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# Voles, trees, and woody debris structures as habitat: Balancing forest crop protection and biodiversity

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ABSTRACT

Relatively homogeneous early successional habitats develop after clearcutting and wildfire that voles of the genera *Microtus* and *Myodes* may colonize and generate population fluctuations. In these habitats, vole populations may reach pest status by their feeding on newly planted tree seedlings. Strategic management of excess woody debris into piles and windrows helps diversify new clearcuts by enhancing populations of forest-floor small mammals, including voles, and some of their predators.

This study tested the hypotheses (H) that ($H_1$) abundance of voles and incidence of feeding damage to tree seedlings will be higher in windrow than dispersed (conventional) sites of woody debris, and ($H_2$) there will be a gradient of damage with the highest incidence immediately adjacent to windrows. A third hypothesis ($H_3$) predicts that feeding damage to trees will increase in relation to windrow size. *Microtus* voles and red-backed voles (*Myodes gapperi*) were live-trapped for three years (2010–2012) in replicated sites with woody debris dispersed and in windrows at three study areas in the southern interior of British Columbia, Canada. Incidence of feeding damage and mortality to tree seedlings by voles was measured in all sites. Mean abundance of *M. gapperi*, *Microtus*, and total voles were all significantly ($P \leq 0.04$) higher (up to 3.4 times) in windrow than dispersed sites, and hence the abundance part of $H_1$ was supported. Mean annual percentage of trees damaged by voles was significantly ($P \leq 0.03$) higher in windrow than dispersed sites over the two winters and for cumulative incidence of damage, and hence the tree damage part of $H_1$ was supported. Mortality of trees followed this pattern but was not formally significant. Trees planted immediately adjacent to a windrow had significantly ($P < 0.01$) greater feeding damage than seedlings planted further away, and hence $H_2$ was supported. There were significant linear relationships between mean percentage of trees killed ($r = 0.67$; $P < 0.01$) and mean number of total voles, and also with mean volume of woody debris per meter length of windrow ($r = 0.98$; $P < 0.01$). Thus, $H_3$ was also supported. To minimize overall mortality of trees, it is likely worthwhile to not plant trees near windrows. Slightly reduced stocking (<5% net forest area) and potential loss of some trees to voles need to be balanced alongside biodiversity and conservation considerations provided by woody debris structures.



## 1. Introduction

Although many alternative silvicultural systems are available to harvest coniferous forests, clearcutting continues to dominate in temperate and boreal zones of North America and Europe. An additional form of clear felling is salvage harvesting of forests influenced by wildfire, wind events, or insect outbreaks (e.g., mountain pine beetle, *Dendroctonus ponderosae*) that typically create very large (>100 ha) openings (Lindenmayer et al., 2008). Relatively homogeneous early successional habitats develop after clearcutting and wildfire and these are ideal situations for voles of the genera *Microtus* and *Myodes* to colonize and generate population fluctuations (Hansson, 1985; Fisher and Wilkinson, 2005; Sullivan and Sullivan, 2010). The grasses, forbs, and shrubs in these habitats provide food and cover for *Microtus* voles (Batzli, 1985; Ostfeld, 1985). The cover and plant productivity components of herbaceous vegetation seem to be related to habitat quality and potential fluctuations of some *Microtus* species (Birney et al., 1976; Laine and Henttonen, 1983; Adler and Wilson, 1989). Post-clearcutting and wildfire habitats are typically dominated by

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herbs and grasses, and often voles, for up to 5–10 years, depending on the ecosystem.

Regeneration of cutover forest lands is normally achieved by planting of nursery-grown tree seedlings in the first year or two after harvest (the sooner the better to reduce competition problems). It is at this stage that voles may feed on the bark, cambium, phloem, and roots of young trees as a food source during winter months when alternative natural foods are limited (Huitu et al., 2009). Damage impacts include direct mortality from girdling and clipping of tree stems, and reduced growth of trees that survive sub-lethal feeding injuries (Sullivan et al., 1990). Nursery-raised tree seedlings with their enhanced nutrient content from fertilizers are more desirable than wildlings that arise from natural regeneration (Sullivan and Martin, 1991). In general, feeding damage is most common during winters with high vole populations in early successional habitats after clearcutting (Huitu et al., 2009; Sullivan and Sullivan, 2010). This increases reforestation costs and the time required to reach "free growing status" and reduces: mean annual growth increment, net productive forested area, and the regeneration of appropriate tree species in certain forest ecosystems. Although occurring on a site-specific basis, feeding damage to deciduous and coniferous tree plantations by *Microtus* and *Myodes* species is prevalent in North America, Europe, and Asia (Byers, 1984; Hansson, 1985; Shu, 1985; Gill, 1992; Huitu et al., 2009).

In an attempt to help diversify new clearcuts to benefit wildlife, particularly mammals, managers have moved excess woody debris into piles and windrows. That has significantly enhanced populations of forest-floor small mammals, including the southern red-backed vole (*Myodes gapperi*), the long-tailed vole (*Microtus longicaudus*), and some of their predators (Sullivan et al., 2011, 2012). Excess woody debris from felling operations is typically burned to reduce a perceived fire hazard, but this material contributes to habitat quality for a wide range of mammal species (McComb, 2003). Although viewed positively from a biodiversity and conservation perspective, there is a cautionary note with respect to vole populations building to "pest status" and subsequent damage to newly planted tree seedlings. Sullivan and Sullivan (2012) reported <10% incidence of feeding damage by *M. longicaudus* to tree seedlings on treatment sites with piles and windrows of woody debris as habitat. However, the sampling distribution of seedlings was random over each clearcut site, with and without piles or windrows, and did not specifically measure the incidence of damage near these constructed habitats. In addition, the confirmation of *M. gapperi* as a seedling predator has been equivocal since this species occurs primarily in mature/old-growth stands of timber and disappears from clearcuts within the first year or two post-harvest, at least in western North America (Zwolak, 2009). However, as reported by Sullivan et al. (2011), *M. gapperi* occurred in piles and windrows on new clearcuts at abundance levels similar to, or higher than, old forest and may, indeed, feed on newly planted trees adjacent to these constructed habitats.

Thus, this study was designed to test the hypotheses (H) that ($H_1$) abundance of voles and incidence of feeding damage to tree seedlings will be higher in windrow than dispersed (conventional) sites of woody debris, and ($H_2$) there will be a gradient of damage with the highest incidence immediately adjacent to windrows. A third hypothesis ($H_3$) predicts that feeding damage to trees will increase in relation to windrow size.

## 2. Methods

### 2.1. Study areas

Three study areas were located in the southern interior of British Columbia (BC), Canada: 1) East Munro (49°40′27″N; 119°51′49″W) and 2) Kathleen Lake (49°44′57″N; 120°06′13″W) on the Okanagan Plateau 25 and 47 km, respectively, west of Summerland, BC, and 3) Blaeberry River (51°28′25″N; 116°58′59″W) in the Rocky Mountains 20 km northeast of Golden, BC. Biogeoclimatic ecological zones were 1) Interior Douglas-fir ($IDF_{dk}$), 2) Montane spruce ($MS_{dm}$), and 3) Interior Cedar-Hemlock ($ICH_{mk}$) (Meidinger and Pojar, 1991). General topography at 1) and 2) is rolling hills at 1300–1520 m elevation, and at 3) ranges from hilly to steep terrain at 870–890 m elevation.

The upper IDF and MS have a cool, continental climate with cold winters and moderately short, warm summers. The average temperature is below 0 °C for 2–5 months, and above 10 °C for 2–5 months, with mean annual precipitation ranging from 30 to 90 cm. Open to closed mature forests of Douglas-fir (*Pseudotsuga menziesii*) cover much of the IDF zone, with even-aged post-fire lodgepole pine (*Pinus contorta*) stands at higher elevations. The MS landscape has extensive young and maturing seral stages of lodgepole pine, which have regenerated after wildfire. Hybrid interior spruce (*Picea glauca* × *Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are the dominant shade-tolerant climax trees. Douglas-fir is an important seral species in zonal ecosystems and is a climax species on warm south-facing slopes in the driest ecosystems. Trembling aspen (*Populus tremuloides*) is a common seral species and black cottonwood (*Populus trichocarpa*) occurs on some moist sites (Meidinger and Pojar, 1991). The ICH has an interior, continental climate with cool wet winters and warm dry summers. Mean annual temperature ranges from 2 to 8.7 °C. The temperature averages below 0 °C for 2–5 months and above 10 °C for 3–5 months of the year. Mean annual precipitation is 50–120 cm, 25–50% of which falls as snow. 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, lodgepole pine, white spruce, Engelmann spruce, their hybrids, and subalpine fir common in these stands (Meidinger and Pojar, 1991).

Prior to harvesting, stands at study areas 1) and 2) were composed of a mixture of lodgepole pine with variable amounts of Douglas-fir, spruce, and some subalpine fir, and at area 3) primarily Douglas-fir with some of the other coniferous species. Mean ages of lodgepole pine ranged from 80 to 120 years and for Douglas-fir ranged from 120 to 220 years. Mean tree heights ranged from 10.5 to 19.5 m for lodgepole pine and from 16.7 to 27.5 m for Douglas-fir. There were no site preparation treatments on any of these harvested sites prior to planting. Mean area of sites ranged from 4.5 to 5.8 ha.

### 2.2. Experimental design

A randomized complete block design with two replicates of the following two treatments (all harvested sites clearcut): (a) dispersed woody debris, and (b) woody debris distributed into windrows (Fig. 1) was used in each of the three study areas. The 12 sites (6 replicates each of woody debris dispersed and in windrows) were selected on the basis of typical operational scale harvest sites, and reasonable proximity of sites to one another within a study area. Sites within a study area were separated by 0.23–0.40 km to enhance statistical independence. A measure of this independence was that no voles were captured on more than one trapping line.

### 2.3. Woody debris treatments

Timber harvesting at East Munro and Kathleen was targeted at lodgepole pine salvage after, or during impending, mountain pine beetle attack; at Blaeberry the target was primarily Douglas-fir.