses of these mammals to our habitat structures during winter months. Considering the large home range sizes (up to 100 s of ha) (Powell, 1994; King et al., 2007) of these mustelid species, our measurements were essentially an index of activity patterns with more signs of mustelid presence likely representing higher activity in that woody debris treatment.

6. Conclusions

To our knowledge, this study is the first to measure responses of small mustelids and forest-floor small mammals to constructed piles of woody debris as a means of habitat restoration on relatively large (mean area 40.5 ha) openings created by conventional clearcutting. Many forest and wildlife managers have suggested that these structures would not be colonized and used by mammal prey species and small mustelids because of the large opening size. At least for this magnitude of clearcut opening sizes in the first four years post-harvest, this prediction was clearly not the case with respect to mustelid presence and abundance, species richness, and diversity of forest-floor small mammal communities. In addition, some reproductive attributes followed the pattern of abundance for the major species: *C. gapperi*, *M. longicaudus*, and *P. maniculatus*.

A series of piles or windrows of post-harvest debris should connect patches of mature or old-growth forest and riparian areas to allow mustelids, small mammals, and other species to access and traverse clearcut openings. This will be particularly important on conventional-sized openings as in our study, but also on much larger (>100 ha) insect-killed as well as burned forests that are salvage-harvested. Piles and windrows of woody debris need to be built at an appropriate real-world scale (e.g., 2–3 m in height and 5–7 m in width or diameter) at the time of forest harvesting and log processing. Piles of debris seem to act as oases for mustelids and their prey species thereby initiating ecological restoration of cutover forest land.

CRediT authorship contribution statement

Thomas P. Sullivan: Conceptualization, Methodology, Writing – original draft, Visualization, Investigation. **Druscilla S. Sullivan:** Data curation, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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