Impacts of Woody Biomass Harvest on Small Mammals and Plants in Northern Minnesota Aspen Forests

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Dedication

This dissertation is dedicated to my incredible wife Leslie and my 3 amazing kids Morgan, Joseph, and Connor.

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Abstract

Over the last decade, interest has grown in expanding the sources of energy produced in Minnesota, including burning wood alongside or in place of fossil fuels in commercial power plants. One possible source of this wood is through the harvest of residual material (hereafter "woody biomass harvest") left behind following traditional timber harvest. Although voluntary site-level harvest guidelines endorsed by the State initially called for the retention of residual material in the forest, the rules were modified in 2007 to allow for such harvest. It remained unclear, however, what, if any, ecological impacts woody biomass harvest could have in the region.

I examined the immediate impacts of woody biomass harvest on small mammals and vegetation in 3 aspen (*Populus* spp) forests in northeast Minnesota. The study used of a "Before-After, Control-Impact" (BACI) design, with baseline surveys prior to treatment, follow up surveys after treatment, and a paired control that was not treated. I had two treatment categories: clearcut with slash left *in situ* (hereafter "slash-retention"), and clearcut with whole tree skidding and no replacement of residue (slash-removal). Each research site received a full complement of two treatments plus a control, giving me three replicates at each site.

Prior to harvest, the stands contained an average coarse woody debris (CWD) density of 465 cm^2/m^2 (SE=49 cm^2/m^2). After harvest, the density of CWD increased in the slash-retention plots by an average of 422 cm^2/m^2, while slash-removal plots lost on average 29 cm^2/m^2. Prior to harvest there were no statistically significant differences in CWD density between treatments, but after logging CWD was higher in the retention than in the control plots, and higher in the control plots than in the removal plots.. In

addition to gross changes in CWD area, the nature of the CWD in the harvested plots also shifted from initial conditions, with slash-retention plots gaining a disproportionate share of wood in less decayed conditions, while the slash-removal treatments shifted to a CWD base dominated by more decayed wood.

Overall shrub stem counts increased similarly in both harvest treatments though more so in the slash-removal plots. Hazel (*Corylus cornuta)* stem counts increased by more than 3 stems/m^2 in the slash-retention treatment, but stem counts were steady in the slash-removal treatment. Both treatments show a strong, similar regeneration of aspen stems after harvest (3-5 stems/m^2).

Both harvest types exhibited a decline in native forb species cover and an increase in bare ground, cover of non-native plants, and cover of graminoids compared to pre-existing conditions and associated control stands. Changes in native forb cover, non-native forb cover, and bare ground were greater in the slash-removal compared to the slash-retention treatment, but there was not a statistically significant difference between the plots for the increase in graminoid cover. Native species richness of survey plots 2 years post-harvest was not different from pre-harvest values for either treatment.

In just over 29,000 trap nights I recorded 4,838 captures of 1,794 individual animals. I captured 15 mammal, 6 amphibian, and 1 snake species. Overall amphibian captures were low, but trended downward post-harvest in treatment plots. Population estimates of deer and white-footed mice (*Peromyscus* spp.) were little affected by treatment type or harvest status, while red-backed voles (*Myodes gapperi*) showed a positive response to harvest. Shrews (*Sorex sinereus* and *Blarina brevicauda*) and chipmunks (*Tamias striatus*) responded negatively to harvest of both types. The

abundance of the communities overall (all individuals of all species lumped) were generally higher after treatment on the harvest sites than either before conditions or the adjacent controls, with slash-retention plots showing slightly greater gains than slash-removal plots.

In conclusion, although many metrics responded similarly to either harvest type, in all cases where there were differences between the two treatments, the slash-removal treatment yielded a less favorable outcome for wildlife and a further shift from pre-treatment conditions than did the slash-retention treatment. This is especially true for the loss of native species cover, the increase in bare ground and non-native species cover, the loss of CWD, and the lower levels of *C. cornuta*. Although overall small mammal population sizes increased across both harvest types, increases were slightly greater for slash-retention treatments. Overall the results here do not warrant an avoidance of woody biomass harvest, but do argue for some caution, careful monitoring, and thoughtful siting. Future work should revisit these stands to assess longer term impacts.

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Chapter 1

Introduction and Background

There is a fundamental tension between resource extraction and conservation. Central to finding a balance between economic returns from resource extraction and ecosystem integrity is careful study of the impacts of the human use activity on the system (see Kemball et al. 2005; Nguyen-Xuan et al. 2000; Kurulok and Macdonald 2007; Reich et al. 2001 for some examples). As our extractive industries evolve, so too must our research and policies. Over the last decade and a half the timber industry in Minnesota has seen a growth in interest to use forests not only as sources of wood fiber for paper and lumber, but also as potential energy sources. Under this new paradigm wood that traditionally had little economic value and which was typically left in the forest—such as tree limbs, tree tops and dead wood—will be harvested for use as an energy source. This additional harvest will be referred to below as “woody biomass harvest.”

Research shows the boreal forests of North America, including those in Minnesota, evolved under a frequent disturbance regime, most importantly fire but also disturbance from wind and insect outbreaks (De Grandpre and Bergeron 1997; Heinselman 1973; Haeussler et al. 2002; Kemball et al.2005; Nguyen-Xuan et al. 2000; Peltzer et al. 2000; Reich et al.2001). These systems are resilient (Holling 1973, Gunderson 2000), absorbing considerable disturbance but returning to a relatively small suite of stable conditions. In this way an aspen forest may burn, but return to aspen forest type in the natural cycle of succession. Shifts between stable systems (aspen dominant versus maple or spruce/pine dominant forests) are a result of lack of disturbance over periods of time long enough to allow for such succession.

Timber harvest is assumed to mimic fire in its impacts (Kurulok and Macdonald 2007; Reich et al. 2001). This assumed mimicry may not hold as logging techniques and products shift. Indeed it is possible that the resilience of the system and its constituent members may be affected by the shift in disturbance; a forest harvested in this manner may not return to the original state as rapidly, and may lose some constituent members whose own resilience may be affected. This dissertation investigates how timber harvest with woody biomass harvest differs in its immediate ecological impacts in comparison to methods which leave this material behind.

Several studies have compared the impacts of anthropogenic and various natural forms of disturbance (Kurulok and Macdonald 2007; Peltzer et al. 2000; Reich et al.2001). These studies show that the severity of the disturbance may play a stronger role in shifting vegetative communities than the specific form of disturbance itself (MacDonald 2007; Nguyen-Xuan et al. 2000; Pidgen and Mallik 2013). Severity of soil disturbance plays an especially important role in guiding the post-disturbance understory plant community (Haeussler and Kabzems 2005; Nguyen-Xuan et al. 2000).

Using knowledge gained from this type of research, foresters and land managers have developed guidelines for how forest harvest should be done to minimize undesired impacts and maintain ecological integrity (Aust and Blinn 2004; Benjamin 2009; Blinn and Kilgore 2001; DeGraaf 2006; Good Forestry of the Granite State Steering Committee and Bennett 2010; Herrick et al. 2009; Lakel et al. 2006; Phillips and Blinn 2007). In

Minnesota the “Voluntary Site-Level Forest Management Guidelines for Landowners, Loggers and Resource Managers” (hereafter “Guidelines”) were developed by the Minnesota Forest Resources Council (MFRC) and first released in 1998 (http://mn.gov/frc/). The guidelines were revised and re-released in 2005. The Guidelines suggest numerous practices to minimize lasting ecological impacts, including recommendations on road building, pesticide use, harvest methods, and reforestation. The original versions of the Guidelines did not directly address woody biomass harvest, but instead recommended that logging slash be redistributed across the harvest area (MFRC 2005). In 2007 the Guidelines were revised to include best management practices from woody biomass harvest.

Forestry is important to the economy of many rural areas in the boreal and sub-boreal landscapes of North America, including northern Minnesota. Direct logging jobs total around 3,200, with direct, indirect and induced economic impacts of forestry and logging adding $1.1 billion to the State (Deckard and Skurla 2011). Further processing of the raw material into paper pulp, lumber, or finished wood products brings the total to $17.1 billion and 86,775 jobs (Deckard and Skurla 2011).

Minnesota’s forestry sector has declined in recent years, in part from the national recession and imports (Deckard and Skrula 2011). Seven paper pulp plants have closed in the last 5 years (D. Chura, personal communication). Overall commercial harvest in the state has fallen from a high of 3.7 million cords in 2005 (Deckard and Skurla2011) to 2.9 million cords in 2012 (http://www.dnr.state.mn.us/faq/mnfacts/forests.html). Land managers, foresters, and loggers are looking for ways to stabilize or grow the sector. One possible avenue for growth is to harvest woody biomass for energy production either through direct use of the heat or as fuel to power electrical generators (Arnosti, et al. 2008; Becker et al. 2011a; Becker et al. 2011b; Deckard and Skrula 2011; Dirkswager et al. 2011; Evans et al. 2010; Janowiak and Webster 2010; Klockow et al. 2013; Minnesota Department of Natural Resources 2012; Minnesota Forest Resources Council 2007). By 2010, harvest for wood energy constituted 10% of the total timber harvest in the state (Deckard and Skurla 2011; Minnesota Department of Natural Resources 2012). Further, Minnesota law requires 12% percent of electricity to come from renewable sources in 2013, a figure which increases to 25% by 2025 (Minnesota Office of Energy Security 2011; Minnesota Stature 216B.1691). While wind will provide much of the renewable energy supply, woody biomass is an alternative, and Minnesota law calls for the State's agricultural, forestry, and working lands to provide 25% of the energy consumed in the state (Minnesota Statute 41A.11). Consequently, Minnesota is now home to 6 large plants (> 200,000 tons wood used/year) capable of producing heat or electricity from waste wood, including logging slash (Dirkswager 2010) and another 30 smaller plants (Anna Dirskwager, personal communication).

Wood for biomass energy production can be of lower quality than wood for timber harvest (Evans et al. 2010). Biomass harvest for energy production can include non-merchantable small trees, brush, dedicated crops such as hybrid poplar or other plantation trees, logging residue, and residue from urban forests. Logging residue—referred to in this paper as slash—has the potential to make up a large portion of total woody biomass demand. The residue is created as a byproduct of forest harvest, and can include stumps, limbs, and the tops of trees (Evans et al. 2010). In addition to harvesting this by-product, existing snags and coarse woody debris (CWD) may also be removed (Evans et al. 2010).

CWD is not simply an economic resource, however. Whether through fire, windfall, disease, or logging, trees die, and become snags, stumps, and ground floor CWD. These features are an important component of forests (see Evans et al. 2010; Feauteux et al. 2012; Franklin et al. 1987; Harmon et al. 1986; Janowiak and Webster 2010; Jonsel 2008; Maser et al 1979; Maser and Trappe 1984; Siitonen 2001; and Stewart et al. 2010 for excellent reviews). Coarse woody debris plays a vital role in forests in nutrient cycling (Harmon et al. 1986; Stewart et al. 2010) and in moisture retention as decaying wood can retain water more efficiently than soil (Fraver et al. 2002; Maser and Trappe 1984). Coarse woody debris also provides habitat for fungi (Boddy 2001; Maser and Trappe 1984; Siitonen 2001), lichens (Astrom et al. 2005), mosses (Astrom et al. 2005; Siitonen 2001), insects (particularly saproxylic insects, Per and Ostlund 2009; Siitonen 2001), amphibians (Bull 2002; Butts and McComb 2000; Maguire 2003; Thompson et al. 2003), birds (Bull 2002), and mammals (Bowman et al. 2000; Bull 2002; Butts and McComb 2000; Carey and Harrington 2001; Ecke et al. 2001; Fauteux et al. 2012; Fisher and Wilkinson 2005; Johnston and Anthony 2008; Keinath and Hayward 2003; Loeb 1999; McCay and Komoroski 2004; Maguire 2002; Moses and Boutin 2001; Riffel et al 2011; and Thompson et al. 2003), as nesting sites, cover, substrate, moisture refugia, or food.

In recognition of the importance of CWD to the ecology of forests, the original Guidelines specifically called for logging slash to be scattered evenly across the harvested area (Minnesota Forest Resources Council, 2005). As discussed above, silvicultural practices that impact the forest floor, and especially the soil, can have a large effect on the plant community and the ecology of forests. Harvest of existing CWD and the collection of created slash may impact the forest soil.

Because of its potential importance in forest ecology, it is important that scientists analyze the impacts of harvesting CWD, including effects on native vegetation, the overall plant community, invasive species, and microhabitat structure, relative to the impact of standard logging, which typically creates a flush of new CWD as tops, branches, and non-marketable trees are left behind. This dissertation compares the impacts of harvesting with slash-retention—as suggested in the original Guidelines—and harvest with full slash-removal. As discussed above, the scale of impact on the plant community is a function of the severity of the disturbance. The extra mechanization involved in removing logging slash as well as the removal of CWD itself both may contribute to the severity of disturbance. At the core of this study is the question of whether clearcutting with slash and CWD removed from the site affects the post-harvest plant and small mammal community differently than clearcutting with slash and CWD retained on site. Only longer term or retrospective studies will answer the underlying concerns of resiliency[1], though we may see here the beginnings of trends if harvest with biomass removal diverges from timber harvest with biomass retention. If the removal of woody biomass impacts the ability of the system to absorb the disturbance we may see a

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[1] Used here to mean “ecological resiliency", as per Gunderson 2000.

shift to an alternate state, or alternate forest type, possibly one which hosts a different suite of species as some who were previously subsidized by the CWD are lost.

Although the scope of possible impacts is wide, I focused on how the two contrasting harvests—clearcutting with slash-retention and clearcutting with slash-removal—differ in their impacts on small mammals, forest vegetation, shrubs, and aspen regeneration. The following paragraphs describe hypotheses about potential impacts that this dissertation will test.

Small mammals.—Timber harvest that leaves existing CWD and adds more in the form of slash creates a structurally complex forest floor—likely a more complex one than existed before the harvesting. Logging that removes the generated slash and existing CWD will result in a less structurally complex forest floor. Structural complexity of the forest floor may allow for more habitat partitioning by the small mammal community, leading to higher species richness and diversity. The CWD may also provide for a larger overall community abundance (sum of population estimates for all species) by increasing food availability through increased abundance of saproxylic insects and fungi, both of which are eaten by a variety of small mammal species (Maser et al. 1979; Rhodes 1986).

Hypothesis 1: *Harvest following either of the two treatment protocols will increase overall small mammal community abundance, but lower the community diversity of the slash-removal plots relative to pre-harvest and control plots.*

Hypothesis 2: Total small mammal abundance and diversity post-harvest will be higher in the slash-retention than the slash-removal treatment.

Herbaceous Vegetation.—Although few vascular plants are obligate CWD users, many species utilize CWD, and some non-vascular plants are known to depend more strongly on this resource (Astrom et al. 2005; Harmon et al. 1986; Siitonen 2001). Removal of logging slash and existing CWD will also remove a source of shade, nutrients, water retention, and rooting substrate. Extra mechanical effort needed to remove the CWD and slash may also damage plants or the soil.

Hypothesis 3: Removal of logging slash and CWD will lead to a less diverse plant community and decline of more sensitive species.

Hypothesis 4: Removal of logging slash and CWD will open the forest to more invasion by non-native species.

Woody shrubs.—Minnesota is home to 247 native shrub and tree species[2], and now hosts 26 non-native shrubs and trees (MNTaxa 2013). Shrubs are an important part of the ecosystem, providing food and habitat for small mammals and birds, stabilizing soil, shading the forest floor, and providing cover for larger animals. Shrubs are also considered a competitor with tree regeneration (Mundell, et al. 2007) and some silviculture methods are intended to remove or inhibit the growth of shrubs. Whole-tree

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[2] Total includes recognized varieties/cultivars of the same species

skidding, a likely method for woody biomass harvest and the method employed here, is one such technique. The soil scarification and the deliberate and incidental uprooting and tearing of shrubs through the dragging of whole trees could negatively affect shade tolerant shrubs, or may benefit less shade tolerant shrubs.

Hypothesis 5: Whole tree skidding will inhibit regeneration of beaked hazel (*Corylus cornuta*).

Hypothesis 6: There will be a noticeable drop in stem density for shrubs commonly found in the interior forest in the slash-removal but not the slash-retention plots.

Hypothesis 7: *Rubus* spp. stem density will increase under both harvest treatments, but by a greater degree on the slash-removal treatment.

Aspen regeneration.—Quacking aspen (*Populus trembloides*) is the most predominant forest type and most harvested tree species in Minnesota (Miles and VanderSchaf 2012; MN DNR 2013; Palik et al. 2003). A related species, *P. balsamifera* (known colloquially by a variety of names including balsam poplar, popple, and Balm-of-Gilead) is also extensively harvested in the state (MN DNR 2013). Along with Largetooth aspen (*P. grandidentata*) the genus represents about 50% of the volume cut in the state (MN DNR 2013) and about 18% of the total live volume of trees on forest land (Miles and VanderSchaf 2012). Each of the harvests on which I conducted research was wholly or partially intended to maximize regeneration of *Populus* species (either *P. trembloides* or *P. balsamifera*). Other goals included preservation/enhancement of red oak (*Quercus rubra*, Carlton site) and paper birch regeneration (*Betula papyrifra*, St. Louis site). One concern of foresters and loggers is a fear that leaving slash on the site would inhibit aspen regeneration. A corollary to this fear is belief that removal of the slash and the increased scarification of whole-tree skidding would improve aspen growth.

Hypothesis 8: Retention of coarse woody debris will lead to lower densities of aspen suckers relative to the removal treatment.

To investigate these hypotheses I selected 4 forest stands in northeastern Minnesota with existing harvest prescriptions that could be amended. Working with the person in charge of timber harvest management for each site[3], we created a harvest management scheme to test the impact of harvesting slash and CWD on various aspects of the forest ecosystem. We divided each harvest site into 3 roughly equal parts: 1/3 of each site designated to be clearcut with slash retained on site and no harvest of downed coarse woody debris (CWD) or snags (retention treatment); 1/3 to be clearcut with whole-tree skidding, no dispersal of slash, and intentional harvest of snags and CWD ("removal treatment"); and 1/3 to remain uncut to serve as an unharvested control. The project design was a Before-After, Control-Impact (BACI) study (Stewart-Oaten et al. 1986), with surveys both before and after site treatment. Since I was beginning with uncut (secondary growth)

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[3] For the 2 Cloquet Forestry Center locations this was Ron Severs (Director of Operations). For Carlton County this was Milo Rasmussen (Land Commissioner) and Richard Rollefson (Forester). For St. Louis County this was Jim Larson (Forester).

forests I was able to survey both before and after harvest, and track any changes rather than simply inferring changes from differences in post-harvest comparisons. Paired, unharvested controls on each site allowed me to "zero out" factors other than harvest that could have affected the forest, including weather, disease, or shifts in herbivory or predation. Without these controls, background changes due to these factors could have been mistakenly attributed to the harvest.

Sites.—I conducted this experiment from May-November in 2005, 2006, 2007, and 2008 on 4 sites in northern Minnesota (Figure 1.1). Two sites were at the University of Minnesota's Cloquet Forestry Center ("CFC1" and "CFC3" in text; N 46º42 17.4 m; W 92º 31 26.5). One site was on county forest land in Carlton County ("Carlton" N 46º33 01.4 m; W 92º 45 46.1). A 4th site was selected in St. Louis County ("St. Louis" N 46º49 03.8 m; W 92º 46 47.9), was surveyed for 2 years pre-harvest in the same manner as the other 3 sites, but was dropped from the project when the logging lease holder relinquished the sale and the site was not harvested in time for follow-up surveys. Work began on the 2 CFC sites in 2005, and expanded in 2006 to the St. Louis and Carlton sites. Site and treatment sizes were not standardized. Sites ranged from 5.5 hectares to over 40 hectares (Figure 1.3).

I was given a short list of possible research sites by each land manager, and selected the sites which were the most similar across all 3 landowners. I was limited, however, by the suite that was available to me. While all 4 sites were predominantly aspen cover types, and aspen regeneration was the silviculture goal of each stand, there were ecological differences between the sites. Using the Minnesota DNR's Native Plant Community Classification System (NPC) (MN DNR 2003) I would classify CFC1 as FDn43, CFC3 as FDn33, and Carlton and St. Louis as MHn35 or 45.

Associated research.—In addition to testing the impacts of harvesting slash and CWD, I also addressed two additional questions. First, I tested a novel marking technique in a small mammal mark and recapture study. Second, I compared the performance of 3 commercial small mammal traps. These two research components are included in this dissertation (Chapters 6 and 7). All four research sites including the one in St. Louis County were used in these two studies.

Overview of thesis.—In Chapter 2 I outline the harvest methods employed and fate of the CWD in each of the sites and treatments. Prior to harvest, the stands contained an average CWD load of 465 cm^2/m^2 (SE=49 cm^2/m^2). After harvest, the amount of CWD increased in the slash-retention plots by an average of 422 cm^2/m^2, while slash-removal plots lost on average 29 cm^2/m^2. Prior to harvest there were no statistically significant differences in CWD load between treatments, but after logging all pairings of treatments were different from each other by a statistically significant amount, with retention > control > removal. In addition to gross changes in CWD area, the nature of the CWD in the harvested plots also shifted from initial conditions, with slash-retention plots gaining a disproportionate share of wood in less decayed conditions, while the slash-removal treatments shifted to a CWD base dominated by more decayed wood.

Chapter 3 explores the response of shrubs and aspen regeneration to the harvest. Overall shrub stem counts increased similarly in both harvest treatments though more so in the slash-removal plots. Hazel (*Corylus cornuta)* stem counts increased statistically significantly in the slash-retention treatment, but not the slash-removal treatment. Both treatments show a strong, similar regeneration of aspen stems after harvest.

In Chapter 4 I address the response of herbaceous vascular plants to harvest. Both harvest types exhibited a decline in native forb species cover and an increase in bare ground, cover of invasive plants, and cover of graminoids compared to pre-existing conditions and associated control stands. Changes in native forb cover, invasive forb cover, and bare ground were significantly greater in the slash-removal compared to the slash-retention treatment, but there was not a statistically significant difference between the plots for the increase in graminoid cover. Native species richness of survey plots 2 years post-harvest was not significantly different from pre-harvest values for either treatment.

Chapter 5 deals with the effects of the harvest on the small mammal community. I live trapped small mammals before and after harvest using a combination of box and pitfall traps. In just over 29,000 trap nights I recorded 4,838 captures of 1,794 individual animals. I captured 15 mammal, 6 amphibian, and 1 snake species. Of the 15 mammals, only 5 (*Peromyscus* spp[4], *Myodes gapperi, Sorex cinereus, Blarina brevicauda,* and *Tamias striatus* were captured frequently enough for statistical analysis. Of these only the mice and red-backed voles had large enough numbers for analysis with Mark-Recapture techniques. Overall amphibian captures were low, but trended downward post-harvest in treatment plots. Population estimates of *Peromyscus* spp. were little affected by treatment type or harvest status, while *M. gapperi* showed a positive response to harvest. Shrews (*Sorex sinereus* and *Blarina brevicauda*) and chipmunks (*Tamias striatus*) responded negatively to harvest of both types. The population abundance of the communities overall were generally higher after treatment on the harvest sites than either before conditions or the adjacent controls, with slash-retention plots showing slightly greater gains than slash-removal plots.

As part of the overall project I compared the effectiveness of 3 common live traps (Chapter 6). I paired Sherman, Ugglan, and Fitch-style live traps in a 3 year small mammal experiment in Northeastern Minnesota to compare capture profile, capture rate, and failure rate. Traps could fail either by springing without an animal entering the trap, springing but allowing the animals to escape, or by failing to spring when an animal entered the trap and raided the bait. Traps were baited using cooked oatmeal wrapped in wax paper to enable identification of disturbed bait indicating that an animal had accessed the bait. Each trap type recorded >7,600 trap nights. Ugglan traps had the

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[4] There are potentially two species of *Peromyscus* in the region, *P. maniculatus,* and *P. leucopus.* Field differentiation can be difficult. All mortalities conclusively identified have been determined to be *P. maniculatus*. All mice are assumed to be *P. maniculatus,* however I shall refer to them as *Peromyscus* spp. to avoid inferring more specificity than I can guarantee, since few animals were positively identified using skulls.

highest total catch rate (26.7%) and lowest failure rate (1.5%) of the three traps. Sherman traps had a similarly high capture rate (24.7%), but nearly 1 in 8 trap nights resulted in either a closed but empty trap or a trap that had been entered but not sprung. Fitch-style traps failed (10.8% of trap nights) more than twice as often as they recorded a capture (4.9% of trap nights). Ratios of the three most common species (*Peromyscus* spp., *Myodes gapperi, and Blarina brevicauda)* did not differ between the three traps.

Chapter 7 presents the first application of Visible Implant Elastomer (VIE) tags in a small mammal mark-recapture study. Difficulties in tagging shrews (*Blarina brevicauda* and *Sorex* spp.) and red-backed voles (*Myodes gapperi*) led me to use a technique first developed for fish to mark animals. Individuals of these three species were tagged with Visible Implant Elastomers (VIE). Voles were also tagged with standard ear tags (Monel #1). Analysis of longevity between the 2 systems in voles shows that VIE tags lasted longer and were lost less often in voles than standard ear tags. The VIE tags were also effective in determining whether *B. brevicauda* and *Sorex* had been previously captured. The VIE tags were effective in determining the individuality of voles re-captured without ear tags, allowing for continuous capture histories.

Figure 1.1: Research Site Locations: Location of the 4 field sites in northeastern Minnesota. Duluth, MN provided for reference

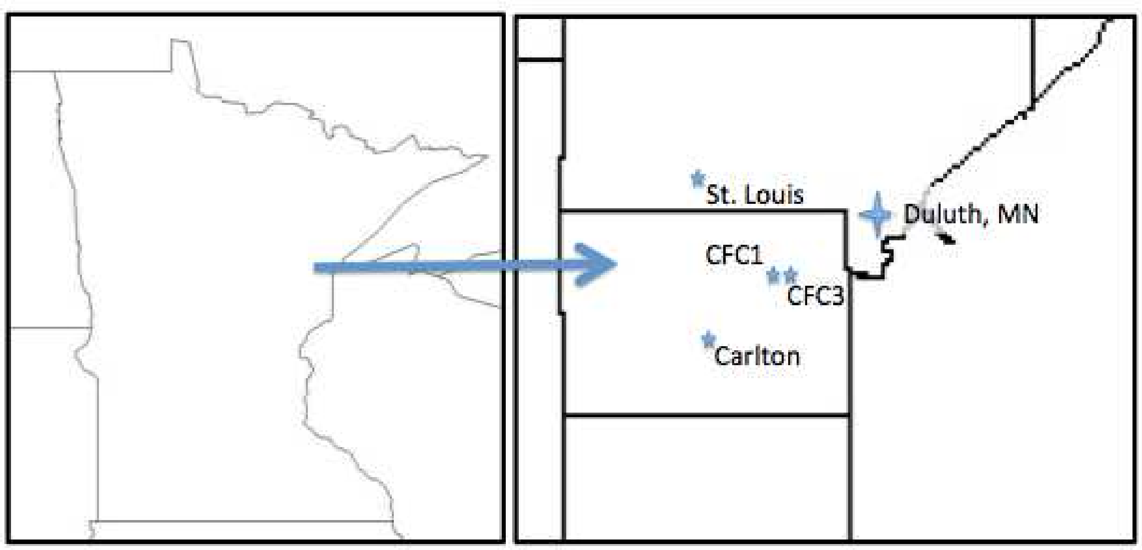


Figure 1.2: Research Site Orientation: Layout of the Carlton Site, as representative of all sites. The star indicates the location of pitfall traps, the parallel lines represent the trap/vegetation grid.

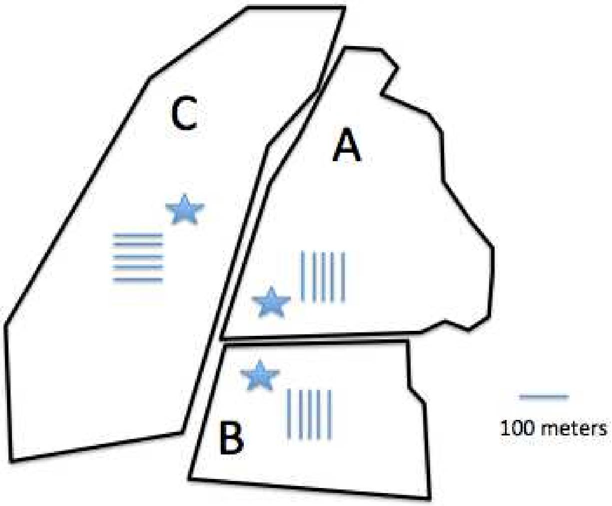
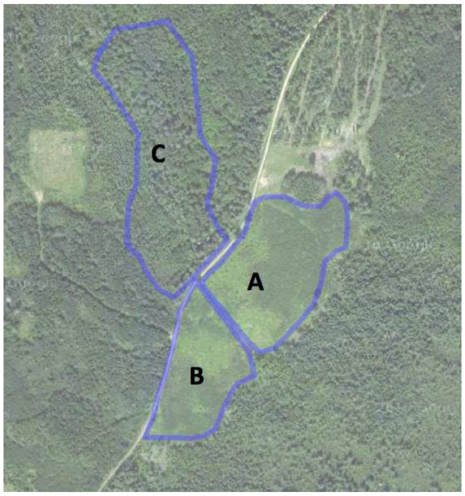


Figure 1.3: Research Sites: Aerial photos (www.maps.google.com) of the 3 principle research sites with treatment boundaries. Sites CFC1 and CFC3 are post-harvest, Carlton is prior to logging. A: slash-retention; B: slash-removal, C: control.

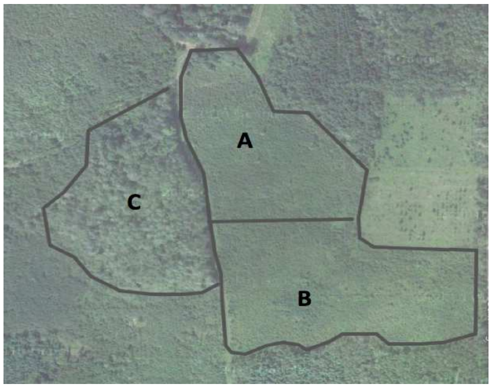


A: 4 hectares

B: 2 hectares

C: 5 hectares

CFC1

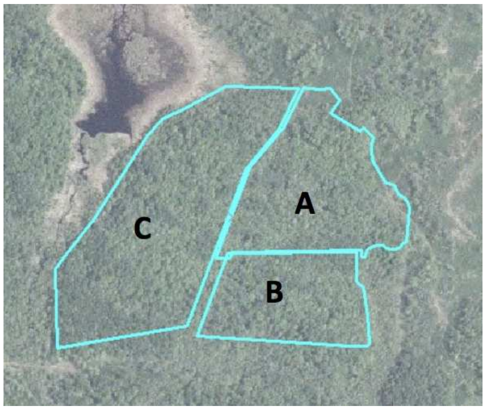


A: 2 hectares

B: 2 hectares

C: 1.4 hectares

CFC3



A: 5.5 hectares

B: 4 hectares

C: 9 hectares

Carlton County

Chapter 2

Harvest Methods and Coarse Woody Debris

Introduction:

The opportunity to produce renewable energy from residue and other previously non-merchantable wood has increased interest in harvesting slash and coarse woody debris. Other work has investigated the economics of the practice (Arnosti et al. 2008). Here I present the harvest methods used, and the resulting differences in CWD between the treatments.

I used three forest sites for this research: Carlton, CFC1, and CFC3. The sites were dominated by aspen (mostly *Populus tremuloides* with some *Populus balsamifera*), but also had considerable red maple (*Acer rubrum*) and balsam fir (*Abies balsamea*). These sites also contained lesser amounts of red oak (*Quercus rubra*, Carlton only) paper birch (*Betula papyrifera*), pine (*Pinus* spp.) and spruce (*Picea* spp.). All forest stands were between 40 and 80 years old at the time of treatment. Management objectives in each site called for a regeneration to young aspen, and after logging the cut treatments were dominated by aspen regeneration.

Each site represents a full BACI replication, with the stand divided into three treatment blocks: clear-cut with slash-retention on site, clear-cut with slash-removal, and control (no cut).

Harvest: Both treatments were clearcut, but varied in the amount of slash retained: no intentional retention (slash-removal) and no intentional harvest (slash-retention). Both treatments used standard heavy machinery logging practices for this region and forest type, with some modifications (outlined below) to achieve the treatment goals. Heavy equipment included a tracked feller-buncher, wheeled skidders to drag trees, and a wheeled delimber.

For the slash-removal treatments, skidders dragged whole trees to a central landing site off of the treatment site for de-limbing. This approach was prescribed by the forest manager under the assumption that the effects of tree dragging would inhibt growth of *Corylus cornuta* on the site (R. Severs personal communication). Loggers were also permitted to harvest snags and existing CWD at their discretion on these treatments.

For the slash-retention treatments, the feller buncher piled trees as it cut into small, scattered piles to which the delimber drove. In this way slash was not re-distributed from a central processing area, but left in place throughout the process, reducing machinery impact on the treatment by avoiding the backhaul trips. Loggers were directed to leave snags and existing CWD in place on slash-retention plots.

A similar process was used for the Carlton site, though here there was less whole-tree skidding overall and the loggers did not remove as much slash from the slash-removal treatment as was called for in the lease, but left scattered piles of limbs and small trees. I removed slash piles from the trapping grid using a chainsaw, all terrain vehicle, and a small wheeled trailer (Figure 2.1). The initial logging was done on frozen ground with > 0.5 meters of snow. The cleanup was done in spring on wet ground. I took care to remove the excess material with as little impact as possible, and consciously chose to keep the ATV and trailer on designated paths.

Logging removed all standing live trees from the cut sections of the Cloquet Forestry Center, and all standing dead trees (snags) and downed logs that could be hauled from the slash-removal treatment. The same prescription was used on the Carlton site, with the exception of oak (*Quercus*) left at the direction of the county forester and a small number of trees situated too close to the standing oaks to safely remove (n < 2 trees retained/ha).

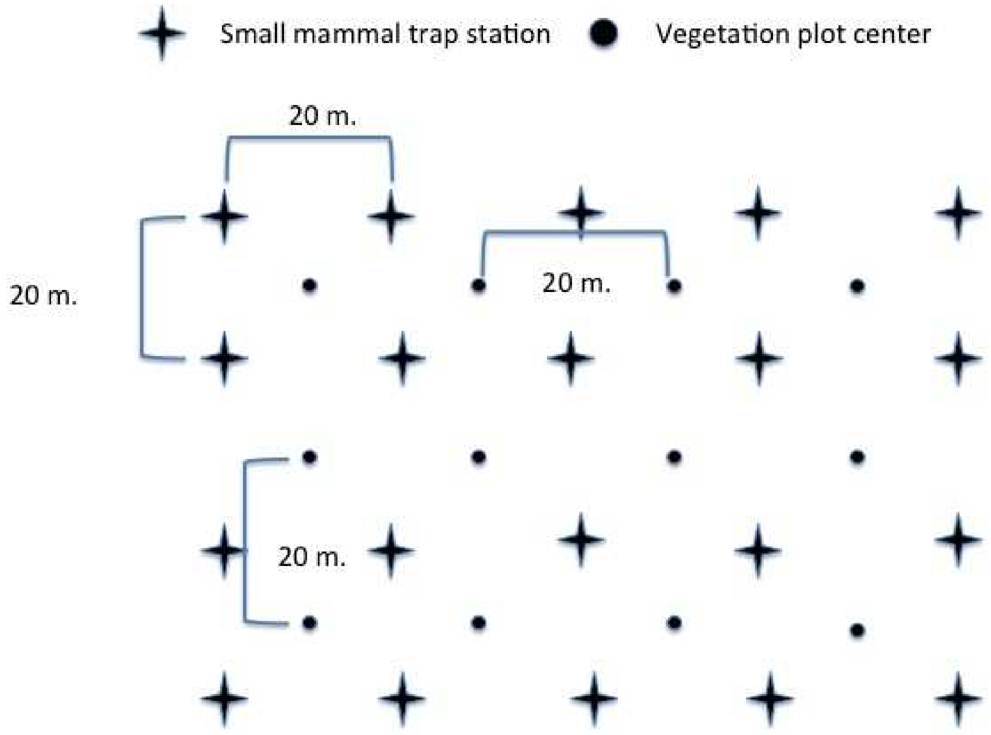
There was a clear effort made to leave or remove slash in the appropriate treatments on the Cloquet site, and differences between the two cut treatments were stark, and immediately noticeable. Differences between the Carlton treatments were not immediately obvious due to lack of compliance by the logger to the harvest prescription, but extra removal resulted in an approximation of the desired results.

Methods

Siting of vegetation plots was driven by the small mammal trapping grid. As part of the overall project, I established a small mammal trapping grid in each treatment area of each stand (Figure 2.2). The initial starting point was selected at random, and from this point I systematically laid out a 4 X 5 grid, with 20 meter spacing between both rows and columns. Twelve permanent vegetation sample plots were laid out in a systematic manner nested inside these grids, with 20 meter spacing. Vegetation plots were situated so that each plot was equidistant from the four nearest trap stations. In this way the vegetation plots were sited to avoid impacting the plots during the normal course of

Figure 2.1: Slash pile from Carlton slash-removal treatment. This is the extra slash hand removed by the author in addition to that done by the loggers. Pile measures approximately 1.5 meters X 5 meters X 10 meters.

Figure 2.2: Research Plot Layout: Standard layout and spacing of small mammal trap stations and vegetation/CWD plot centers.



trapping. Plots were only visited when sampled, and all other foot traffic was kept away from the plots.

I marked plot centers with rebar driven into the ground, allowing for exact relocation with a metal detector. Pins that were pulled from the ground by logging equipment and could not be confidently re-located were dropped from analysis. From these central points I swept a 5.64 meter radius (100 m^2) plot with a thin wire cable.

I surveyed coarse woody debris (CWD) in the harvest plots in late fall prior to winter logging, and in the spring following harvest. The two surveys were roughly six months apart. I surveyed the control plots once, concurrent with the post-treatment survey for the harvested sites. There may have been some natural accumulation or decomposition of the control plots over the winter, but limited time and resources required me to prioritize the harvest plots during the narrow window between vegetation senescence and the start of logging.

I measured each piece of CWD at least 5cm in diameter inside each plot. For each piece I recorded total length within the plot (to the nearest 5cm) and midpoint diameter. CWD was also categorized into a decay scale modified from Maser et al. (1979). This scale was initially developed for use with conifers and required some modification for northern hardwoods (see Figure 2.2 for description). Briefly, as wood decays it progresses from intact stems with bark attached (Class 1), through logs that are nearly all soil, and may be only a raised bump of moss with chunks of rotted wood underneath (Class 5). Because the categorization was subjective, in the final analysis the categories were collapsed to “hard” (scores < 3, recently dead), “mid” (scores 3, 3.5, decay becoming obvious), soft (scores 4-5, advanced decay) and chunks (score 6). The chunk category was added to incorporate more decayed stems that had become crushed and scattered from vehicle traffic. It was only present in the slash-removal treatment post-harvest.

As a log decays through the classes it becomes less circular, and in the final stages a diameter measurement parallel with the ground would not be representative of the total volume of wood (as an example, a class 5 log may be 10cm across, but only 4cm high). For this reason planar area (length X diameter) was used rather than volume.

RESULTS

A summary of the pooled CWD results for the three sites surveyed is presented in Table 2.1. There were few differences between the treatments prior to harvest in regards to CWD (Table 2.1, Figure 2.4). After harvest total CWD (planar area) increased in all but two slash-retention plots and declined in more than half of the slash-removal plots (20/36). Final CWD densities ranged from 300-1775 cm^2/m^2 (mean=1000 cm^2/m^2, SE= 90 cm^2/m^2) in slash-retention plots, and 20-1060 cm^2/m^2 (mean=365 cm^2/m^2, SE= 58 cm^2/m^2) in slash-removal plots. This compares with pre-harvest average density of 465 cm^2/m^2 (SE=49 cm^2/m^2) for these plots. The average slash-retention plot saw an increase in CWD area of just over 42,174 cm^2 (422 cm^2/m^2). Slash-removal plots lost on average just over 2,900 cm^2 (29 cm^2/m^2). The change in plot CWD amount was significantly different for the two cut treatments (t = -7.59, df = 66.73, $p < 0.001$). The bulk of the difference between the plots was the addition of less decayed wood (stages 1 and 2) to the slash-retention plots, and the net loss of same from the slash-removal plots. Both treatments saw a similar increase in CWD of more advanced decay stages.

Although the number of pieces of CWD increased in the harvested treatments, each piece of CWD was on average shorter and thinner than the existing CWD prior to harvest. This relationship held true for both slash-retention and slash-removal treatments, though the reduction in median stick length was significantly greater for slash-removal plots (median change 134 cm) than for slash-retention plots (median change 72 cm) (t = -2.7, df = 67.37, p-value = 0.009). The sheer number of pieces of CWD was also much greater in the slash-retention plots (52 sticks/plot) than in the slash-removal plots (26 sticks/plot) (t = 6.65, df = 62.61, $p < 0.001$).

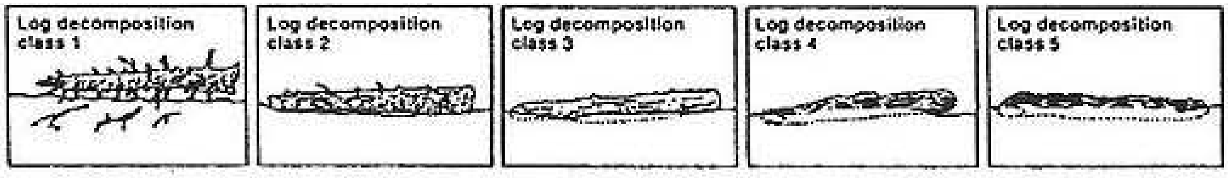
As mentioned above, the two treatments differed in how the ratios of the three lumped classes (hard, mid, and soft) to the total CWD area changed pre to post harvest (Table 2.2). In general the portion of total CWD that was of the more decayed "soft" classes increased in the slash-removal treatments and decreased in the slash-retention plots, while the opposite relationship was found for the harder, more intact debris.

Discussion

As expected from the design of the study and the intent of the slash-removal harvest, the amount of CWD was starkly different between the treatments. The nature of the CWD also differed in each treatment before and after treatment. In both treatments, median stick length dropped, as single, long logs were broken by machinery, or supplemented with new, smaller logs. Median diameter also dropped as the new additions were smaller

Figure 2.3 Coarse Woody Debris Decay Classes

Original citation: Maser et al. 1979.
Figure from http://www.ilmb.gov.bc.ca/risc/pubs/teecolo/fmdte/cwdcom.htm



	Class 1	Class 2	Class 3	Class 4	Class 5
Wood Texture	Intact, hard	Intact, hard to partly decaying	Hard, large pieces, partly decaying	Small, blocky pieces	Many small pieces, soft portions
Portion on Ground	Elevated on support points	Elevated but sagging slightly	Sagging near ground, or broken	All of log on ground, sinking	All of log on ground, partly sunken
Twigs < 3 cm (if originally present)	Present	Absent	Absent	Absent	Absent
Bark	Intact	Intact or partly missing	Trace	Absent	Absent
Shape	Round	Round	Round	Round to oval	Oval
Invading Roots	None	None	In sapwood	In heartwood	In heartwood

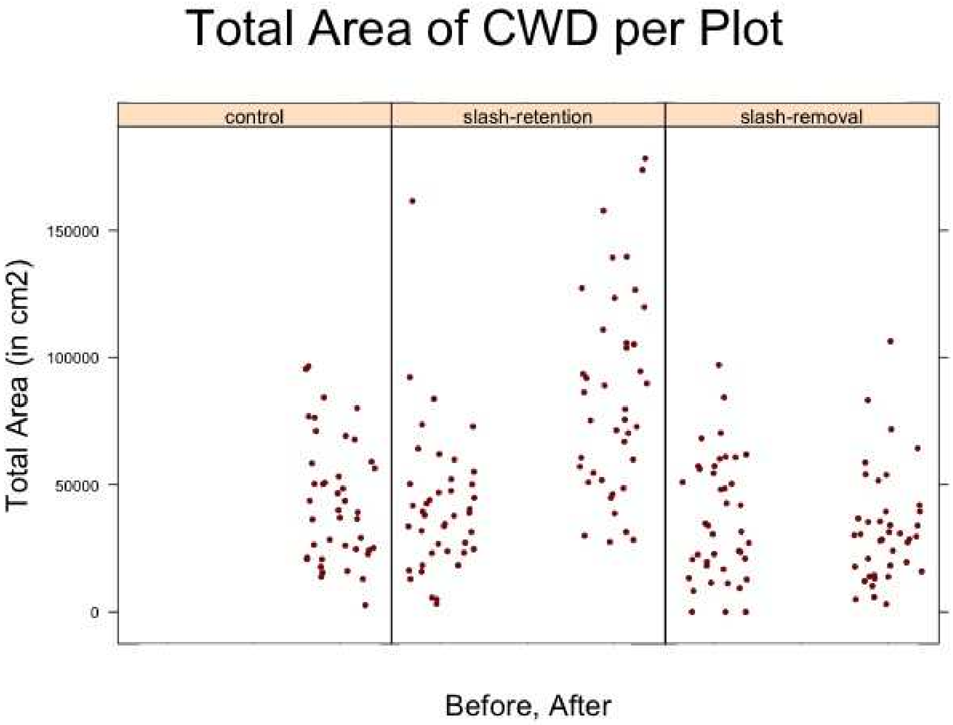
Table 2.1. Mean mid-planar area of Coarse Woody Debris (CWD). Treatments include: Control (no harvest), Slash Retain (clearcut with slash retained on site) and Slash Remove (clearcut with whole tree skidding and/or mechanical removal of slash). Statistical significance of changes within each treatment due to harvest is noted, and designated as ns (not statistically significant, $p > 0.1$), + (statistically significant at $p<0.1$), * ($p<0.05$), ** ($p<0.01$, *** ($p<0.001$). Comparisons across treatments are noted where one or more treatments has a statistically significantly stronger response ($p <0.05$) than the others. Values between treatments across a year are significantly different ($p<0.05$) when denoted with differing letters.

Data for the Control plots was only collected once (corresponding to the after period for the harvest treatments).

Treatment		**Control**	**Retain**	**Remove**
Total	Before	43242 cm^2 *a* (3699 cm^2)	42822 cm^2 *a* (4413 cm^2)	38680 cm^2 *a* (4067 cm^2)
	After	43242 cm^2 *a* (3699 cm^2)	84996 cm^2 *b* (6275 cm^2)	32929 cm^2 *c* (3636 cm^2)
	Change		↑ 42174 cm^2 *** *a* (4698 cm^2)	↓ 2907 cm^2 ns *b* (3527 cm^2)

Decay class “hard”	Before	9438 cm^2*a* (1858 cm^2)	12614 cm^2 *a* (1722 cm^2)	12609 cm^2 *a* (2220 cm^2)
	After	9438 cm^2*a* (3699 cm^2)	43338 cm^2*b* (5014 cm^2)	5707 cm^2*a* (844 cm^2)
	Change		↑ 30724 cm^2 ****a* (4698 cm^2)	↓ 6902 cm^2 ** *b* (2301 cm^2)
Decay class “mid”	Before	13696 cm^2*a* (2056 cm^2)	14832 cm^2 *a* (2650 cm^2)	16749 cm^2 *a* (3002 cm^2)
	After	13696 cm^2*ab* (2056 cm^2)	18736 cm^2*a* (1958 cm^2)	10741 cm^2*b* (1720 cm^2)
	Change		↑ 3904 cm^2 ns *a* (2251 cm^2)	↓ 6008 cm^2 + *b* (2344 cm^2)
Decay class “soft”	Before	22007 cm^2*a* (2497 cm^2)	18312 cm^2 *ab* (2608 cm^2)	14589 cm^2 *b* (2038 cm^2)
	After	22007 cm^2*a* (2497 cm^2)	27437 cm^2*a* (4226 cm^2)	20051 cm^2*a* (3046 cm^2)
	Change		↑ 9125 cm^2 **a* (3589 cm^2)	↑ 5462 cm^2 ns *a* (3572 cm^2)

Figure 2.4: Plot of CWD area in sample plots before and after treatment. Control plots were only sampled once. A Welch t-test of difference in change of area between slash-retention and slash-removal: $p < 0.0001$



than the pre-harvest CWD. Smaller, shorter, CWD may be less useful than larger, longer pieces for small mammals as runways and refugia (Maser et al. 1979).

The shorter CWD may actually increase food availability in the short term by hosting a greater density of insects and fungi though a greater proportion of exposed surface area. In the longer term, as this debris decomposes more quickly than longer, larger, and less degraded sticks would have, the overall availability of food would likely decrease as well.

Coarse woody debris is important for small mammals as a microhabitat for cover and also as habitat for insects, fungi, and plants that in turn serve as a food source for the small mammals. As CWD decays, its role in the ecosystem changes (Franklin et al. 1987; Maser et al. 1979; Maser and Trappe, 1984; Pyle and Brown, 1999). Young, harder CWD can serve as refugia for small vertebrates and protect against erosion. More decayed but still intact wood can retain water better than surrounding mineral soil and provide habitat for bryophytes, ferns, and other vascular plants. Even older debris—classes 4 and 5 in this study—can persist in the forest for years and continue to hold water and allow for rooting of plants. This debris is of less use as cover for larger vertebrates, but may still be useful for smaller and especially fossorial and semi-fossorial animals such as herpetofauna, shrews, and moles. Finally, CWD may serve as a carbon sink, delaying the release of carbon into the atmosphere.

Perhaps the starkest difference in CWD response between the treatments is the case of the less decayed, more intact “hard” CWD. This wood still has its bark and

Table 2.2 Mean fraction of total CWD for each decay class by plot, treatment, and harvest status. Data for the Control plots was only collected once (corresponding to the after period for the harvest treatments). Difference is the resultant p value found in paired t-tests. Inter-treatment comparisons are noted where one or more treatments has a significantly stronger response (p <0.05) than the others. Values between treatments across a year are statistically significantly different when noted with differing letters.

Treatment		**Control**	**Retain**	**Remove**
	Before	21.8% *a*	30% *a*	24.6% *a*
Class Hard	After	21.8% *a*	48.7% *b*	15.2% *a*
	Difference		↑$p < 0.01$	↓$p=0.07$
	Before	36.1% *a*	33.5% *a*	34.5% *a*
Class Mid	After	36.1% *a*	22.9% *b*	34.1% *a*
	Difference		↓$p<0.01$	↓*NS*
	Before	42.1% *a*	36.4% *a*	40.5% *a*
Class Soft	After	42.1% *a*	28.4% *b*	50.6% *a*
	Difference		↓ *NS*	↑ *NS*

smaller branches intact, and will persist on the landscape for longer. This class of CWD greatly increased in the slash-retention plots as new wood was added to the pool from the harvest, but decreased in the slash-removal treatments as a result of harvest of not only the pool of potential CWD in the form of branches and tops removed, but from the deliberate harvest of the pre-existing pool of CWD as well. As the fraction of the CWD in the less decayed classes increases in the slash-retention plots and decreases in the slash-removal plots, we can expect this difference to persist and even strengthen over time as the more decayed wood gradually decomposes and is replaced at a lower rate in the slash-removal plots.

Chapter 3

Response of Shrubs and Aspen Regeneration to Coarse Woody Debris Harvest

Introduction:

This chapter examines the differences between the two slash retention levels on aspen regeneration and the general shrub and tree community, with a special focus on beaked hazel (*Corylus cornuta*). Hazel is ubiquitous across the region, and typically constitutes 60-90% of stem density in older stands (Kurmis and Sucoff 1989). On each site, vigorous regeneration of aspen or aspen and birch was the primary management objective. *Populus* species such as aspen regenerate after disturbance—either fire or harvest—via clonal root suckers (Frey et al. 2003). Initial stem densities decrease as the stands age through self-competition mortality (Perala 1977, Perala et al. 1999). This process is known as the "stem exclusion phase." High hazel density may suppress initial aspen sucker density and vigor (Mundell, et al. 2007; Ron Severs, personal communication).

Whole-tree skidding and slash-removal was initially prescribed on the two Cloquet Forestry Center sites to scarify the soil, damage or remove existing hazel (*C. cornuta*) and prevent its regeneration. Logging slash may also inhibit aspen regeneration (Doucet 1989). Trees and shrubs are expected to regenerate at greater densities on the slash-removal plots.

Methods

Sites: I selected two sites for full analysis of shrubs and tree regeneration: CFC1 (NPC[5] FDn43) and Carlton, NPC MHn35). The second site on the Cloquet Forestry Station (CFC3) was not chosen for this part of the study due to low plant alpha diversity, only a single shrub species (*Diervillia lonicera*) dominated and the pre-harvest canopy was almost pure aspen (*Populus* spp).

Vegetation sampling: As described in Chapter 2, on each site I established a 4 X 5 small mammal trapping grid with 20 meter spacing in each treatment area of each stand. Permanent vegetation sample plots were laid out in a systematic manner nested inside these grids, with 20 meter spacing. From these central points, I swept a 2.82 meter radius (25 m^2) plot with a thin wire cable.

On each 25m^2 plot I did a complete stem count for each taxon. Shrubs were identified to species in most cases, though analysis is done on shrub genera. Particularly difficult genera such as *Amalanchier* and *Salix* were identified only to the genus. Stems were counted ~5 cm above mineral soil to avoid the need to remove litter or herbaceous vegetation. As a result, some spreading shrubs such as *Dirvelia lonicera* may have received a higher number of stem counts than there were actual individual plants, but sampling efficiency was greatly improved. Stump sprouting regrowth from logged trees was also counted individually, though each stem was a clone of the parent tree and few of the stems would survive.

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[5] The Minnesota Department of Natural Resources' Native Plant Communities Classification System: http://www.dnr.state.mn.us/npc/classification.html

Stems were categorized by height class: up to 0.5 m, between 0.5 and 1.5 m, and greater than 1.5 m. Tree saplings were considered shrubs if they were less than two meters tall. Trees greater than two meters tall prior to harvest were classified as trees and counted separately (not reported here). All regenerating stems after harvest were included in this survey, regardless of height.

Sampling was conducted between July and September of each year, with the bulk of sampling done in July and early August. Care was taken to balance the sampling effort across a site between treatments to avoid observation date from affecting one treatment differently than another. Thus, while it may have taken some time for an entire site to be surveyed each year, the three treatments within each site did not differ in the timing of the survey effort.

Nomenclature follows the State of Minnesota Vascular Plant Checklist (MNTaxa, MN DNR 2013).

Analysis: Shrub stem counts were compared with paired T-tests or Wilcoxon signed rank tests before harvest and one and two years post-harvest. Differences within treatments pre-post harvest were analyzed with T-tests. I used linear models to determine the effect of CWD area and hazel density on aspen regeneration. I used R (R 2.13.1 for Mac OS X GUI 1.40) for all statistical analyses.

RESULTS

Few differences in shrub density existed between the treatments prior to harvest (Table 3.1). Although there was heterogeneity across the sites, especially in the presence or absence for less common shrubs, these differences were generally not statistically significant. Other pre-harvest differences between treatments not included in Table 3.1 are noted in the text.

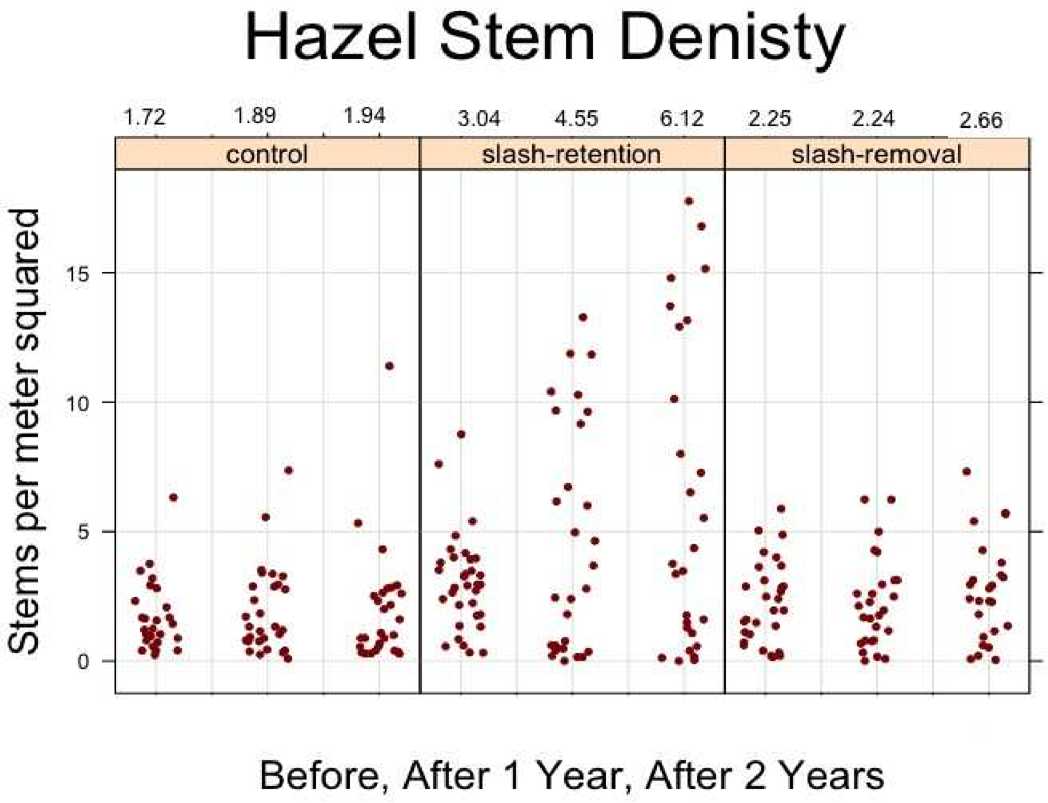
Both harvest treatments had a negative effect relative to pre-harvest conditions on stem desnity for *Acer* spp. and a positive effect on *Populus* spp., *Rubus* spp. and total shrubs. There was little difference between the two harvest treatments in shrub and tree response for most species, with the exception of beaked hazel (*Corylus cornuta)* which doubled in the slash-retention treatments plots, but remained essentially constant in the slash-removal plots (Table 3.1).

Hazel: Hazel counts in the slash-retention treatment increased relative to the pre-harvest condition for both the one and two year post treatment counts (t = 1.87, df = 35.24, p= 0.07 and t = 2.5, df = 27.98, p = 0.019). The increase in the slash-retention treatment was significantly different than changes in the control and removal plots, each of which remained essentially constant (Year 2: control-retention -> t = -2.72, df = 25.76, p = 0.01; Year 2 removal-retention -> t = -2.78, df = 25.45, p = 0.01; Figure 3.1). While the overall sum of hazel was not statistically significantly different for the slash-removal plots after harvest, the smallest size—those shrubs under 0.5 meters—increased by an average of 40 stems per plot (1.6/m^2). The increase in the smallest size class was significant (t = -4.59, df= 28.71, p <0.0001) when compared to pre-harvest conditions, but the increase of 40 stems/plot was also significantly less than the increase of the 93 stems/plot (3.7/m^2) of this size class on average in the slash-retention treatment (t =2.56, df =37.94, p= 0.015).

Table 3.1 Density (stems/m2) of selected shrub species: 1) before harvest; 2) one year after harvest; and 3) two years after harvest. Treatments include: Control (no harvest), Slash-retention (clearcut with slash retained on site) and Slash-removal (clearcut with whole tree skidding and/or mechanical removal of slash). Standard errors are in parenthesis. Letters following values indicate relationships between treatments; treatments that share a letter do not differ by a statistically significant margin, those with differing letters do (p <0.05). Change between years within a treatment are labeled as ns (not statistically significant, p > 0.1), * (p<0.1), ** (p<0.05), *** (p<0.01).

Species	**Status**	**Control**	**Retain**	**Remove**
Hazel	**Before**	1.72 (0.269) a	3.04 (0.325) b	2.25 (0.296) ab
	After 1	1.89 (0.395) ns a	4.55 (0.819) * b	2.24 (0.336) ns a
	After2	1.94 (0.429) ns a	6.12 (1.147)** b	2.66 (0.386) ns a
Aspen	**Before**	0.22 (0.059) a	0.16 (0.042) b	0.24 (0.045) a
	After 1	0.34 (0.083) ns a	3.54 (0.559) *** b	5.22 (0.867)*** b
	After 2	0.34 (0.09) ns a	3.54 (0.551) *** b	4.87 (0.522) *** b
All stems	**Before**	6.9 (0.938) a	7.95 (0.538) a	8.39 (0.753) a
	After 1	8.43 (0.69) ns a	11.8 (1.21) *** b	12.05 (1.0) ** b
	After 2	11.58 (1.086) ** a	16.65 (1.604) *** b	15.22 (0.871) *** b
Rubus	**Before**	0.2 (0.066) a	0.93 (0.268) b	0.53 (0.124) b
	After 1	0.73 (0.158) *** a	1.0 (0.219) ns a	1.07 (0.221) ** a
	After 2	0.87 (0.228) *** a	2.71 (0.446) *** b	2.95 (0.451) *** b

Figure 3.1 Hazel stem density (stems/m^2) in sample plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean of plots for that sampling period.



Densities of hazel for the 0.5-1.5 m. size class differed between the harvest treatments on the CFC1 site (2.76 stems/m^2 vs. 0.93 stems/m^2, t = 2.0791, df = 14.22, $p = 0.056$) but did not differ for the combination of the two sites (1.5 stems/m^2 vs. 0.8 stems/m^2, $t = 1.32$, df = 33.39, $p = 0.2$). Hazel densities of the highest class (> 1.5 meters) were not different between the harvest plots (0.04 stems/m^{2,}, $t = 0.31$, df = 25.43, $p = 0.76$).

Populus: Aspen and balsam poplar trees (*Populus* spp) increased in both harvest cuts (Figure 3.2). *Populus* stems increased from an average of 2.7 stems/plot (0.1/m^2) in slash-retention plots before harvest to 86.6 stems/plot (3.4/m^2) by two years after harvest (t = -5.45, df = 23.12, p= <0.001) and from 7.3 stems/plot (0.3/m^2) to 116.6 stems/plot (4.7/m^2) in the slash-removal plots (t = -7.73, df = 21.38, p= <0.001). The difference in increase between the harvest types was not statistically significant (t = 1.23, df = 43.86, p = 0.23).

There was a difference between the harvest treatments for the middle size class. Stems between 0.5 and 1.5 meters in height had an average density two years post-harvest of 1.82 stems/m^2 in the slash-retention plots, and 2.98 stems/m^2 in the slash-removal plots ($t = -2.49$, df = 44.87, $p = 0.016$). The trend was driven by the Carlton site and its density gap between treatments of 1.72 stems/m^2 ($t = -2.91$, df = 23.54, $p = 0.008$). The difference of 0.5 stems/m^2 on the CFC1 site was not statistically significant ($t = -0.73$, df = 17.73, $p = 0.47$). Both treatments averaged one stem/m^2 of the tallest heights class (>1.5 meters).

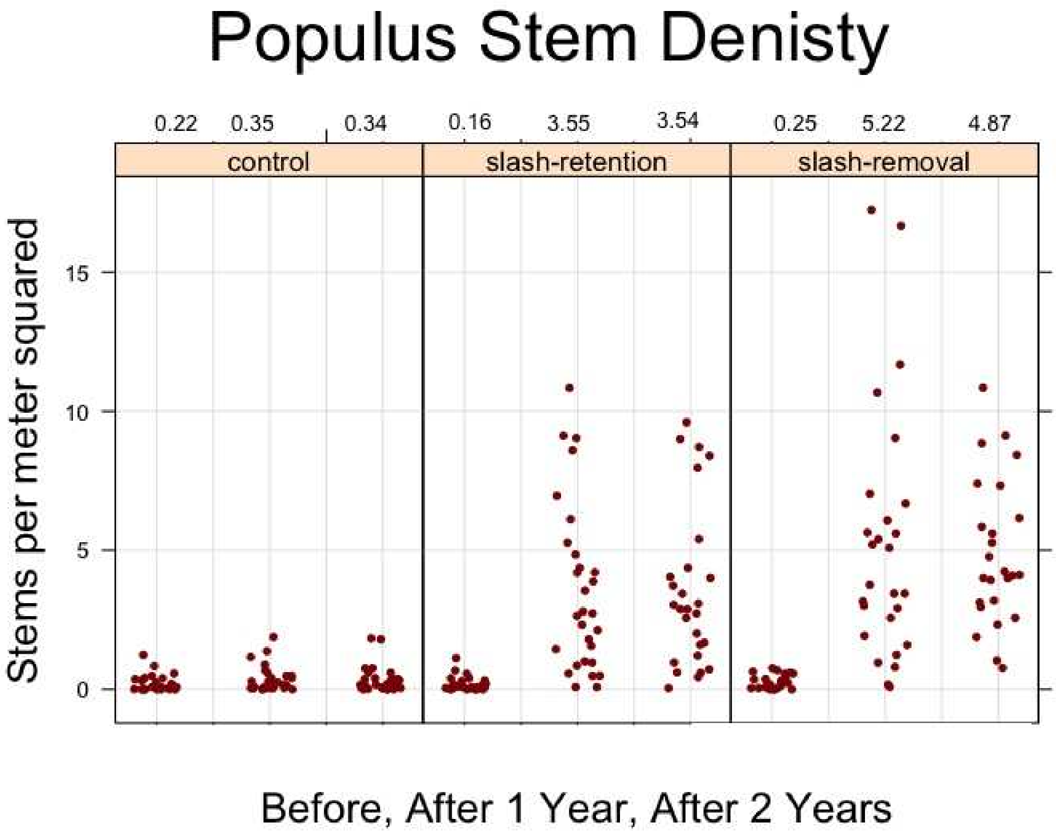
When aspen stem density across both treatment types was regressed against area of CWD there emerged a moderate negative correlation between stems/plot and area of CWD. The sum of a plot's CWD of each hardness class (hard, mid and soft) explained 31% of the variation in stem density (Adjusted R-squared: 0.31 *F*= 5.64 df= 3/28, p = 0.004). Debris of the "hard" class alone explained 17% of the variation (Figure 3.3, Adjusted R-squared: 0.1765 *F*=: 8.93 on df=1/36, *p* = 0.005).

The trend was especially evident when only the slash-retention treatment was analyzed; the strength of the relationship increased to almost 43% (Adjusted R-squared: 0.427 *F* =5.22, df = 3/14, *p* = 0.01). Aspen regeneration on the slash-removal treatments was not affected by the lower levels of CWD found on those plots (Adjusted R-squared: 0.104, *F*= 1.5, df=3/10, *p* = 0.27).

There was no correlation between aspen regeneration and density of either hazel (Figure 3.4; Adjusted R-squared: -0.02, *F* = 0.22, df=1/39, *p* = 0.64) or raspberry stems (Adjusted R-squared: -0.02 *F*= 0.097, df= 1/39, *p* = 0.76).

Maple: Maple (*Acer* spp.) stems decreased across the majority of both slash-removal plots (n = 13 of 24) and slash-retention plots (n = 14 of 24), however the slash-removal plots lost on average 64 stems/plot (2.56/m^2) after two years, versus 24.2 stems/plot (0.97/ m^2) on average in the slash-retention plots. This difference was fairly large, but not quite statistically significant (*t* = -1.80, df = 40.88, *p* = 0.08). Changes in *Acer* spp. were largely driven by the higher pre-harvest densities on the Carlton site. Prior to harvest the CFC1 cut plots averaged 24 (0.96/ m^2, retention), 5 (0.2/m^2, removal) and 11

Figure 3.2 Aspen (all *Populus* species) stem density (stems/m^2) in sample plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean of plots for that sampling period.



(0.44/ m^2, control) stems per plot, while the Carlton plots had 105 (4.2/m^2, retention), 205 (8.2/m^2, removal) and 215 (8.6/m^2, control) on average.

The Carlton site showed the same pattern the combined sites did, but slightly more strongly. The slash-removal plots showed a 35% loss of *Acer* stems on average, compared to a drop of 18% in the slash-retention plots (Wilcox rank sum: 28, p= 0.035).

Rubus: Raspberries and dewberries increased by a statistically significant amount in all treatments, including the unharvested controls (Figure 3.5). Because *Rubus* counts were considerably lower in control plots than harvest plots prior to harvest, stem counts were log transformed to better reflect relative changes rather than absolute changes in denisty. The harvest treatments showed larger increases in stem count relative to the control (Control-Retention: t = -2.42, df = 46, p-value = 0.02; Control-Removal: t = -1.86, df = 43.5, p-value = 0.07). The harvest plots showed similar relative increases (t = -0.735, df = 43.46, p-value = 0.47)

Blueberries: Two years after harvest *Vaccinium* spp. stem counts had decreased in both harvest treatments (Retention: 63 -> 41 (2.5->1.6 /m^2); Removal: 58 -> 1 (2.3-> 0.04/m^2)). Blueberries increased over the same time span in the control treatments (45 -> 90 (1.8->4.6/m^2)). Although the difference in loss was not statistically significant between the harvest treatments due to the low beginning numbers per plot, there was a near extirpation of *Vaccinium* from the slash-removal treatments.

Figure 3.3: Relationship between hard CWD and aspen regeneration

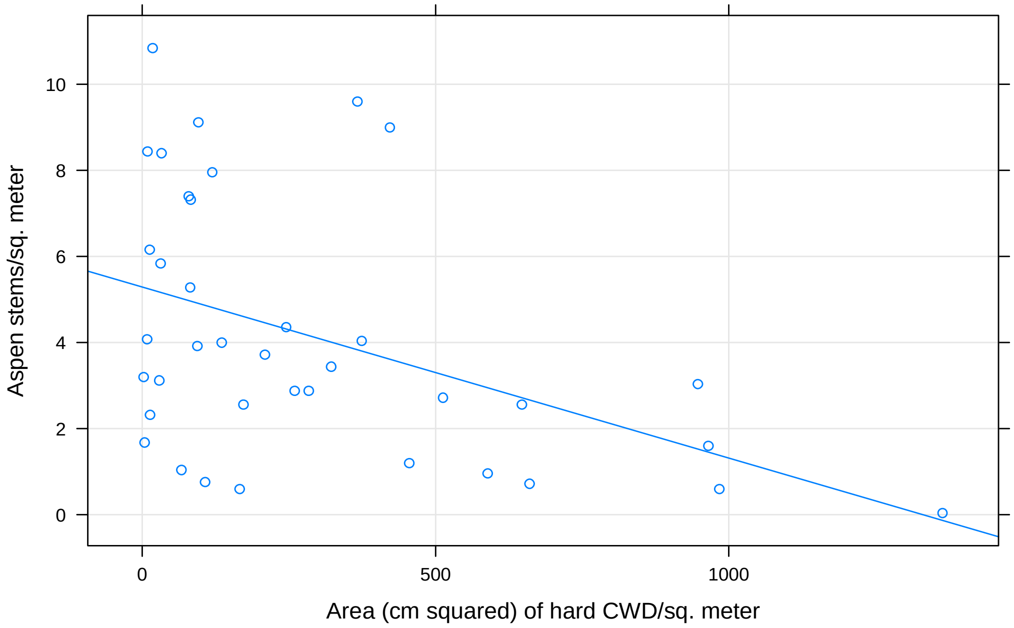


Figure 3.4: Relationship between hazel stem density and aspen regeneration

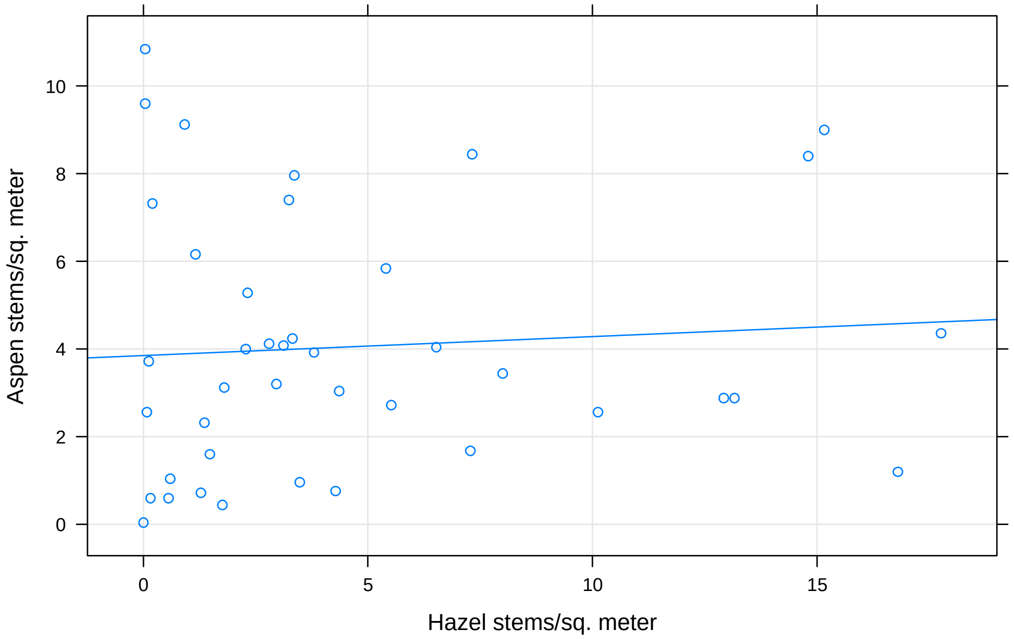
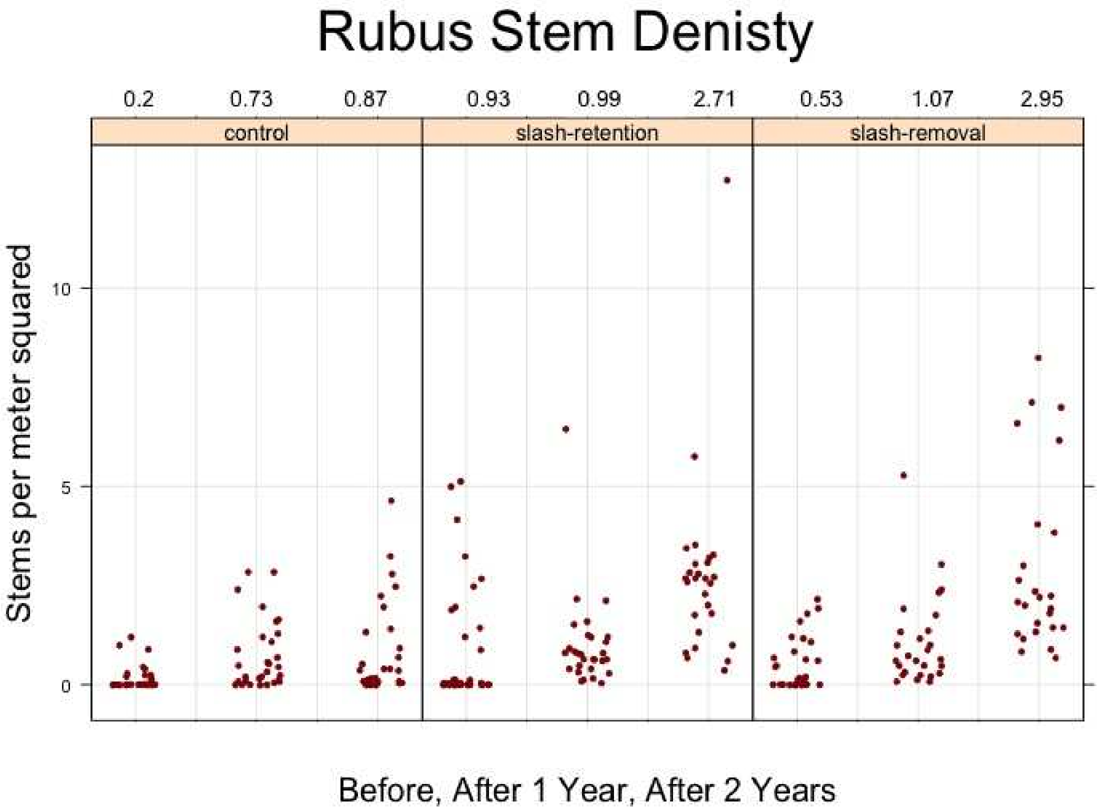


Figure 3.5 Density of *Rubus* spp. (stems/m^2) in sample plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean of plots for that sampling period



Other shrubs: The woody understory was dominated by hazel and maple seedlings before harvest and by aspen regeneration and hazel after cutting, however, another 17 genera of native shrubs and vines were recorded in at least one treatment. These shrubs included species of *Cornus, Prunus, Salix, Ribes, Amalancheir, Physocarpus, Sambucus, Viburnum, and Lonicera.* Not all of the species were present on every plot or even every treatment, but most treatments had at least a few examples of each of the above.

The most ubiquitous shrub after hazel was northern bush honeysuckle (*Diervilla lonicera*), occurring on over half of the plots surveyed. None of the treatments saw a notable increase or decrease in *D. lonicera* counts after one year, and only the removal treatments showed a statistically significant change in stem counts after 2 years (from 9.5 stems/plot to 29 stems/plot (0.4->1.2 /m^2): t = -2.27, df = 28.25, p = 0.031). The increase of 20 stems/plot was not different than the 9.7 stems/plot increase observed in the slash-retention plots (t = 1.28, df = 35.4, p = 0.21) but was statistically significant when compared to the control treatment and its 2.5 stems/plot increase (t = -2.52, df = 22.59, p = 0.019).

Willow (*Salix* spp.) were absent from all control and slash-retention plots prior to harvest, and were found on two slash-removal plots (n=7). After harvest, *Salix* were still absent from control plots, but were now found on 12 of 24 slash-retention plots and 10 of 22 slash-removal plots. While *Salix* spp. increased in both treatments, there was a noticeable but not quite significantly greater increase in the slash-retention plots than the slash-removal plots (an average of 6.8 stems/plot (0.3/m^2) versus 2.3 stems/plot (0.1/m^2): t = -1.83, df = 30.47, p = 0.077).

Dogwoods (*Cornus* spp.) were never found on slash-removal plots prior to or after harvest, but were a relatively common shrub in the control and slash-retention plots before harvest. Dogwood counts did not decrease after two years on the control plots, but after harvest only a single dogwood stem was found in the slash-retention plots.

Other than the results detailed above, no single taxa of shrubs showed a statistically significant response to harvest. When all shrubs other than hazel and *D. lonicera* were lumped, a positive relationship was found. Prior to harvest these less common shrubs averaged 21 stems/plot (0.8/m^2) in the slash-retention treatment, 15 (0.6/m^2) in the slash-removal, and 15 (0.6/m^2) in the control treatments. Two years after harvest these counts were 48, 28.5, and 16 stems/plot (1.9, 1.1, 0.6/m^2) respectively. The increase was significant for the slash-retention plots (t = -3.05, df = 32.66, p = 0.005), and was not quite significant for the slash-removal plots (t = -1.856, df = 30.9, p = 0.073). The increase in the slash-retention plots was not greater than that in the slash-removal plots by a statistically significant amount. (t = -1.39, df = 43.5, p = 0.17).

DISCUSSION:

The study sites were pre-planned timber sales, and in each case the harvest was intended to regenerate aspen. Work on aspen over the last two decades indicates that initial sucker density may affect final stand productivity, with higher initial densities leading to more vigorous suckers (Palik et al 2003). Contrary to the original hypothesis, aspen regeneration appeared little affected by treatment type. There was a non-statistically significant difference of about one stem/m^2 for all sizes class summed, but both treatments averaged the same one stem/m^2 of the tallest height class. I believe it is unlikely that the overall difference will affect final tree density after the stem exclusion phase, especially in light of the lack of a difference in the stems of the tallest height class. Aspen stem mortality during stand maturation is high, and mostly through intraspecific competition (Perala et al. 1999). Clearly a density of >4 trees/m^2 as found on the slash-removal treatment is not sustainable, and these sites will experience considerable mortality as they age. Whether the difference in density was great enough to impact sucker vigor—as suggested in Palik et al. (2003) was not considered here.

While there was not a statistically significant difference in aspen regeneration between the treatments as a whole, CWD density can have an effect on *Populus* stem density. The retention plots had considerable heterogeneity in CWD density as noted above, and aspen regeneration in slash-retention plots was negatively correlated with CWD density. However, even plots with 500-1,000 cm^2/m^2 of CWD managed ~2 aspen stems/m^2, a density that translates to 8,000 stems/acre, well above the 4-5,000 stems/acre deemed adequate for regeneration in northern Minnesota (Bates et al. 2002; Perala 1977). It seems clear that aspen sucker regeneration can be negatively affected by CWD density—as predicted—but that final tree density at maturity is unlikely to be affected by all but the densest CWD piles. Further, there was no difference between the plots in the density of the tallest suckers, each averaged ~1 stem/m^2. These taller stems are presumably the most likely to survive the intra-specific competition and a lack of difference in this size class is likely more meaningful than the density of stems under 0.5 meters.

As with aspen, there was little difference between the treatments in shrub or tree response for most other woody species, with the exception of beaked hazel (*C. cornuta*) which increased by a statistically significant amount in the slash-retention treatments plots, but not in the slash-removal or control plots. This difference was the only statistically significant difference among shrub and tree responses between the two treatments post-harvest on the CFC1 site, and was not different between the three treatments on the Carlton site. This last fact may be a function of the relative lower initial density of hazel on Carlton plots (0.6 stems/m^2) versus those at CFC1 (1 m^2/plot). Hazel response to harvest type matched the predicted hypothesis, nearly all hazel over 0.5 meters in height present prior to harvest was removed on the CFC1 removal site via the whole tree skidding and the corresponding slash-retention treatment retained more hazel of this class.

It is worth noting that although the hypothesis that whole-tree skidding would inhibit hazel regeneration was supported, there is no support for the prediction that reduced hazel density would enable greater aspen regeneration. Although hazel is considered an undesirable competitor to aspen regeneration by loggers and forest managers (Mundell, et al. 2007; Ron Severs, personal communication), it is important food for wildlife. Aspen and balsam poplar are rapidly growing trees that regenerate from root suckering, and they can quickly outpace hazel and other potential competitors that could shade them out (Bates et al. 2002; Perala 1977). A high enough density of hazel taller than the regenerating aspen could impede the aspen suckers, and there was some evidence here. Only 5 plots had a density of hazel >0.5 meters above $3/m^2$, and those plots had a density of taller class aspen well below $1/m^2$, but for lower densities of tall hazel there was no statistically significant effect of the hazel. Given the importance of this native shrub species for wildlife, the limited impact hazel has on aspen regeneration, and the severity of the whole-tree skidding method to inhibit it, it seems unwise to pursue this practice.

There was also no statistically significant relationship between *Rubus* density and aspen regeneration, indicating that aspen clones likely respond quickly enough to grow past a height where *Rubus* stems could shade them.

Harvest of woody biomass neither promoted nor inhibited aspen regeneration to an extent that exceeded the natural range of sucker density in the region. Allowing for natural thinning of suckers through stand development, I find it unlikely that any difference will be evident at stand maturation compared to non-CWD harvested areas. Although locally dense CWD piles can inhibit aspen suckers, density across most of a harvested stand would not rise to an inhibitory level even if all logging slash was left on site as was done here. The extra scarification of whole-tree skidding and the increased equipment pressure to harvest the woody biomass did partially suppress possible aspen sucker competitors such as hazel, but did not improve aspen regeneration itself.

I found non-statistically significant differences between the two harvest treatments for the genus *Acer*, which on these sites included two tree species (*A. rubra* and *A. saccharum*) and a shrub species (*A. spicatum*). The maple species are important winter browse for both ungulates (Dodds, 1960; Moose Advisory Committee et al. 2009) and snowshoe hare (Dodds 1960). Moose maple *(A. spicatum*) may be especially important for wildlife due to its shorter growth form and its tendency to grow in thick, cover providing clumps. While I did not find a statistically significant difference in post-harvest stem density, the degree of loss was greater for slash-removal plots. Maples are an important enough group to warrant further investigation.

The impact on *Vaccinium* species also merits more study. In addition to providing food for wildlife such as bears, blueberries are important culturally for the Native American community of the area and culturally and economically for the broader community through harvest by locals and tourists. Both harvest treatments saw a decrease in stem density, with an apparent steeper decline in the slash-removal plots. The small number of plots on which blueberries was found and the low initial densities kept the difference between the treatments from being statistically significant, but the almost complete loss on the removal treatments is noteworthy.

Lastly, the sites I used had no non-native shrubs present. As non-native species such as common buckthorn (*Rhamnus cathartica*) and introduced honeysuckles (*Lonicera* spp.) expand their presence in the region, differences in how these shrubs respond to biomass harvest should be investigated. In the following chapter on herbaceous vegetation I note that non-native forb species increased more in slash-removal plots than slash-retention ones as a result of greater forest floor scarification. The increased light and bare soil associated with harvest—especially biomass harvest—may open these forests to increased invasion when these species are present in the area.

Chapter 4

Impact of Woody Biomass Harvest on Forest Floor Vegetation

Introduction:

Coarse woody debris can be important in systems as a source of water retention (Maser et al.1979; Maser and Trappe 1984; Stewart et al. 2010), a rooting site for vascular plants including trees (Maser et al.1979; Maser and Trappe 1984), and a growth substrate for lichens, fungi, and non-vascular plants (Astrom et al. 2005; Siitonen 2001; Stewart et al. 2010). Removing existing CWD and generated slash depletes this resource. As a result, biomass harvest may lead to changes in the plant community as taxa that are adapted to more light and drier conditions may outperform more shade tolerant plants.

I hypothesized that biomass harvest would lead to a decline in native forest forb cover, and an increase in those plants that are less shade tolerant relative to both pre-harvest levels and paired control plots. I also expected an increase in disturbance associated taxa, including graminoids, annual and seed dispersed forbs, and members of the Rubiaceae family. I further expected an increase in non-native, or non-native species. This last category could include both species that are non-native but not necessarily invasive and some species that could have an invasive potential (as defined by Beck et al. 2006).

Given the greater severity of the harvest, I hypothesized that the slash-removal treatments would experience greater changes in these categories than the associated slash-retention treatments, where the logging slash may provide shade and act as a moisture reservoir. While I predicted that overall cover of native forbs would decline in harvested stands, I did not expect the local richness of these species to be affected in only two years. Indeed, I predicted overall species richness would likely increase as non-native species and shade intolerant species become established. As a result, overall diversity of the plots was predicted to increase, but the share of this diversity that was native forbs was expected to decline, and to decline more steeply in the slash-removal plots.

Whole-tree skidding and slash-removal was initially prescribed on the two Cloquet Forestry Center Sites to scarify the soil, damage or remove existing hazel (*Corylus cornuta*) and prevent the regeneration of hazel (p 31, this thesis). As such I hypothesized that this harvest method would result in more soil disturbance, leading to increased area of bare soil. Bare soil may allow for increased colonization of the site by annual plants and non-native species. Establishment of non-natives is expected to be lower on the slash-retention sites, and lower still on paired control plots where no harvest occurs.

Methods

Sites: I selected two sites for full analysis of shrubs and tree regeneration: CFC1 (NPC[6] FDn43) and Carlton, NPC MHn35). The second site on the Cloquet Forestry Station (CFC3) was not chosen for this part of the study due to low plant alpha diversity (an understory dominated by only two herbaceous species (*Eurybia macrophylla* and

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[6] The Minnesota Department of Natural Resources' Native Plant Communities Classification System: http://www.dnr.state.mn.us/npc/classification.html

Clintonia borealis), one shrub (*Diervillia lonicera*), and a pre-harvest canopy of only aspen (*Populus* spp).

Vegetation sampling.—I used the same sample plots I set up for the shrub and CWD censuses. Plots were sited to avoid impacting the plots during the normal course of trapping. Plots were only visited when sampled, and all other foot traffic was routed away from the plots. From the central point of each plot I used the cable from the shrub survey to place 4, 1m^2 sub-plots in the cardinal directions (NESW) centered 2.82 m. away. I swept circular one m^2 plots with a pin/cable setup with a 56.4 cm radius. I chose circular plots over Daubenmir frames (Daubenmir 1959) as they are easier to establish and transport, and are less sensitive to bias from placement. Herbaceous plots were not permanently marked, but given the standardized location protocol, the plots likely had considerable overlap from year to year.

I collected samples once prior to harvest, once the summer immediately following harvest and once two summers following harvest.

I identified herbaceous plants down to the most specific taxonomic level to which I could be confident—species in most cases, though some sister species were identified only to genus. Grasses, sedges and rushes were identified only to family, and analysis was done on the summed families within the Order Poales. Bryophyte mosses were also enumerated as a single taxa, as were the club mosses (Lycopodiopsida), which included three species of *Lycopodium*.

For current taxonomy I followed the State of Minnesota Vascular Plant Checklist (MNTaxa, MN DNR 2013).

Within each plot taxa present were noted and each was assigned a cover percentage of the plot. Looking straight down at the plot, I determined what percentage of the plot was covered by that taxa or physical characteristic. Percentages were classified as deciles (1-10% through 91-100%). Taxa could also be assigned a value of 1% cover when it was present, but covered less than 2% of the plot, and 100% cover when it filled the entire plot. Because of layering cover totals could exceed 100%. Overall cover was estimated similarly to individual taxa, and thus could be less than the sum of all taxa, again due to layering. Data are treated this way rather than as percentages of total cover to preserve true abundance and reflect the open nature of the forest ground floor; fully 30% of each plot on average was not covered by any vegetation prior to harvest.

Litter cover and bare ground were also categorized in the same way for each plot. Bare ground as used here refers to exposed mineral soil; ground which was not covered by a vascular plant but which was covered by litter, duff, or moss would not be considered “bare.” Litter depth was measured in representative points (typically plot center and any points which appeared to differ from the center) in each plot and averaged. Beginning in 2007 I also began categorizing the percentage of each 1m^2 plot that was covered by either coarse or fine woody debris (FWD). This measurement is lacking for CFC1 prior to harvest.

Sampling was done between July (when vegetation was considered to be fully grown) and September (before senescence became problematic) of each year, with the bulk of sampling done in July and early August. Care was taken to balance the sampling effort across a site between treatments to avoid observation date from affecting one treatment differently than another. Thus, while it may have taken some time for an entire site to be surveyed each year the three treatments within each site did not differ in the timing of the survey effort. The four sub-plots were lumped for each plot, and the summed data was compared over years to minimize the chances of differences in sub-plot placement between years causing Type I errors.

As many taxa were rare or present on only a few plots or one site and not the others, plants were grouped into higher-order categories for analysis. It is not unusual for even the most ubiquitous of forest herbs to be only minor components of the community, as Reich et al. (2012) found for the herb *Maianthemum canadense*, which was found on more plots than any other species in their study of forests in northern Minnesota, but ranked only 73rd in percentage cover in plots on which it was found. These higher order categories have either a taxonomic rationale—such as the Order Liliales—or an ecological one such as native forbs, non-native forbs, or graminoids.

Analysis.**—**The four vegetation sub-plots were averaged into a single measurement for each plant variable at each plot. Percent cover of physical characteristics, individual plant taxa and combinations of taxa—such as order or ecological status—was compared in paired T-tests or Wilcoxon signed rank tests before harvest and one and two years post-harvest. Although care was taken to situate the grids in similar conditions between the treatments, and as noted above each treatment was one portion of a single forest stand, there were some variables in which site heterogeneity led to statistically significant differences in variables—such as percent fern cover—prior to harvest. In these situations, the data was transformed with natural logarithms to preserve relative changes. These transformations are noted when they were required. Differences in treatments were analyzed with T-tests.

I used R (R 2.13.1 for Mac OS X GUI 1.40) for all statistical analysis.

RESULTS

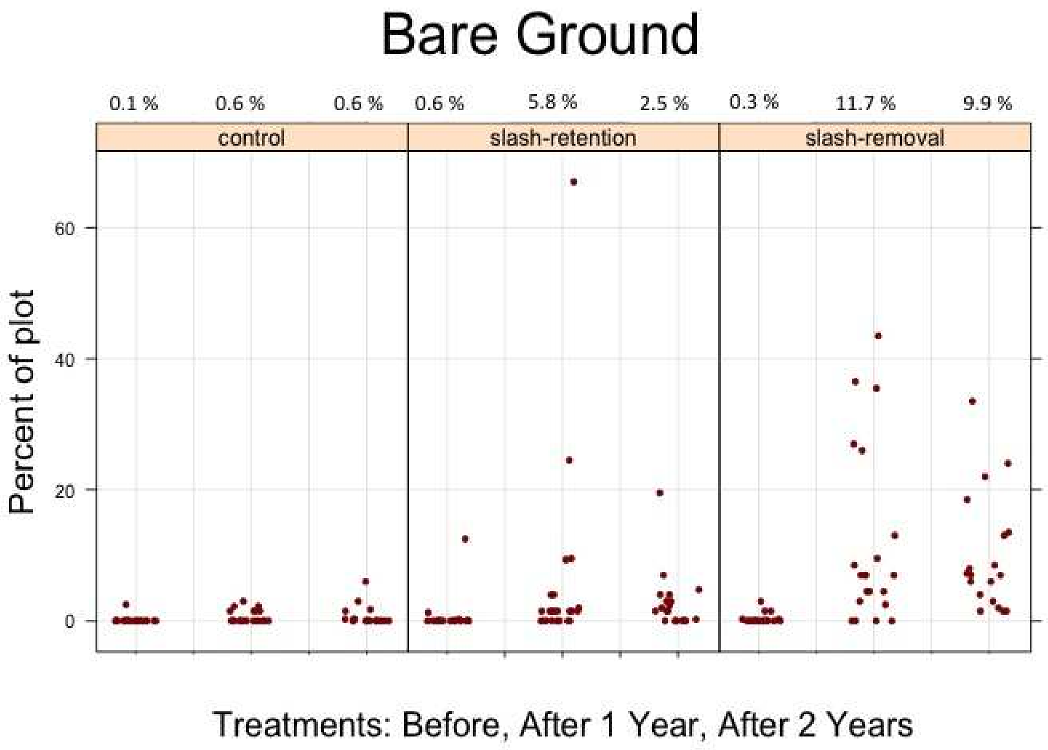
There were few differences between the treatments prior to harvest for either plant taxa coverage or physical characteristics. Table 4.1 summarizes the main pre-treatment conditions, including differences between treatments where they existed. There was a noted difference in total vegetation coverage (both overall cover and summed cover) and native species coverage between the treatments, with the slash-retention plots lagging in each. Although there was heterogeneity across the sites, especially in the presence or absence for less common shrubs and forbs, these differences were generally not statistically significant, especially after plants were lumped into higher order taxa. Other pre-harvest differences between treatments not included in Table 4.1 are noted in the text.

Species richness.—Overall forb species richness (native and non-native species combined) increased on all three treatments. Prior to harvest 25 native and 2 non-native species were known to be present in at least one plot for one treatment. By two years after harvest these numbers had increased to 28 and 15 respectively. Two native species found on only one or two plots prior to harvest were not located again following harvest (*Linnaea borealis* and *Coptis trifolia*) while six native species not encountered prior to harvest were now present in at least one plot (*Polygala paucifolia, Viola spp., Iris versicolor, Geranium bicknelli, and Impatiens capensis*). Both slash-retention and slash-removal treatments lost *Thalictrum diocium* (present on only one and two plots respectively) and *Pyrola* spp. (three and one plots respectively). Slash-retention plots lost two further species which were still present on slash-removal plots (*Sanicula* spp. and *Asarum canadense*) while the only species whose loss was unique to the slash-removal plots was *Coptis trifolia*.

Bare ground.—Bare ground was rare before harvesting, comprising on average less than 1% of all plots, but increased in both cut treatments after harvest (Figure 4.1). The increase in bare ground was not statistically significant between the treatments after one year, but by year two the slash-removal treatment showed a larger increase in bare ground relative to pre-existing levels than slash-retention plots.

Overall cover Prior to harvest, overall vegetation cover inside the sample plots averaged 57.1% (s.e. 3.27) in the slash-retention treatment plots, 74.4% (s.e. 2.71) in the slash-

Figure 4.1 Plot of bare ground in treatment plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean percentages for that sampling period.



removal treatment plots and 68% (s.e. 1.96) in the control plots. Overall cover prior to harvest was significantly lower in slash-retention plots than in either the removal treatment (t = -4.09 df = 41.19, $p < 0.001$) or the control plots (t = -3.09, df = 35.83, p= 0.004), but cover did not differ between the removal and control plots (t = 1.66, df = 36.87, $p = 0.10$). Summed coverage of all plants (which took into account layering) followed a similar trend.

Overall and summed cover both decreased on the harvested treatments in the year immediately following harvest, only to increase to levels higher than pre-harvest after two years (Figure 4.2). Summed coverage was over 130% of pre-harvest levels on average in both slash-retention and slash-removal plots.

Non-native species.—Non-native species increased in all harvested treatments over pre-logging levels (Figure 4.3). The increase in non-native species cover was greater for the slash-removal than for the slash-retention treatment after one year (t=3.19, df=24.57, p=0.004) and was still almost statistically significant after two years (t=1.77, df=29.82, p=0.088). Non-native species were a greater issue on the CFC1 than the Carlton location, and the difference between the two treatments held after two years, with these species covering 7% of slash-retention plots and 14% of slash-removal plots (t=2.66, df=10.49, p=0.023).

Non-native species richness was also statistically significantly greater for both cuts after treatment compared to before counts. Slash-removal treatments showed a greater increase in non-native species richness relative to slash-retention treatments after

Figure 4.2 Plot of overall cover in treatment plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean percentages for that sampling period.

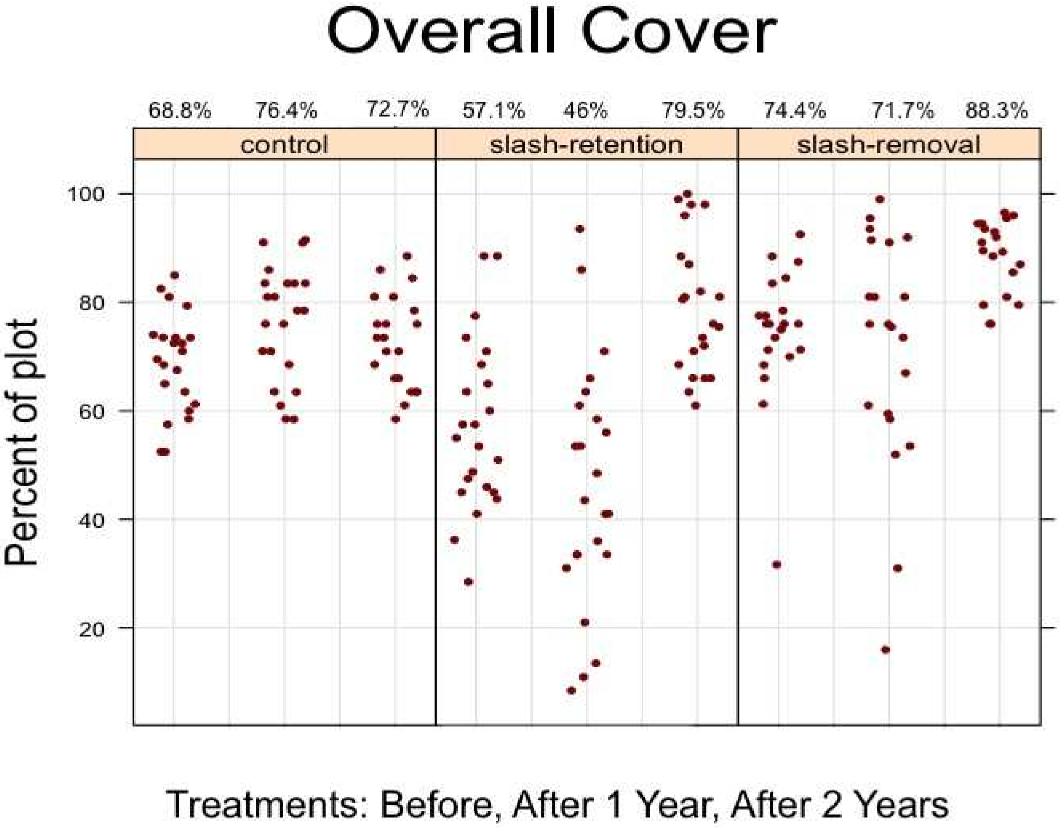
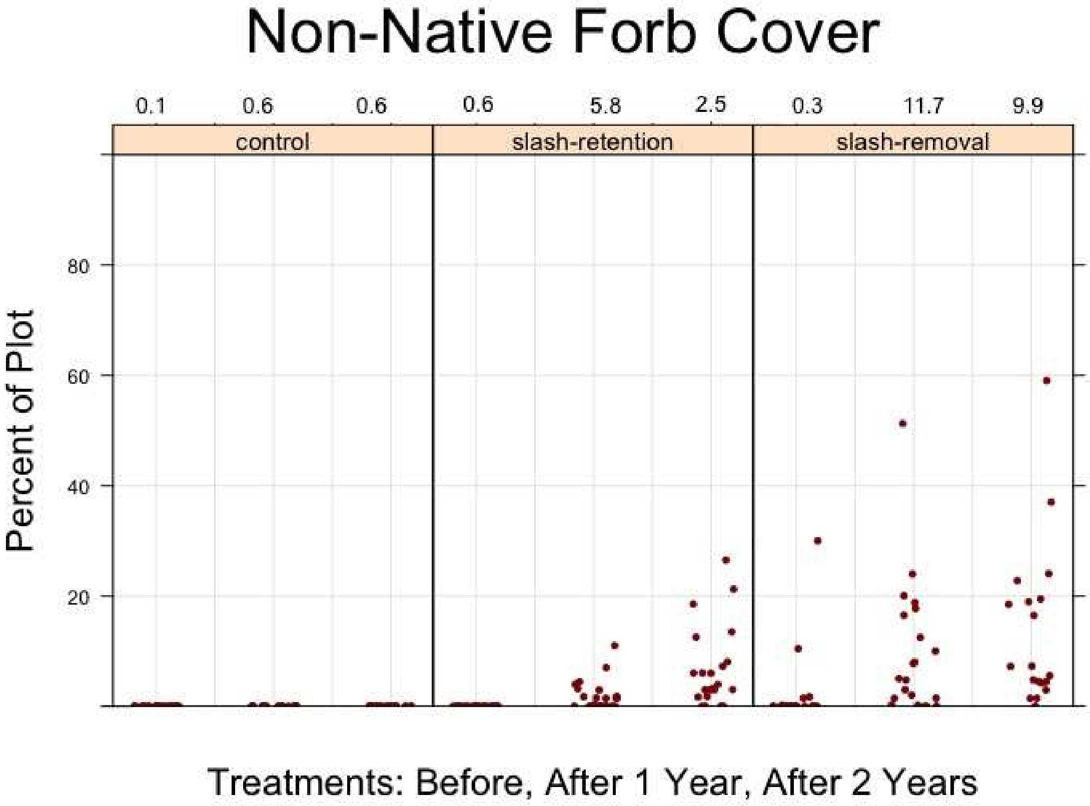


Figure 4.3 Plot of non-native forb cover in treatment plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean percentages for that sampling period.



one year, but by the second year the difference was no longer statistically significant as non-native plants colonized the retention treatment plots.

Non-native species were absent in the slash-retention and control plots prior to logging, and represented only about 3% of overall cover in the slash-removal plots prior to harvest, a number largely driven by one plot on CFC1, which had considerable cover of two non-native species—hemp nettle (*Galeopsis tetrahit*; 22% of plot) and bindweed (*Fallopia cilinode*; 8% of plot). These two species were the only non-native forbs found in any of the sample plots prior to harvest. In all, only 5 of 22 slash-removal plots had any invasive species prior to harvest, a number that increased to 18 of 22 plots by one year after harvest. By the second year only one slash-removal plot was free of invasive species.

The original two species of non-native plant increased to 10 species by one year after harvest, though many of the species were limited to only a handful of plots. After two years 19 of 22 slash-retention plots had at least one non-native species (mean 2.3 species/plot), and 18 of 19 slash-removal plots had at least one non-native plant species (mean 2.7 species/plot). The total species richness of non-native plants had increased to at least 15 species. These numbers represent a minimum presence, as some species may have been found outside of plots, and others presented here as a single taxa—such as non-native thistles—likely contained more than one species. Further, I did not identify grasses, reeds, or sedges to species, and any non-native graminoids would have escaped notice. This is especially noteworthy as several non-native grasses are known to invade recently logged boreal forests, including smooth brome grass (*Bromus inermis*) which is noted as being of particular threat (Sumners, 2005).

As explained in the methods above, each vegetation plot is an average of the values of four, 1 m^2 sub-plots spaced 2.82 m. from a central point in the cardinal directions. Although looking at the plots does not reveal a difference in spread of non-native species, a difference is apparent in considering these sub-plots, with a greater proportion of slash-removal sub-plots containing non-native species than slash-retention treatments (Figure 4.4).

Non-native species also increased their proportion of overall vegetation. By one year after harvesting non-native species represented 2.4% of summed cover in the slash-retention plots and 8.6% in the slash-removal plots. Both increases were statistically significant (t = -3.44, df = 22, p = 0.002 and t = -2.75, df = 27.69, p = 0.01 respectively) though the increase was significantly greater for the slash-removal plots (t = 2.5, df = 23.63, *p*= 0.02). After two years non-native species represented 3.8% of the overall vegetation on slash-retention plots, and 7.8% on slash-removal plots, while non-native species represented only 1/10 of 1% of overall cover in the control plots. The increase in proportion of overall cover that was non-native species was no longer significantly different between the harvest treatments after two years (t = 1.68, df = 24.4, p = 0.106).

Native species.—Native species richness of sample plots declined slightly, but not by a statistically significant level in the harvest treatments by two years post-harvest. A Shannon Index of native forb diversity showed a decline for both treatments after one

Figure 4.4 Plot of non-native species presence in vegetation sampling sub-plots 2 years post-harvest.

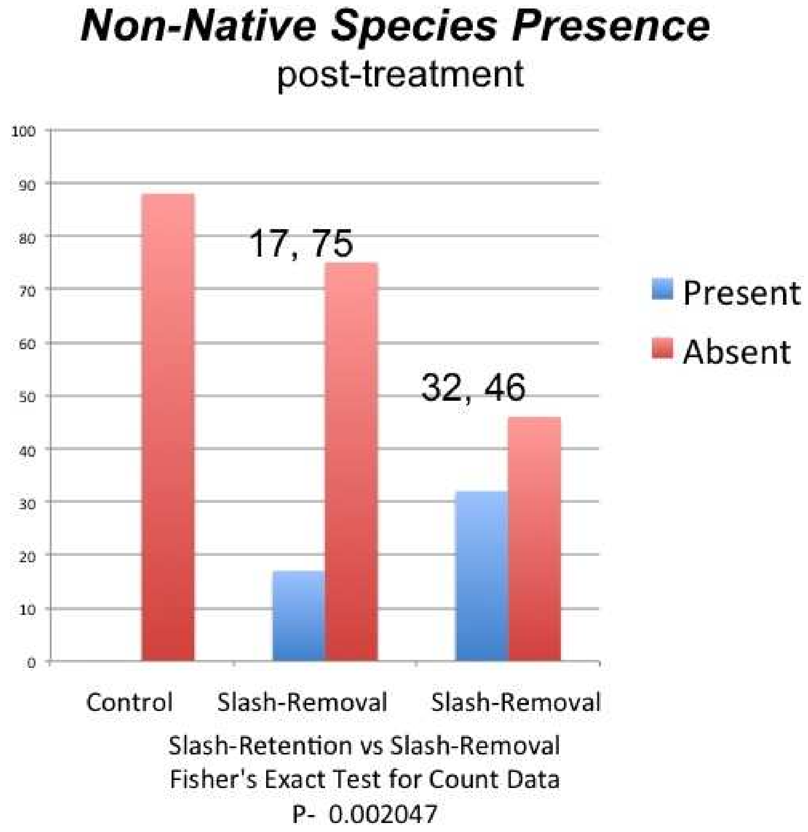


Table 4.1 Mean values for selected physical characteristics and groupings of plants: 1) before harvest; 2) 1 year after harvest; and 3) 2 years after harvest. Treatments include: Control (no harvest), Slash-retention (clearcut with slash retained on site) and Slash-removal (clearcut with whole tree skidding and/or mechanical removal of slash). The corresponding value columns are mean percent cover (with the exception of Litter Depth which is mean depth in mm). Standard errors are in parenthesis. Values between treatments across a year followed by a different letter are statistically significantly different. Change between years within a treatment (🡅/🡇, no change (nc)) are noted in each treatment's **Δ** column, and are designated as ns (not statistically significant, p > 0.1), * (p<0.1), ** (p<0.05), *** (p<0.01, **** (p<0.001). Differences between treatments in magnitude of change are noted where one or more treatments has a statistically significantly stronger response (p <0.05) than the others. Two treatments may have similar values within a year, but may still show statistically significant differences in how they changed.

Condition	**Status**	**Control**		**Retention**		**Removal**	
		Value	**Δ**	**Value**	**Δ**	**Value**	**Δ**
Overall cover*	Before	68.8% (1.96) *a*		57.1% (3.27)*b*		74.4% (2.71)*a*	
	After 1	76.4% (2.25) *a*	↑** a	46% (4.65) *b*	↓* *b*	71.7% (4.65) *a*	↓ns *b*
	After 2	72.7% (1.86) *a*	↑* *a*	79.5% (2.7) *b*	↑**** *b*	88.3% (1.57) *c*	↑**** *b*
Summed Cover**	Before	100.6% (4.75) *a*		80.9% (4.72) *b*		109.5% (5.46) *a*	
	After 1	86.5 (6.51) *a*	↓** *a*	59.0% (5.64) *b*	↓*** *a*	91.8 (6.98) *a*	↓* *a*
	After 2	93.6 (5.28) *a*	↓ns *a*	129.2(6.51) *b*	↑**** *b*	145.8 (5.50) *b*	↑**** *b*
Native forbs (All species) [1]	Before	44.3% (3.45) *a*		26% (2.33) *b*		40.5% (4.45) *a*	
	After 1	37.3% (3.14) *a*	↓ns *a*	11.4% (1.43) *b*	↓**** *b*	14.1% (1.42) *b*	↓**** *c*
	After 2	44.4% (3.16) a	nc *a*	25.4% (2.43) *b*	↓ns *a*	29.8% (3.37) *b*	↓* *b*
Native forbs (NBLA) [2]	Before	24.8% (2.18) *a*		21.3% (2.45) *a*		34.4% (3.75) *b*	
	After 1	22.7% (1.82) *a*	↓ns *a*	7.5% (1) *b*	↓**** *b*	10.1 (1.24) *b*	↓**** *c*
	After 2	31.8% (3.2) *a*	↑* *a*	17.4 (2.06) *b*	↓ns *b*	20.8 (2.11) *b*	↓**** *c*

	Before	0.0% (0.0) *a*		0.0% (0) *a*		2.46% (1.48) *a*	
Invasive Spp.	After 1	0.0% (0.0) *a*	nc *a*	1.8% (0.57) *b*	↑*** *b*	9.75 (2.67) *c*	↑** *c*
	After 2	0.0% (0.0) *a*	nc *a*	6.74% () *b*	↑**** *b*	13.7% () *b*	↑*** *b*
	Before	0.1% (0.11) *a*		0.6% (0.54) *a*		0.3% (0.17) *a*	
Bare ground	After 1	0.6% (0.21) *a*	↑** *a*	5.8% (3) a*b*	↑ns, *ab*	11.7% (2.9) *b*	↑**** *b*
	After 2	0.6% (0.32) *a*	↑ns *a*	2.5% (0.91) *a*	↑* *a*	9.9% (2) *b*	↑**** *b*
	Before	41mm (1.95) *a*		37mm (2.54) *ab*		33mm (2.59) *b*	
Litter Depth	After 1	43mm (3.72) *a*	↑ns *a*	25mm (3.85) *b*	↓** *b*	19 (3.31) *b*	↓*** *b*
	After 2	48mm (3.06) *a*	↑* *a*	20 mm (3.77) *b*	↓**** *b*	19mm (3.3) *b*	↓*** *b*

[1] Native forbs only, not inclusive of graminoids or ferns. [2] NBLA: No bigleaf Aster (*E. macrophylla*)
* Overall cover: Fraction of plot covered by any vascular vegetation.
** Summed Cover: The sum of all taxa coverage. May exceed overall cover and 100% due to layering.

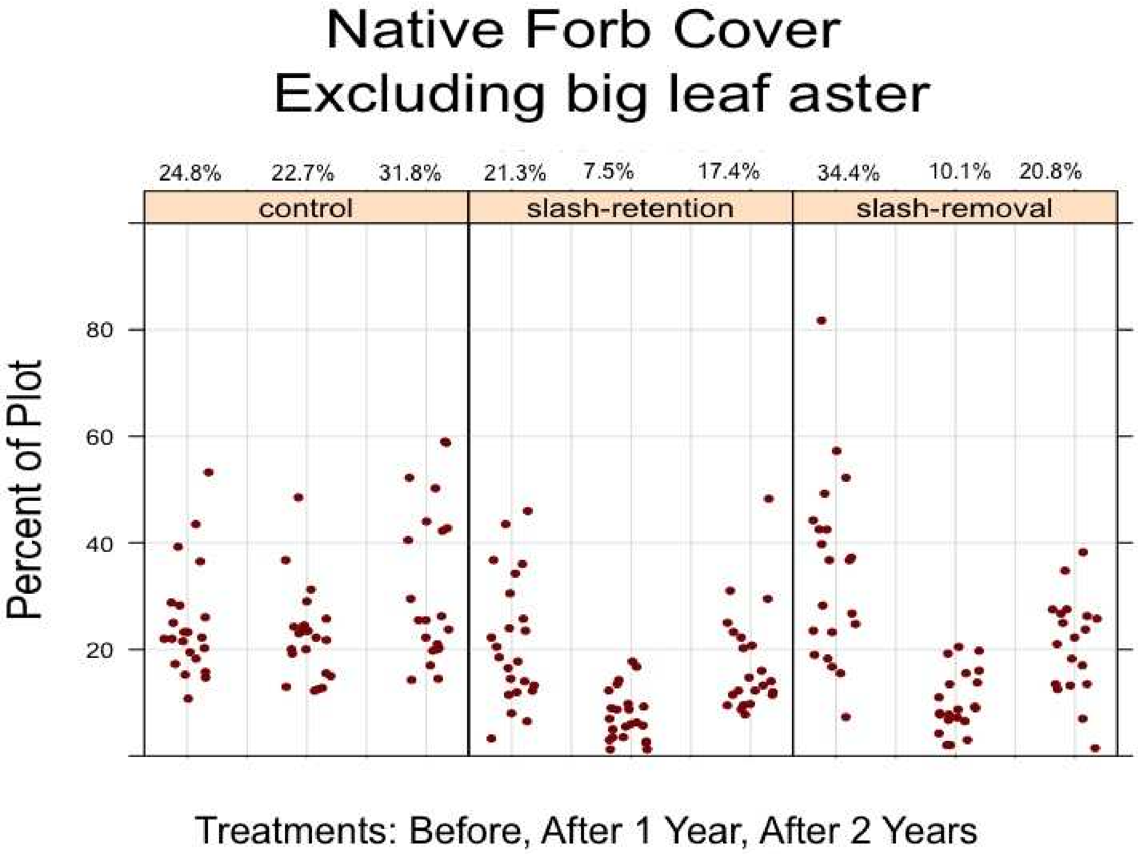
year (retention: H’ 1.213 -> 0.691, t=5.093, df = 42.85, $p < 0.001$; removal: H’ 1.229 -> 0.819, t=3.563, df= 38.996, $p < 0.001$) but native forb diversity had largely recovered in those treatments by the second year (retention: H’ 1.202; removal: H’ 1.219).

While species presence was not immediately impacted, changes in the prevalence of some species and some groupings of species were observed. Both treatments showed a statistically significant decrease in the sum of total cover represented by native species after treatment, and slash-removal plots showed a greater decrease in the amount of native species cover in both years relative to control and slash-retention plots. Slash-retention plots showed a statistically significant decrease in native species cover relative to control plots after one year, but by the second year after harvest the plots had improved enough to no longer be statistically significantly different from control plots.

Native species cover was dominated by a few common species, most notably bigleaf aster (*Eurybia macrophylla*) which is among the most frequently occurring herbaceous species in this region (Reich et al. 2012). Bigleaf aster was weakly but positively affected by both slash treatments, increasing by non-statistically significant amounts on the harvested plots, and decreasing on the paired controls. When I dropped *E. macrophylla* from consideration native cover was lower for both treatments relative to control in both years, and slash-removal showed a more statistically significant drop than slash-retention for both years (year 1: t=-2.719, df=31.73, $p = 0.01$; year 2: t=-2.682, df= 28.79, $p = 0.01$).

Additionally, native species cover loss was greater by a statistically significant margin in the slash-removal than slash-retention treatments and was slower to recover as

Figure 4.5 Plot of native forb cover in treatment plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean percentages for that sampling period.



well (p=0.01 both years) at least in the immediate term. By two years after logging native forb species cover excluding bigleaf aster in the slash-retention treatments had improved enough (only 3.5% below pre-treatment levels) to no longer be statistically significant, while the slash-removal treatment was still down 15% from pre-treatment levels (p=0.004 compared to pre-existing cover). Non-native species filled the void left by native species cover, and in fact overall vegetation cover was greater in the treatment plots two years after harvest than it was pre-harvest. The fraction of this cover that was native forbs had fallen however.

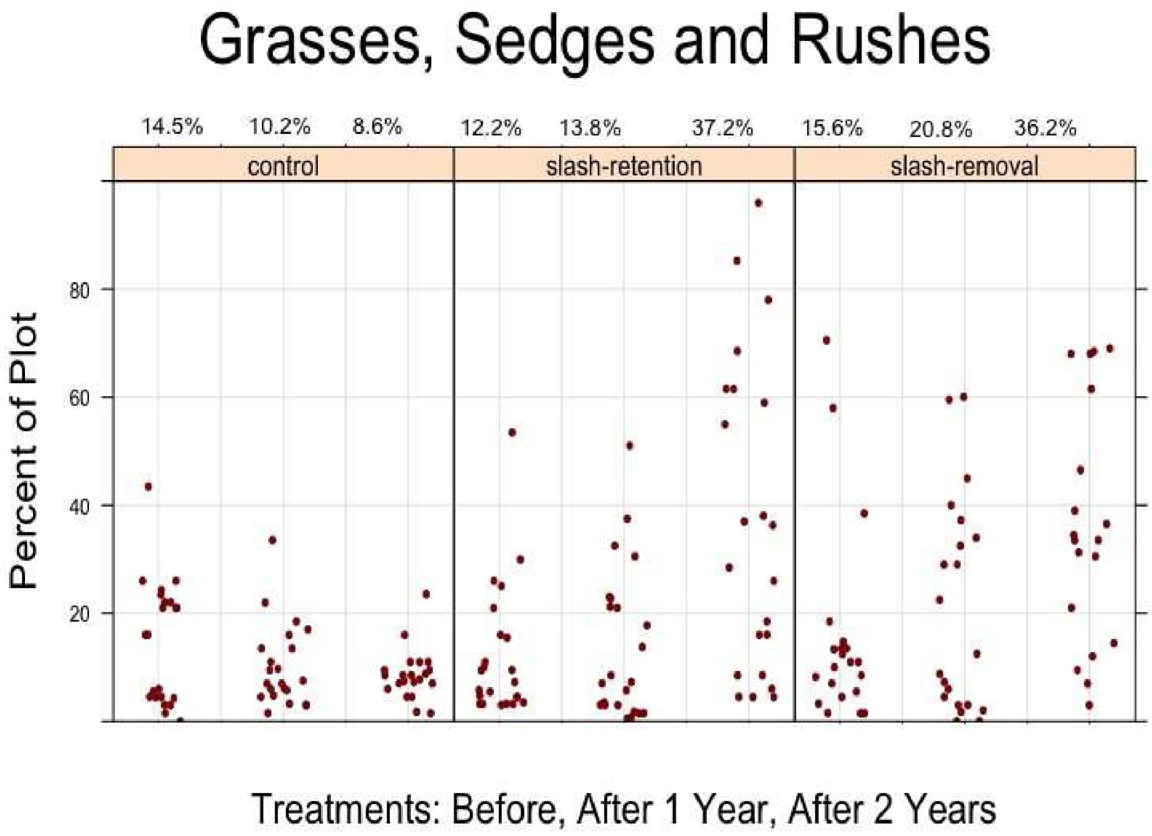
Prior to harvest native forbs represented 26% of the summed vegetative cover on the control plots, 28% in the slash-retention plots, 31% of the slash-removal plots, with the remaining cover made up of grasses, shrubs, and ferns. By two years after harvest these ratios were 35%, 13% and 14.5% respectively. The decreases for the harvested plots were significant (slash-retention: t = 4.15, df = 26.71, $p < 0.001$; slash-removal: t = 5.34, df = 30.13, $p < 0.001$). Changes in the fraction of summed cover that was made up of native forbs were also significantly different from control for both slash treatments (slash-retention: t = 5.68, df = 40.94, $p = 0.001$; slash-removal: t = 6.68, df = 37.43, $p < 0.001$) but not from each other.

Liliales.—Though each individual species was a small enough component of the community to make year-year comparisons difficult, in aggregate the Liliales showed statistically significant declines in cover after treatment relative to prior levels and a statistically significant reduction in cover relative to control. The reduction was greater the first year following harvest, and had improved by the second year, but coverage was still lower than pre-harvest levels for both treatments. Differences between the harvest treatments were not statistically significant for either year.

Graminoids.— Graminoids showed a positive, but delayed response to harvest (Figure 4.6). After a single year post-harvest there was not a statistically significant change in graminoid cover, but after two years grasses covered a much larger percentage of the plots for the harvest treatments than they had prior to harvest (retention: 12.5% ->37.2%, t=-3.67, df=28.33, p = 0.001; removal: 13.6% -> 36.2%, t=-3.58, df=32.82, p = 0.001). Control plots lost graminoid cover over those two years (14.2% -> 8.6%, t=2.05, df=26.42, p = 0.05). There was not a difference in graminoid cover between the harvested plots (t = 0.12, df = 38.52, p = 0.9), though both the retention and removal treatments had more grass than the control plots (t = -4.56, df = 22.2, p < = 0.001; t = -5.29, df = 19.45, $p < 0.001$ respectively).

Grasses and sedges also became a more dominant component of the plant community in both treatments after harvest compared to pre-existing conditions and to controls. As noted above, some of these plants may be non-native or even invasive, but I lacked the skill to separate non-flowering grasses in the field. Prior to harvest grasses and sedges combined represented 14% of the total cover (all taxa summed) in both slash treatments. By two years after harvest these ratios had increased to 28% (t = -2.81, df = 32.30, p < 0.001) and 41% (t = -2.65, df = 35.99, p = 0.001) while grasses actually

Figure 4.6 Plot of graminoid cover in treatment plots before harvest, 1 year after harvest, and 2 years after harvest. Numbers above the column heading indicate mean percentages for that sampling period.



declined as a percentage of cover in the control plots (13%-9%; $t = 1.89$, $df = 27.82$, $p = 0.07$).

Ferns.—Fern cover decreased in all treatments—including control plots—however plots in which the slash was removed showed a steeper decline than corresponding plots with slash-retention ($t = -2.80$, $df = 20.54$, p-value = 0.01). Ferns were not equally distributed across treatments, but covered a much greater proportion of slash-removal plots prior to harvest than slash-retention plots (Retention: ~4.7% coverage of plots; Removal: ~19% cover of plots; $t = -3.73$, $df = 27.93$, p-value < 0.001. Prior to harvest ferns were absent in 12 of 23 slash-retention plots, compared to only 2 of 21 slash-removal plots with no ferns.

Bryophyte moss.—Mosses declined statistically significantly in harvest plots from roughly 5% of plots to less than 1% one year after harvest. Control plots also lost a non-statistically significant amount or moss. There was not a statistically significant difference in loss of moss cover between the harvest treatments, nor was the loss of moss cover in either harvest treatment significantly greater than that in the control plots. By the second year after harvest moss was beginning to recover, with small but statistically significant increases in cover. Moss was still a smaller component of the plots in harvest treatments than it was prior to logging, but the difference continued to be non-statistically significant with respect to control plots.

Club moss.—Lycopodium spp. showed the inverse to the relationship that ferns demonstrated: they were much more frequent components of the slash-retention treatments than the slash-removal treatment plots prior to harvest, covering an average of 14% of slash-retention plots, versus only 1.7% of slash-removal plots (t = 3.16, df = 24.43, p-value = 0.004) again making meaningful comparisons between treatments problematic. Lycopodium cover did not show a significant decrease in either the slash-removal (1.73% -> 0.7%, t = 1.088, df = 24.89, p = 0.287) or control plots (3.91% -> 2%, t = 1.01, df = 31.24, p = 0.32). The club mosses did decline in coverage after harvest in the slash-retention treatment plots (year 1: 13.96% -> 1.22 %, t = 3.37, df = 22.45, p= 0.003; year 2: 1.81%, t = 3.19, df = 21.64, p = 0.004). A log transformation of the data did not change any of these relationships.

DISCUSSION

Slash-removal following clearcut logging resulted in a greater loss of native species cover, and a greater increase in both bare ground and cover of non-native species relative to clearcut logging with slash-retention. There were not statistically significant differences between the harvest treatments for either species richness or diversity, cover of grasses, or total vegetation cover (sum of all taxa).

Previous researchers have compared the changes in plant community after disturbances from fire, insect outbreaks, or logging in boreal systems and have found the community to be resilient to a single stressor—either fire, or insects, or, logging—but note more changes in the community as the severity of the disturbance increases (Donato et al. 2006; Kurulok and Macdonald 2007; Pidgen and Mallik 2013). A similar phenomena appears to be at work here: harvesting led to changes versus no harvesting but slash-removal led to larger changes that were more persistent for some metrics. Many impacts were similar between the harvests, while others—such as those noted above—were statistically significantly different. In most cases where there were differences the slash-removal treatment yielded a less favorable outcome for wildlife or natural conditions than did the slash-retention treatment.

Contrary to prediction, there was not a statistically significant difference in response of graminoids between the treatments, though both harvest treatments experienced a statistically significant increase in grass cover after harvest compared to pre-harvest levels. Other research has found an increase in graminoid cover following harvest, including Harvey and Brais (2002) who found grasses and sedges increased in cover in plots along skid trails much more than in protected harvest strips not impacted by machinery. Peltzer et al. (2000) found grass cover to increase with increasing severity of post-harvest site preparation, with the lowest values for naturally regenerating forests, and increasing as more equipment was used to disturb the soil for tree regeneration. The absence of a difference in overall graminoid cover between the two harvest treatments is worth noting. Grasses and sedges have been implicated in inhibiting tree regeneration (Frey et al. 2003, Kemball et al. 2005, Lieffers et al. 1993; Palik et al. 2003; Peltzer et al. 2000) and given the prevalence of this practice it is encouraging that the greater scarification associated with the slash-removal did not increase grass cover. There was also no relationship between grass cover and native forb cover, indicating that grasses were not negatively affecting the overall native forb community either.

As predicted, non-native species—both richness, frequency in plots, and percent cover inside plots—increased after logging in harvested sites. There was also a difference in invasion between slash-retention and slash-removal sites as hypothesized, with greater gains by non-native species in the slash-removal treatments. This is likely due to the greater scarification associated with whole tree skidding and the increased bare ground associated with the removal of CWD. This finding is consistent with other work that has found greater mechanical site manipulation during and after harvest increases non-native species colonization (Haeussler et al. 2002; Haeussler and Kabzems 2005; Pidgen and Mallik 2013; Scheller and Mladenoff 2002; Stewart et al. 2010).

It is important to note that the differences found here between the treatments are measured one and two years post-harvest. Differences could increase or decrease with time, however by two years out it appears that the slash-removal treatments may be slower to recover from the logging than the paired slash-retention plots. Harvey and Brais (2002) tracked plots for seven years after harvest. They found that differences in *Rubus* stem density between plots took three years to emerge. These same differences had begun to lessen by seven years post-harvest, but were still noticeable. In the same study grass cover was still greater in skid trails than either strip edges or strip centers after seven years. Further work with the sites in this study is advisable to determine how long of a recovery lag there is between the sites, or even if the differences persist for the length of the harvest rotation (about 40-70 years for these forests). The degree to which these differences are maintained over time between the harvest types should inform any updates to the Voluntary Site-Level Guidelines.

The native plant community of the boreal forest is adapted to disturbance, and the frequency of disturbance has selected for a plant community that can tolerate a wide range of environmental conditions, including differences in light and water levels. The forests investigated here were also second growth forests, and likely any truly shade dependent species had been removed from the community by previous harvests. While there was a drop in native species richness in plots of both harvest types in the first year post-harvest, plot average species richness had recovered back to pre-harvest levels by the second year. Competition with non-native species may pose a greater threat to individual native species tenure in the plots than the flush of light and drier conditions immediately post-harvest.

A temporary change in community composition is to be expected following disturbance, the magnitude and tenacity of the change are the metrics of concern. If specific species are locally extirpated rather than simply reduced in numbers, recovery will likely take longer. Six species that were found on fewer than six sub-plots prior to harvest were not found in any sub-plots by the second year after harvest; two in the control treatment (*Sanicula* spp. and *Linnaea* borealis) four in the slash-retention treatment (*Thalictrum* spp., *Sanicula* spp., *Asarum canadense*, *Pyrola* spp.) and three in the slash-removal harvest (*Thalictrum* spp., *Pyrola* spp., and *Coptis trifolia*). The native plant community has also evolved with a limited suite of species, relative to other forest types. The relationships between species and the natural succession after a disturbance may be upset by non-native species, and particularly invasive species. Logging that increases invasive species richness and abundance may hamper normal post-disturbance succession/recovery. In this research, the extra removal of the slash did result in greater invasion of non-native species. Haeussler et al. (2002) found a similar relationship between invasive species cover and intensity of management (increase in mechanization).

The extra impacts of the slash-removal treatment are likely due to the increase in machinery use on the site, and the dragging of whole trees across the harvested area. Purdon et al. (2004) hypothesized a similar impact of machinery on forest floor to explain differences between salvage logged and non-salvaged forests following fires. It is not possible with this design to separate out the proportion of impacts due to the harvest method from the simple absence or reduction of woody material. The Carlton County site may come close to this ideal, with the slash removed mostly by hand with chainsaw and all-terrain-vehicle. It is unlikely, however, that one rotation's worth of removed wood would have a substantial impact on the vegetation, were it not for the damage caused in removing it.

Haeussler and Kabzems (2005) compared bole-only harvest and whole tree harvest with post-harvest slash removed by machine *from outside the sample plots* and by hand inside the plots, and found no detectable differences in vascular plant community response. Alternatively, Harvey and Brais (2002) situated post-harvest plots in the center of skid trails, along the edge of protected harvest strips (which had no vehicle compaction) and in the center of the protected harvest strips. These authors found statistically significant differences in vegetation response between the plots, with grasses a much more dominant component of skid trails than the protected strips.

I intentionally made no attempt to shield vegetation plots from logging activity, and loggers were not made aware of plot location. If the results from Haeussler and Kabzems (2005) are a realistic expectation of forest plant community response to the disappearance of CWD—as opposed to the mechanical removal of CWD—that would be further evidence that it was the impact of the machinery and the removal method that impacted the community. The impact of stacking cut trees in smaller, scattered piles and bringing the de-limber to the cut trees, rather than dragging whole trees to one location for de-limbing was likely the greatest difference between the sites. Dragging of whole trees scoured the forest floor and damaged or killed understory shrubs, while the scattered delimbing of trees left behind a more homogenous scattering of limbs upon which the machines traveled with less impact on the soil.

However, the hypothesis that whole-tree dragging is to blame for the difference between the treatments finds little support in a comparison of the two sites. CFC1, with whole-tree skidding at the time of harvest, and Carlton, where I removed excess logging slash with an ATV, did not differ statistically significantly in response of native forb cover or bare ground. The slash-removal treatment on Carlton had only about half as much non-native species cover as the same plots on CFC1, but the difference was not statistically significant. Since I did not witness the harvest on this site, I cannot speak to the exact methods used, and too much time had passed for reliable recollection from those who were involved. It seems possible that whatever extra mechanization the loggers used, plus the extra mechanization of my removal, had a similar impact as the whole-tree skidding on CFC1.

I would argue it is likely that the greatest impacts come from the level of harvest traffic itself. The skidders on the slash-retention sites moved only bole wood, and stuck to defined trails, while the machines moved across the entire harvest in the slash-removal treatments. Zenner et. al (2007) attached GPS receivers to skidders on a logging site in north central Minnesota to more precisely gauge the extent and intensity of vehicle traffic in an aspen clearcut. They found nearly 1/3 of the site was affected by skidder traffic, and that over 7% of the site experienced > 100 vehicle passes, with a high of 603 (Zenner et al. 2007). Soil compaction and reductions in aspen sucker vigor were noticeable after only a single pass, and the bulk of the impact occurred in the first ten passes (Zenner 2007). Zenner et al. (2007) pre-determined skid trails and thus could place sampling effort accordingly. I did not do that here, and cannot quantitatively separate plots on or adjacent to skid trails from those that were farther away. Qualitatively, it was quite obvious where vehicle traffic was most intense, and the plots nearest well-used trails on the slash-retention treatments had higher rates of non-native species and bare ground cover. Skid trails were still visible from satellite images five years after harvest.

Biomass harvest is unlikely to directly affect local abundance of most native forbs, but has the potential to open the sites to more non-native and potentially invasive species, and possibly impact native species through this competition as more time passes. From this work it seems that the extra mechanization required to harvest the slash has more impact on the forb community than the fraction of the slash that is removed.

Harvest should proceed with caution until the long-term impacts on native and non-native species are known.

Chapter 5

Impact of Woody Biomass Harvest on Small Mammals

Introduction

Small mammals play many roles in forest ecosystems, including serving as seed dispersers, insect predators, primary consumers, and as a prey source for mammalian, avian, and reptilian predators. Coarse woody debris is important for many small mammals (Bowman et al. 2000; Bull 2002; Butts and McComb 2000; Carey and Harrington 2001; Fauteux et al. 2012; Fisher and Wilkinson 2005; Johnston and Anthony 2008; Keinath and Hayward 2003; Loeb 1999; McCay and Komoroski 2004; Moses and Boutin 2001; Riffel et al 2011; though see Craig et al 2006), and even their predators (Andruskiw et al. 2008). Coarse woody debris provides small mammals many services, including nesting/denning substrate, cover, and source of food including fungi (Maser and Maser 1988; Rhodes 1986), and insects, as well as structure that can provide for habitat partitioning (Carey and Harrington 2001; Sharkey 2008) and movement corridors (Bowman et al 2000; Pearce and Venier 2005).

Results from work on the effects of forest harvest on small mammals are inconsistent (Kirkland 1990; Pearce and Venier 2005; Riffel et al. 2011; Thompson et al. 2003; Zwolak 2009). Impacts may vary by species, forest type, climate, and silviculture methods and goals, arguing for research matched to specific situations (Kirkland 1990; Pearce and Venier 2005; Sullivan et al 1999; Sullivan and Sullivan 2001; Sullivan et al 2001; Thompson et al. 2003).

In this chapter I evaluate differences in the response of the small mammal community across harvest treatments before and after harvest following the BACI (Before-After, Control-Impact) method. At the core of this study is the question of whether the difference in net CWD post-harvest affects 1) the size and diversity of the small mammal community, and 2) the local abundance, survival, and demographics of specific small mammal species, focusing on the most common species.

Small mammals were chosen as study organisms because of their population density, short life-spans, and small home ranges. One known problem with using small mammals such as mice and voles in this type of research is the cyclic nature of their populations, both within and across years (Etcheverry et al. 2005; Pearce and Venier 2005). Local populations of small mammals in boreal systems build through the summer and early fall, decrease in late fall and over the winter and reach a nadir in spring, before building again through the summer. As such, comparisons across months even within a year are not always informative. Populations also change over years confounding inferences of what is the treatment effect and what is background fluctuation. Pairing the treatments with immediately adjacent controls, staggering the harvest across sites by a year, and trapping across most of the snow-free year was intended to address these issues.

I hypothesized that the total small mammal abundance would increase following harvest, and that both total population size and total small mammal biomass after treatment would be smallest in the control plots and largest in the slash-retention plots.

The small mammal community in northeastern Minnesota is dominated by a few very abundant generalist species (deer mice (*Peromyscus* spp[7]), southern red-backed voles (*M*yodes *gapperi*), and northern short-tailed shrews (*Blarina brevicauda*)), and a few common sciurids (Eastern (*Tamias striatus*) and/or least (*Tamias minimis*) chipmunk, and red squirrels (*Tamiasciurus hudsonicus*)), with lower numbers of species with narrower niches (J. Erb and M. Rentz unpublished data; Timm 1975). Even these narrow-niche species are generally not old-forest or CWD specialists, but are either associated with water (*Sorex palustris, Synaptomys cooperi*) or forest openings (*Microtus pennsylvanicus*, *Poliocitellus franklinii*). As such I did not expect species richness to change over time within the plots regardless of treatment, though I did expect diversity to remain steady on control plots and decline on the harvested treatments as *Peromsycus* spp. abundance increased. I also expected the two treatments to differ in degree of change; with slash-removal treatments likely becoming more dominated by mice than the slash-retention plots and showing larger declines in measures of diversity. I anticipated that overall abundance post-harvest would be higher in the slash-retention plots compared to the slash-removal plots.

Methods

The study followed the overall project's BACI methodology, with mammal surveys both before and after intentional site treatment, and in adjacent control plots. This design was

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[7] As noted previously, two species of *Peromyscus* potentially exist in the region, *P. maniculatus*, and *P. leucopus.* Field differentiation can be difficult, and field measured morphometrics are only somewhat reliable. Analysis of morphometrics reveals that both species were present on all sites and treatments both before and after harvest.

replicated in full at each of 3 sites: CFC1, CFC3, and Carlton. I trapped small mammals for two years prior to treatment to establish a baseline and for one to two years after harvest depending upon the site.

Trapping.—For field season 2005 I used only Sherman folding and pitfall traps. Beginning in 2006 I used a combination of Sherman folding traps, Ugglan Special #3 and Fitch-style traps. Sherman traps had underperformed in the previous season, and I wanted to compare their performance with other common traps[8]. I treated all live box traps equally, with common bait, bedding, and placement. In all years I ran a series of live pitfall traps.

I ran traps for four consecutive nights in May, June, July, August, September, October, and November on the two CFC sites in 2005, and over the same period for all three sites in 2006, 2007, and 2008. Trapping was sporadic and opportunistic in July to avoid capturing animals in what is typically the hottest and most humid month. For analysis I consider each four night run as its own primary period.

Each site was divided into three separate harvest treatments. Each harvest treatment was assumed to be independent of other treatments. Treatments across a site were trapped on the same nights. On each trapping site I laid out a 4 by 5 trapping grid of 20 trapping stations with 20 meter spacing (~ 0.5 ha total enclosed by traps). At each trap station two traps were placed independently within three meters of the trap center. I intentionally maximized capture probabilities by utilizing best available microhabitat within this three meter radius. When using all three trap types I cycled them between the

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[8] See Chapter 6 of this thesis for descriptions and photographs of the traps.

stations in alternating pairs (Sherman/Ugglan; Sherman/Fitch; Fitch/Ugglan), with each type of trap missing from every third station. At the first and last trap station all three types were used. Thus, for the years 2006-2008 each treatment consisted of 42 total traps, 14 of each type.

Traps were baited with "bait balls" consisting of cooked oatmeal and a chunk of raw potato (source of water) wrapped in waxed paper. Bait balls were hung from the back door of the trap in Sherman and Ugglan traps or placed in the nest box of the Fitch trap. I replaced bait balls when they had been consumed or if they became moldy. I chose bait balls over loose bait such as oats or bird seed to prevent gumming up of trap mechanisms, to ensure more consistency in bait volume and quality between traps, and to determine if bait had been disturbed or not. I added a small volume of polyfill batting (roughly the same volume as a mature mouse) in each trap for bedding, and covered traps with cedar shakes to prevent the metal traps from overheating. Traps were checked at a minimum once daily in the morning.

Ideally, traps would have been checked twice daily or locked open during the day, however limited field personnel and the remoteness of some sites made this difficult. Traps were checked twice a day in 2005, and captures during the day were minimal (63 captures over 48 calendar nights of trapping). In light of the low rate of capture in 2005 and the greater remoteness of the sites added in 2006 (which made two trips a day prohibitive) I made the decision to drop the afternoon check and treat each of the sites alike. I minimized thermal stress on captured animals by shading traps under existing vegetation where possible and covering all traps with cedar shakes to minimize solar gain on the metal traps. Chunks of raw potato were added to the bait balls as a source of water, and these were frequently chewed on by trapped animals. Finally, trapping periods were deliberately chosen to avoid the hottest periods of the year (late June through early August), and during the trapping season I would delay opening traps for a particular run if the forecast indicated unusual heat or end a run early if heat became a problem.

Along with the live trap grid, each location also had a four trap live pitfall array, in a Y configuration with pits at the end of each arm and in the center. Arm lengths were 5-10 meters long, constructed with aluminum flashing buried at least 5 cm into the ground, and at least 20cm tall. The pitfalls were 35cm square and 33cm deep and constructed of durable plastic. Pitfalls were baited with whole oats and bee moth larvae (*Galleria mellonella*), stocked with 10-20cm of coarse wood sawdust and had an overhanging plywood cover suspended over the trap to provide shelter from sun, rain, and meso-carnivores. Each pitfall also contained a small plastic tray filled with water and a synthetic sponge. The water setup was placed with amphibians in mind, though small mammals were also observed either hiding under it or utilizing it. Pitfalls were locked closed when not in use with bolts and cotter pins.

Although trapping occurred from May-November of each year capture rates were very low in May and were excluded from the analysis.

Marking and measuring of animals.—Upon initial capture animals were identified, weighed, sexed (rodents only), and marked. I marked rodents with a single ear tag

(Monel #1005-1, National Band and Tag Company) and shrews (*Blarina brevicauda* and *Sorex* spp.*)* with a Visible Implant Elastomer (VIE) tag (Northwest Marine Technology, Shaw Island, Washington). In response to frequent losses of ear tags in *M. gapperi* I began tagging this species with both ear and VIE tags in the 3rd and 4th seasons (2007 and 2008). For all species I applied the VIE tags through injection of the elastomer in the underside of the tail with a 29 gauge insulin syringe. A small (<0.01CC) amount of the mixed elastomer was injected sub-cutaneously. On subsequent captures animals were identified to individuals and weighed.

These methods were reviewed and approved by the Institutional Animal Care and Use Committee at the University of Minnesota (Protocol Number: 0703A03590).

Analysis.—

Closed population models assume that the population in question does not change during the study: no births, deaths, immigration or emigration (Williams et al. 2002). The population size can then be estimated from the capture histories. For example, a researcher can capture and mark animals on one day and then re-sample the next day and look at the ratio of previously captured animals to estimate the population size. Open population models allow the population to change during the study; the population can experience births, deaths, immigration, and emigration. Open population models can also estimate apparent survival (1- (emigration plus mortality)) in the population by combining a count of animals known to have survived with an estimate of animals that survived in the study area, but were not re-captured.

Common to both model types are the assumptions that: 1) All animals present have the same probability of capture[9]; 2) No tags are lost or incorrectly read; 3) Sampling and release are instantaneous; 4) The capture probabilities of each animal are independent of all other animals; 5) Capture of an animal does not affect its probability of future capture. Open models also include the assumption that all animals present in time *i* have the same probability of surviving to sample time *i* + 1 (Cowen 2005; McDonald et al. 2003; Seber 1986; Williams et al. 2002).

Estimates of abundance for the two most frequently captured species—*M. gapperi* and *Peromyscus* spp.—were derived from application of Pollock's Robust Design model (Kendal 2001; Williams et al. 2002; hereafter "Robust Design"). Robust Design combines open and closed models into one model to produce discrete estimates of population size within closed periods (the individual trapping runs, four nights in this study) and estimates of survival between trapping runs when the population is considered to be open.

Survival for these species was estimated with the open population Cormack-Jolly-Seber Model (CJS; Williams et al. 2002).

As noted above, population and demographic studies utilize maximum likelihood estimation to calculate values for the parameters in question. Several computer programs have been developed to do such analyses. MARK (Cooch and White 2013; White and Burnham 1999) is one such program, and has become the industry standard. Program MARK can be used directly through its own interface, however an extension for the

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[9] While any of the assumptions can be violated, there are extensions to the basic model specifically to address non-constant capture probabilities. The models used here did not use these extensions.

open-source statistical program R has also been developed. All analysis is done inside MARK itself, but the R programming interface allows for more rapid testing of models, with more flexibility in model design than the stand-alone program MARK. In either interface, investigators select the appropriate population model *a priori* and MARK produces parameter estimates from the supplied data. In the case of the Robust Design Model used here the estimated parameters are population size, survival, immigration, emigration, and capture probability. The data input is capture histories for each individual animal. For the CJS model, the capture histories are collapsed to a single value for each trapping period (either captured or not).

The Robust Design and CJS models described above are general models, which can be further modified by adjusting parameters. Each modification is itself a model; a Robust Design model where immigration/emigration is allowed to vary between each trapping period is slightly different than one in which they are held constant, and in turn is different from one in which they are constrained to 0 (no immigration or emigration). Each sub-model will produce different values for the parameters being estimated, making model selection an important step.

Model selection was done here using Akaike's Information Criterion (AIC; Williams et al. 2002), which balances overall fit of a model to the data and the number of parameters included in the model. Models that are the most parsimonious—models that explain the data well with fewer parameters needed—produce the smallest AIC scores and are considered the most appropriate model. As an example, a Robust Design model may be more parsimonious if it sacrifices model complexity by holding survival constant across each gap between sampling, rather than allowing survival to vary. True survival is unlikely to be constant, however the model may better fit the population size data by doing so.

Robust Design models with constant survival had lower AIC scores than models with variable survival. Hence, although this model is capable of estimating apparent survival rates, survival was instead estimated independently of population size with a CJS model. Generally survival estimates from the Robust Model were similar to the CJS estimates, though only the CJS results are presented here.

I used RMark to test multiple Robust Design models for population size estimation, and chose the best-fit model via AIC. Each year-site-species combination was independently estimated, and best-fit models were allowed to vary between each combination. I used MARK for survival estimation via a CJS model on all years of each site-species combination, and report here an AICc weighted average survival from best-fit models.

Abundance of species other than *M. gapperi* and *Peromyscus* spp. is presented as Minimum Number Known Alive (MKA), a total enumeration of all captured individuals within a primary period. I was not able to estimate survival for these species due to low re-capture rates.

I calculated indices of diversity for each trapping session on each site-treatment pair using MKA for all other species. I selected Shannon's Index of Diversity to favor rare species; this measure reflects the diversity of the population by calculating uncertainty of predicting the species of an individual organism selected at random from the population. As the number of species rises the uncertainty (and hence diversity) does as well. The Index also factors in the proportion of each species to the community total; a community with four species but which is dominated by a single species is less diverse than one in which the four species occur in roughly equal numbers.

I also calculated Pielou's Evenness values for the combination of red-backed voles and mice for each session-treatment-site combination. Pielou's Evenness is a measure of how close the calculated Shannon's Diversity of a community is to the maximum possible value given the number of species. May values for survival and evenness were included in this analysis.

I compared abundance, survival and weight of animals, as well as measures of evenness and diversity within treatments before and after harvest with T-tests. I compared values between treatments within a harvest status with paired T-tests. Significance is considered to be evident when $p < 0.05$, and is considered possible when $p < 0.1$. Because of the low degrees of freedom possible with so few sites, I note when tests give significance values between 0.05 and 0.1.

Within the trapping grid I established 12 vegetation sampling plots. Each plot was equidistant from the nearest 4 trapping points, essentially a nested sub-grid inside the larger trapping grid. In this way the vegetation plots were sited to avoid impacting the plots during the normal course of trapping. Plots were only visited when sampled, and all other foot traffic was kept away from the plots. With this design it was not possible to directly characterize the vegetation and CWD of a trap point, but the trap points could be described as an average of adjoining vegetation plots. I measured vegetation cover, shrub stems, and CWD at these points before and after harvest.

I calculated total surrounding CWD for each trap location by summing CWD in adjacent vegetation survey plots and regressed capture rates for voles and mice against these values. A full description of the vegetation and CWD methods and results is found in Chapters 2, 3, and 4 of this thesis.

Results

Trapping resulted in 4,838 total captures of 1,794 individuals over 29,053 trap nights. I recorded at least a single capture of 15 mammalian, 6 amphibian, and 1 snake species. Of the 15 mammal species, only 2—deer mice (*Peromyscus* spp.) and red-backed voles (*Myodes gapperi*)—were caught in sufficient numbers for analysis with Mark-Recapture techniques. Three additional species—masked shrew (*Sorex cinereus*), northern short-tailed shrew (*Blarina brevicauda*), and eastern chipmunk (*Tamias striatus*) were captured frequently enough for statistical analysis using minimum numbers known alive at any one time. This community assemblage is common for habitats in this region, which are typically dominated by either *Peromyscus* spp. or *M. gapperi*, followed by lower numbers of shrews, squirrels, and other vole and mouse species (Erb and Rentz, unpublished Data, Timm 1975).

Overall capture numbers for amphibians were low both before and after harvest. Amphibian capture totals are presented in Table 5.1, but are not otherwise analyzed here.

Table 5.1 Total Captures*

Species	**Control**		**Slash-retention**		**Slash-removal**		
	Before	**After**	**Before**	**After**	**Before**	**After**	**Total**
Peromyscus spp	480	332	409	346	452	275	2294
Myodes gapperi	149	86	166	503	160	442	1506
Blarina brevicauda	37	24	80	5	58	11	215
Tamias striatus	70	43	99	33	127	3	375
Microtus pennsylvanicus	4	8	7	5	4	9	37
Sorex cinereus	31	44	51	4	57	12	199
Sorex hoyii	1	0	0	0	0	0	1
Sorex palustris	1	0	1	0	1	0	3
Sorex arcticus	1	2	0	2	1	2	8
Tamiasciurus hudsonicus	0	4	3	0	2	0	9
Poliocitellus franklinii	0	0	0	1	0	0	1
Zapus hudsonius	2	0	0	1	0	1	4
Mustela erminea	0	2	1	6	1	4	14
Glaucomys sabrinus	1	1	1	0	1	0	4
Synaptomys cooperi	2	0	0	1	0	0	3
Condylura cristata	0	0	1	0	3	0	4
Bufo americanus	9	22	7	2	15	5	60
Rana sylvatica	5	22	12	7	20	0	66
Ambystoma laterale	1	19	2	0	6	0	28
Pseudacris crucifer	0	0	1	0	0	0	1
Rana pipiens	0	0	0	1	0	0	1
Hyla versicolor	0	0	0	0	2	0	2
Thamnophis sirtalis	0	0	0	1	2	0	3
Total	**793**	**610**	**840**	**916**	**911**	**763**	**4838**

* Note these are total captures and are not corrected for variation in trap effort.

Pre-harvest comparisons:

Pre-treatment population sizes and mean biomass for the sum of all small mammals were similar across the three treatments on each site and for the sum of sites (Table 5.2). Only two treatments differed from other treatments within a site; the slash-retention treatment at Carlton had a higher mean biomass than either the slash-removal (t = -3.2, df = 7, p = 0.015) or control plots (t = -2.59, df = 7, p= 0.036) and the control treatment on CFC1 had a lower pre-harvest abundance (but not lower biomass) than the slash-retention treatment (t = -2.02, df = 7, p = 0.08).

Post-harvest findings:

Biomass and community population response.—In the two years following harvest, slash-retention plots had higher overall captures (all species summed, Tables 5.1 and 5.2), greater numbers and higher estimated survival rates of red-backed voles (tables 5.3 and 5.4), and higher values for both species diversity and evenness (Table 5.2) than either of the other treatments. Control plots showed little change in overall numbers for the small mammal community from pre-harvest conditions, with the exception of a decline in mice numbers on the Carlton site. The slash-removal sites varied in response, with a statistically significant increase in abundance at CFC1, a non-statistically significant decline at CFC3, and a significant decline at the Carlton location.

Mean biomass per session of the entire small mammal community was significantly higher in slash-retention (451 grams/session) than in either slash-removal

(385 grams; $t = -2.07$, df = 21, $p = 0.05$) or control treatments (239 grams; t = -4.37, df = 21, $p < 0.001$; Table 5.2, Figure 5.1). The retention plots also had higher average community abundance than either the removal ($t = -1.8$, df = 21, $p = 0.087$) or control plots ($t = -4.21$, df = 21, $p < 0.001$). Slash-removal treatments in turn had higher values than control plots for community abundance ($t = -3.33$, df = 21, $p = 0.003$) and biomass ($t = -3.29$, df = 21, $p = 0.003$; Table 5.2). This trend in community population size (with Retention > Removal > Control held for all three sites (though not by a statistically significant margin at each site individually) and was statistically significant for the sum of sites. Notably, the trend was present both in the sites with an overall increase in small mammal abundance (CFC1 and CFC3) and the site that experienced a sharp drop in overall small mammal abundance (Carlton).

On the two Cloquet Forestry Center sites population size and biomass means per session increased for the harvested treatments and decreased for the control plots (Table 5.2). Average calculated biomass for the two control treatments declined by a non-statistically significant 40 grams/session ($t = 0.677$, df = 30.92, $p = 0.5$) and increased by a non-statistically significant 130 grams/session on the sum of the two slash-removal treatments ($t = -1.53$, df = 30.53, $p = 0.13$). The slash-retention treatments saw an almost statistically significant increase of just under 200 grams/session ($t = -1.99$, df = 28.33, $p = 0.056$). All treatments at the Carlton site underwent a statistically significant drop in biomass and community population size due to a decline in *Peromyscus* abundance.

Table 5.2 Mean biomass (grams), Shannon Diversity, Species Richness (SR) and mean abundance (all species) before and after logging with slash-retention, logging with slash-removal, and no-harvest. Statistical significance of intra-treatment changes is noted, and designated as ns (not significant, p > 0.1), * (significant at p<0.1), ** (p<0.05), *** (p<0.01). Inter-treatment comparisons are noted where one or more treatments has a significantly stronger response (p <0.05) than the others. Values between treatments within a harvest status followed by a different letter are significantly different. Standard errors are given in parentheses.

Treatment	**Mean Biomass**[1]		**Mean Shannon Diversity**		**SR**[2]		**Mean Abundance**	
	Before	**After**	**Before**	**After**	**Before**	**After**	**Before**	**After**
All sites								
Control	378 (48) *a*	239 (35)** *a*	0.797 *a*	0.797 ns *a*	11	11	19.46 (2.3) *a*	13.1 (1.92) ** *a*
Retention	411 (59) *a*	451 (67) ns *b*	0.883 *a*	0.942 ns *b*	10	10	21.17 (2.89) *a*	25.88 (4.01) ns *b*
Removal	367 (51) *a*	385 (55) ns *c*	0.861 *a*	0.788 ns *a*	11	9	19.92 (2.56) *a*	22.15 (3.3) ns *c*

Carlton								
Control	527 (106) *a*	153 (32) ** *a*	0.425 *ab*	0.579 ns *a*	6	5	25.84 (5.18) *ab*	7.4 (1.33) *** *a*
Retention	604 (110) *b*	255 (57) ** *a*	0.638 *a*	0.996 ns *b*	7	7	29.2 (5.63) *a*	13.72 (3.3) ** *b*
Removal	494 (95) *a*	220 (67) ** *a*	0.317 *b*	0.869 ** *ab*	7	5	24.89 (4.88) *b*	12.21 (3.33) * *ab*
CFC1								
Control	348 (59) *a*	305 (63) ns *a*	0.985 *a*	0.703 * *a*	8	9	17.04 (2.6) *a*	16.64 (3.29) ns *a*
Retention	425 (66) *a*	701 (109) ** *b*	1.187 *ab*	0.963 ns *b*	8	4	22.19 (3.61) *b*	40.57 (6.74) ** *b*
Removal	369 (81) *a*	589 (86) * *b*	1.292 *b*	0.820 *** *ab*	10	8	20.22 (4.45) *ab*	33.82 (5.55) * *b*
CFC3								
Control	260 (52) *a*	218 (58) ns *a*	1.075 *a*	1.020 ns *a*	7	8	15.51 (3.1) *a*	12.69 (3.29) ns *a*
Retention	199 (76) a	292 (60) ns *b*	0.885 *a*	0.889 ns *b*	9	8	12.13 (4.05) *a*	16.95 (3.35) ns *b*
Removal	239 (72) a	258 (52) ns *ab*	1.11 *a*	0.709 ** *c*	4	7	14.67 (3.68) *a*	15.25 (2.97) ns *ab*

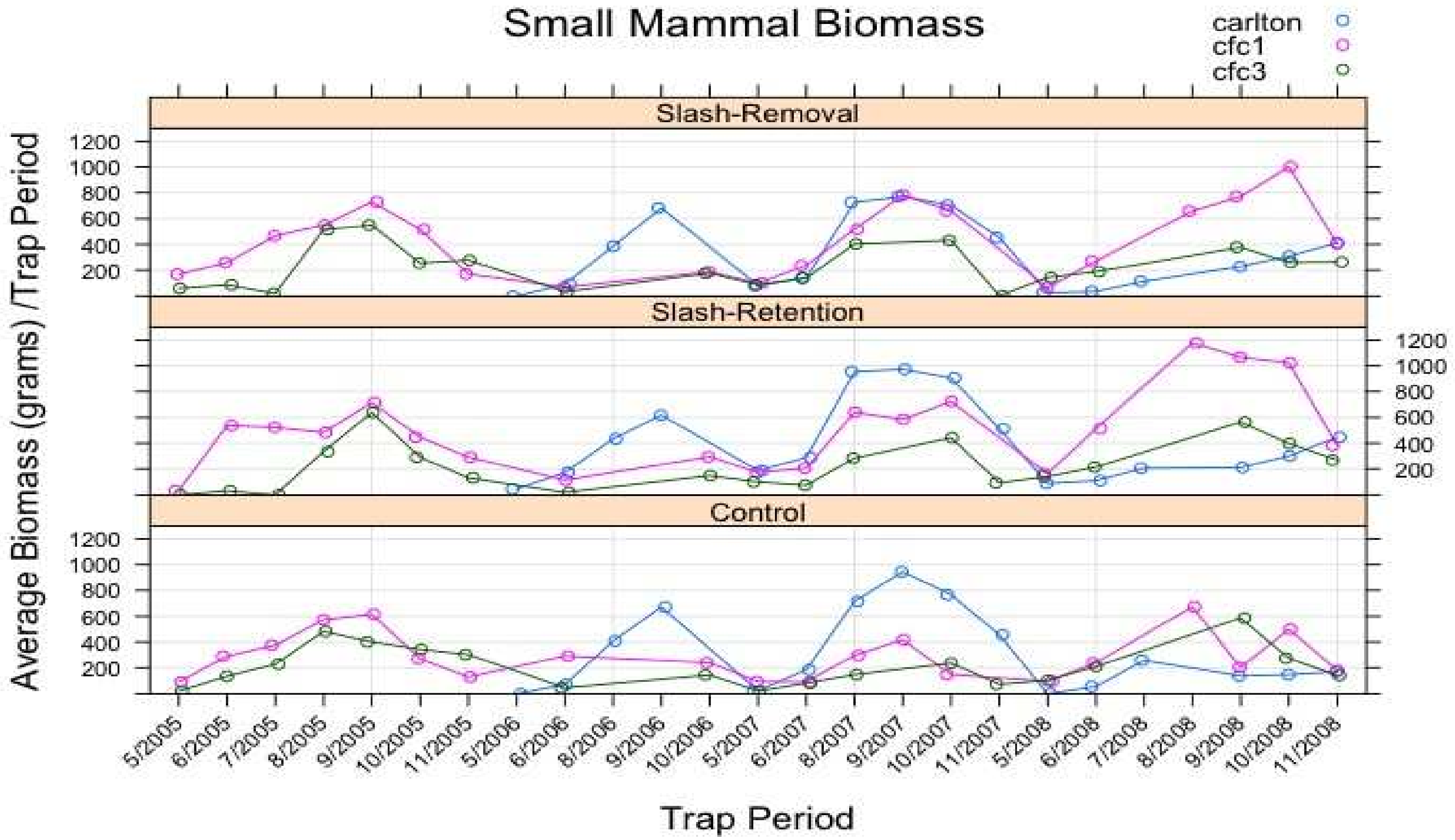
1 Not including sciurids
2 Species Richness: Includes all mammal species captured at least once.
3 Sum of population estimates using MR for *M. gapperi* and *Peromsycus* spp. and Minimum Numbers Known Alive for all other small mammals, excluding squirrels.

Table 5.3 Abundance (number per 0.5 ha trapping grid) and evenness of *Myodes gapperi* and *Peromsycus* spp. before and after harvest in paired treatments of clearcut with slash-retention, clearcut with slash-removal, and control. Average monthly abundance from M-R models (SE). Statistical significance of intra-treatment changes is noted, and designated as ns (not significant, p > 0.1), * (significant at p<0.1), ** (p<0.05), *** (p<0.01). Inter-treatment comparisons are noted where one or more treatments has a significantly stronger response (p <0.05) than the others. Values between treatments within a harvest status followed by a different letter are significantly different.

Treatment	***M. gapperi***		***Peromyscus* spp.**		**Species evenness**	
	Before	**After**	**Before**	**After**	**Before**	**After**
All sites						
Control	5.39 (1.05) *a*	1.91 (0.69) *** *a*	12.77 (2.84) *a*	8.56 (1.55) ns *a*	0.51 *a*	0.41 ns *a*
Retention	6.12 (1.17) *a*	15.6 (3.34) ** *b*	13.51 (3.45) *a*	7.79 (1.28) ns *a*	0.59 *a*	0.76 * *b*
Removal	5.72 (1.18) *a*	12.19 (2.43) ** *c*	13.38 (2.75) *a*	6.91 (1.09) ** *a*	0.54 *a*	0.59 ns *a*

Carlton						
Control	1.63 (0.78) *a*	0.8 (0.37) ns *a*	22.8 (5.2) *a*	4.64 (1.49) *** *a*	0.25 *ab*	0.38 ns *a*
Retention	2.38 (1.34) *a*	7.61 (2.56) ns *b*	25.95 (6.38) *a*	4.25 (1.44) ** *a*	0.4 *a*	0.88 *** *b*
Removal	1.13 (0.74) *a*	4.4 (1.72) ns *ab*	23.26 (4.99) *a*	5.46 (2.53) *** *a*	0.13 *b*	0.6 * *ab*
CFC1						
Control	4.35 (1.73) *a*	0.45 (0.44) * *a*	8.24 (2.11) *a*	13.47 (2.99) ns *ab*	0.72 *a*	0.18 *** *a*
Retention	11.64 (1.6) *b*	24.51 (6.81) * *b*	8.03 (1.49) *a*	12.19 (2.15) ns *a*	0.78 *a*	0.79 ns *b*
Removal	9.88 (1.82) *b*	22 (3.77) ** *b*	9.22 (1.56) *a*	7.42 (1.82) ns *b*	0.91 *a*	0.62 ** *b*
CFC3						
Control	10.02 (1.33) *a*	4.13 (1.59) ** *a*	3.94 (0.89) *a*	5.47 (0.98) ns *ab*	0.62 *a*	0.67 ns *a*
Retention	5.61 (1.26) *b*	10.59 (2.77) ns *b*	2.42 (0.72) *a*	5.06 (1.26) * *a*	0.63 *a*	0.69 ns *a*
Removal	7.68 (1.87) *b*	6.01 (1.72) ns *a*	4.37 (0.97) *a*	7.24 (1.72) ns *b*	0.66 *a*	0.54 ns *a*

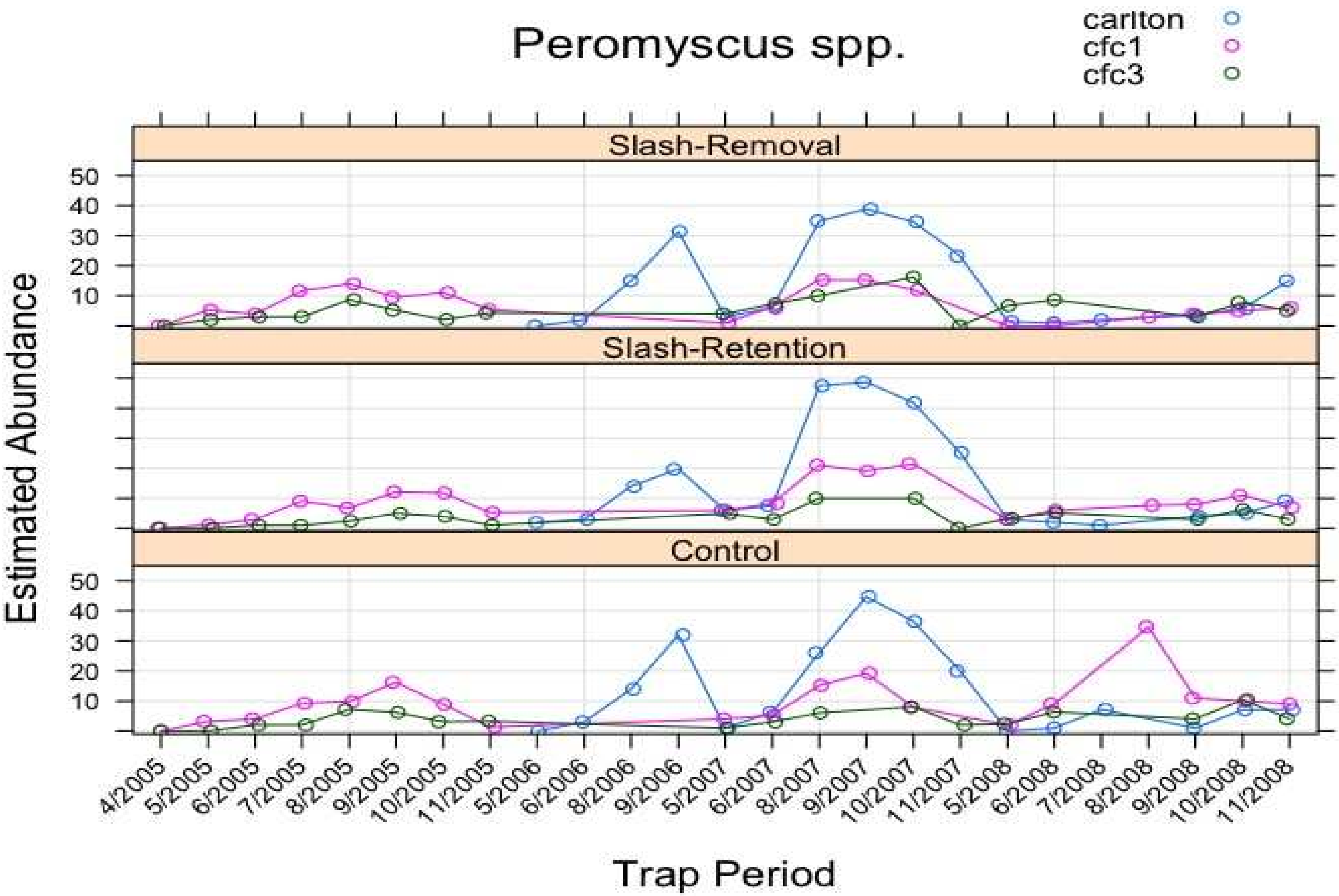
Figure 5.1 Small mammal biomass on trap sites over time.



Response of Peromyscus to harvest.—Harvest did not have a statistically significant effect on abundance estimates for *Peromyscus*. There was a notable decline in captures and abundance estimates of *Peromycus* spp. after harvest in all treatments on the Carlton site, but population sizes in all treatments remained largely steady at both locations on the Cloquet Forestry Station (Table 5.3, Figure 5.2). Declines on the Carlton site were similar and statistically significant across the three treatments. Mouse abundance decreased by non-statistically significant levels on the CFC1 slash-removal treatment (t = 0.75, df = 12.92, p = 0.466), increased by an amount that approached significance on the CFC3 slash-retention treatment (t = -1.83, df = 10.68, p = 0.096) and increased by non statistically significant amounts on the CFC3 control (t = 1.16, df = 11.94, p = 0.270), CFC3 slash-removal (t = 1.45, df = 10.63, p = 0.175) CFC1 slash-retention (t = 1.59, df = 12.81, p = 0.135) and CFC1 control (t = 1.43, df = 12.85, p = 0.176) treatments.

Differences in abundance estimates before-after harvest were not statistically significant for combined sites due to the diverging trends between the CFC and Carlton sites. Combining only the two CFC sites into a single analysis shows a growing population in the control treatment (before= 6.4, after 10.5; t = -1.819, df = 24.47, p-value = 0.08117), retention treatments (before= 5.1, after 9.4 t = -2.154, df = 26.295, p-value = 0.04057) and a stable population in the slash-removal (before 6.5, after 7.8; t = -0.87, df = 30.07, p-value = 0.3912). Pairwise comparisons between treatments were not statistically significant after harvest, indicating that the estimated population sizes were

Figure 5.2 Estimated abundance of *Peromsycus* spp. on trap sites over time.



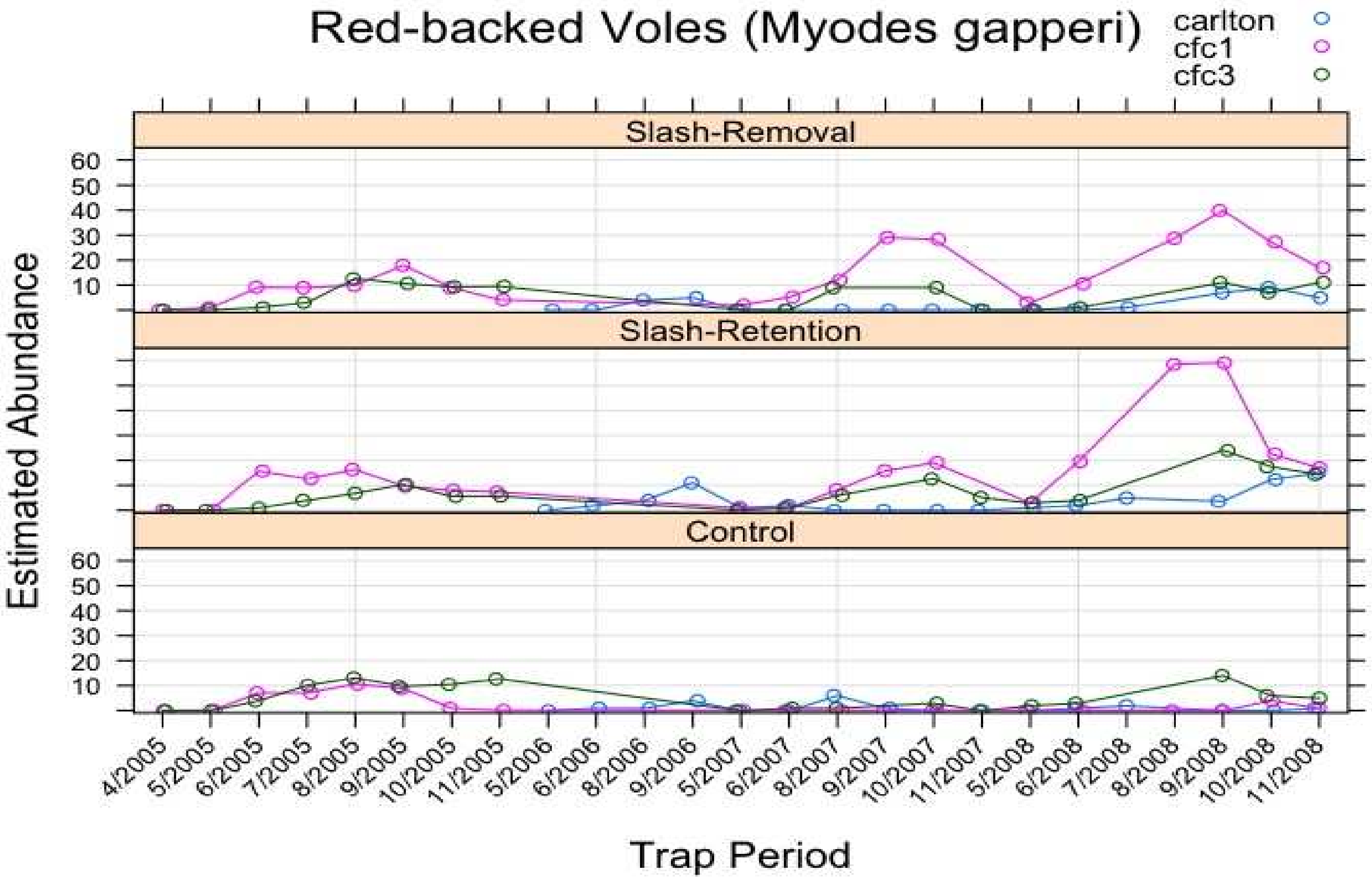
not different between either of the harvested treatments or between the control plots and the harvested plots.

Mean weights for mice were not different before versus after harvest for any site/treatment pairing, with the exception of mice in the control treatment on the Cloquet Forest Station Site #1, which were heavier prior to harvest than after harvest by an average of 2.6 grams ($t = 2.71$, df = 66.09 $p < 0.01$). This result is largely driven by a single female who was captured while pregnant in multiple months during a period of low population density.

Mean estimated survival rates for *Peromyscus* spp. were consistent between treatments and across treatment status within sites (Table 5.4). The only exception to this finding was a statistically significant increase in survival for mice on the CFC1 slash-retention treatment after harvest (22.2% ->49.3%, $t = -2.65$, df = 16.15, $p = 0.017$). Survival fluctuated across time in each site and treatment, but it did so in similar ways between treatments.

Response of red-backed voles to harvest.—Across the sum of sites estimated abundance of *M. gapperi* increased in both slash-retention and slash-removal treatments, while the species decreased in control plots (Figure 5.3). On an individual site basis vole numbers increased in all but one harvested treatment and decreased in all control treatments on the three sites. The changes were not statistically significant in every case (Table 5.3) but trends were similar across site and the changes in vole numbers were statistically significant for each treatment when all three sites were combined.

Figure 5.3 Estimated abundance of *Myodes gapperi* on trap sites over time.



Red-backed vole abundance estimates increased in harvested treatments and decreased in controls (all sites combined: slash-retention: t = -2.68, df = 26.05, p = 0.013; slash-removal: t = -2.39, df = 30.25, p = 0.02). Red-backed vole counts decreased on the control treatments (t = 2.77, df = 33.35, p = 0.009). The estimated abundance of red-backed voles was generally higher in the slash-retention than the slash-removal treatments, however the differences were not statistically significant when looking at data from both post-harvest years (t = 1.698, df = 21, p = 0.1). Using only 2008 data from the two CFC sites (two years post-harvest) a statistically significant difference did emerge in the estimated abundance of red-backed voles between slash-retention and slash-removal treatments (t = 2.63, df = 8, p = 0.03).

Mean weights of red-backed voles after harvest were not different from pre-harvest means for any treatment.

Red-backed voles show a difference in survival, but only on the CFC1 site (Table 5.4). Prior to harvest, estimated vole monthly survival averaged less than 30% on all three treatments. Post-harvest survival rose by a statistically significant amount on all three treatments, but the increase was of a significantly greater degree on the slash-retention (to 72%) and removal treatments (to 54%) than on the control treatment (to 38%) (t = -4.41, df = 10, p = 0.001; t = -2.31, df = 10, p = 0.043 respectively). The difference between the two harvest treatments was not statistically significant (t = 1.53, df = 10, p = 0.16). All other pairings of treatments within a harvest status and within treatments across harvest status were not statistically significant on the other two sites, though the strength of the change in the CFC1 retention treatment is strong enough to make the sum of the three sites also significant (t = -2.64, df = 48, p = 0.01).

Response of other species.—Only two species of shrew (*B. brevicauda* and *S. cinereus*) and eastern chipmunks (*T. striatus*) were captured in great enough numbers for trend analysis, and then only on the two CFC sites. Captures were sporadic and rare on the Carlton location, and numbers presented below are only for the two CFC locations.

Captures of *B. brevicauda* show an overall abundance decline across the years on the Cloquet Forestry Station sites. Declines in mean number of *B. brevicauda* known to be alive during any one trapping session were statistically significant for the slash-retention (from 3.7-0.3 mean animals/session, t = 2.32, df = 13.29, p = 0.037) and slash-removal treatments (from 2.9-0.6 mean animals/session, t = 1.98, df = 15.29, p = 0.066), but not the control treatments (from 2.0-1.0 mean animals/session, t = 1.01, df = 30.68, p = 0.32). No pairing of treatments showed a statistically significant difference in *B. brevicauda* MKA before harvest. Although the decline in short-tailed shrews was statistically significant for the harvested and not for the control treatments, there was not a statistically significant difference in abundance for the control versus either harvested treatment post-logging.

There were no detectable changes in mean weights for *B. brevicauda* after harvest for any treatment or site.

As with *B. brevicauda, S. cinereus* recorded a lower capture rate and lower minimum numbers known alive after harvest across all treatments. Numbers of *S.*

cinereus after harvest were statistically significantly lower than those prior to harvest for all three treatments. The two slash-retention plots lost an average of 2.2 shrews/session (from 3.4 animals before harvest to 1.2 after, *t* = 2.7, df = 18.69, *p* = 0.014), the slash-removal plots lost 1.8 animals/session (from 3.2-1.4 mean MKA/session, *t* = 2.21, df = 19.58, *p* = 0.039), and the control plots showed a decline of 1.8 animals/session (2.8-1.0 mean MKA/session, *t* = 2.31, df = 18.31, *p* = 0.033). Pairings of treatments both before and after harvest show no statistically significant differences in counts between the treatments.

Among sciurids only the Eastern chipmunk (*T. striatus*) was caught in large numbers. Chipmunks were common captures in the two Cloquet Forestry Sites prior to harvest, but rare after harvest. Comparisons of chipmunk MKA before and after harvest showed a statistically significant decrease for both the slash-retention treatments (a decrease of ~2 animals/trapping session (2.9/session before treatment, 0.8/session after); *t* = 2.82, df = 18.52, *p* = 0.01) and the slash-removal treatments (a decline of 3.2 animals/trapping session (3.4-0.2); *t* = 3.80, df = 15.39, *p* = 0.002). The decline in the control plots (from 1.9 to 1.1 animals/session) was not statistically significant (*t* = 1.26, df = 25.5, *p* = 0.22).

Prior to harvest, control plots had fewer chipmunks than either removal plots (mean of 1.5 animals/session difference, *t* = -1.85, df = 15, *p* = 0.08), or retention plots (mean of 1 animal/session difference, *t* = -1.79, df = 15, *p* = 0.09). After harvest this

Table 5.4: Survival of *M. gapperi* and *Peromyscus* spp. before and after logging. Statistical significance of intra-treatment changes is noted, and designated as ns (not significant, p > 0.1), * (significant at p<0.1), ** (p<0.05), *** (p<0.01). Inter-treatment comparisons are noted where one or more treatments has a significantly stronger response (p <0.05) than the others. Values between treatments across a year followed by a different letter are significantly different.

		M. gapperi		***Peromyscus*** **spp.**	
	Treatment	**Before**	**After**	**Before**	**After**
All sites	**Control**	0.274 *a*	0.312 ns *a*	0.43 *a*	0.503 ns *a*
	Retention	0.3 *a*	0.52 ** *b*	0.403 *a*	0.475 ns *a*
	Removal	0.289 *a*	0.392 ns *b*	0.436 *a*	0.5 ns *a*
Carlton	**Control**	0.321 *a*	0.341 ns *a*	0.493 *a*	0.597 ns *a*
	Retention	0.331 *a*	0.355 ns *a*	0.508 *a*	0.528 ns *a*
	Removal	0.368 *a*	0.317 ns *a*	0.375 *a*	0.373 ns *a*

CFC 1	**Control**	0.139 *a*	0.384 *** *a*	0.51 *a*	0.531 ns *a*
	Retention	0.291 *b*	0.724 *** *b*	0.222 *b*	0.493 ** *a*
	Removal	0.221 *ab*	0.541 *** *b*	0.452 *ab*	0.502 ns *a*
CFC 3	**Control**	0.367 *a*	0.215 ns *a*	0.247 *a*	0.416 ns *a*
	Retention	0.269 *a*	0.394 ns *a*	0.474 *a*	0.422 ns *a*
	Removal	0.264 *a*	0.274 ns *a*	0.495 *a*	0.576 ns *a*

was reversed, chipmunks were more numerous in control plots than slash-removal plots by a statistically significant margin of almost one animal/session ($t = 3.77$, df = 16, $p = 0.002$), and slash-retention plots by a non-significant 0.35 animals/session ($t = 1.3765$, df = 16, $p = 0.1876$). Prior to harvest there was not a difference in chipmunk abundance between the two harvest types ($t = 0.71$, df = 15, $p = 0.49$), though there was after harvest ($t = -2.58$, df = 16, $p = 0.02$).

Diversity.—Slash-retention sites post-harvest had higher overall Shannon Diversity Index scores, a more even small mammal community, and a more even balance between deer mice and red-backed voles than either of the other treatments (Tables 5.2 and 5.3). Overall, Shannon Diversity decreased across all treatments on the Cloquet Forestry Center, and increased on all three treatments at Carlton (Table 5.2). Estimates of Shannon's Diversity Index were statistically significantly different pre/post-harvest for the control and slash-removal treatments on the CFC1 site, the removal treatment on the CFC3 site and the two harvested treatments on the Carlton site. The retention treatment had the highest diversity value for the CFC1 and Carlton site, and the second highest value on the CFC3 site.

Discussion

I hypothesized that harvest would lead to an overall increase in small mammal total abundance, but a decline in diversity as a single species—expected to be *Peromyscus*—

increased and other species declined in proportion if not outright abundance. This prediction was somewhat borne out on the CFC sites, which saw slight declines in Shannon's Diversity Index scores. On these two sites harvest with slash-retention resulted in higher community abundance and biomass density post-harvest compared to either of the other treatments. Overall community diversity scores declined on five of the six CFC harvest treatments, though only significantly for the two removal plots, and the control plot at CFC1. Diversity declines in the control treatment at CFC1 are largely due to the background reduction in shrew numbers and the near-absence of voles in the post-harvest period. The similar reductions in shrews and chipmunks but a greater balance in vole and mice numbers in the slash-retention plots explains the difference in diversity values for the two harvested treatments at CFC.

The prediction that abundance and summed biomass of small mammals overall would increase on harvested areas was supported. Populations grew in estimated abundance in the harvested areas of the two CFC sites (and were statistically significant for CFC1) and retained more of their pre-harvest abundance levels at Carlton relative to the paired controls. On every site post-harvest population size and biomass was greatest on the slash-retention treatments, and lowest on the control. The trend appeared consistent, and adding more sites would likely have improved the chances of a statistically significant result.

Although I anticipated a short-term increase in overall community abundance, I had also hypothesized that *Peromyscus* numbers would increase, while those of *M. gapperi* would decrease. Previous studies have shown populations of deer and white-footed mice (*Peromyscus* spp*)* increase after logging while red-backed voles (*Myodes* spp.) generally decline in population following timber harvest, and decline in relation to the proportion of forest removed (Bowman et al. 2001; Craig et al. 2006; Le Blanc et al. 2010; Martell 1983; Moses and Boutin 2001; Pearce and Venier 2005; Sullivan and Sullivan, 2001; Sullivan et al., 2008; Sullivan et al., 2011; Zwolak 2009; but see Gunther et al. 1983 for a different relationship). On my plots red-backed voles increased in harvested plots following logging, and decreased in the paired controls. Setting aside the overall population crash on the Carlton site, mouse abundance increased slightly in both control and slash-retention treatments. *Peromsycus* abundance was actually lowest in the slash-removal treatment, in contrast to expectations and other research which has shown mice to either be unaffected by severity of disturbance or positively affected by it (Craig et al. 2006; Moses and Boutin 2001).

As the increase in voles was similar across all 3 sites I find it unlikely to be simply a result of local, yearly fluctuations or an error in methodology. The differing findings here may be a result of differences in habitat requirements across its range, or habitat differences between northeast Minnesota and the Rocky Mountains and Pacific Northwest where most previous work has been done.

There is some evidence that red-backed voles differ in habitat requirements across their range. The species is known to be associated with intact and older forests in the West. In more mesic habitats such as those in the Appalachians or boreal forests of the Great Lakes region red-backed voles may not be limited by clear-cutting as they are in the more xeric mountain west. My findings are similar to those in more eastern portions of the vole's range like Appalachia (Kaminski et al. 2007; Menzel et al. 1999) and Maine (Fuller et al. 2004). In Maine Fuller et al. (2004) found voles in multiple habitats, but did find reduced abundance in regenerating clearcuts. However, Kaminski et al. (2007) found voles were more abundant in disturbed versus undisturbed habitats. Minnesota voles seem to have niche more similar to the eastern portion of its range than the western; in prey surveys in northern Minnesota, both the Minnesota DNR and the 1854 Treaty Authority (an Inter-Tribal natural resources management agency which manages member Bands' access to ceded territory) have found red-backed voles in a wide variety of habitats (Erb, unpublished data; Wilson 2011). Red-backed voles seem to be limited in the west by cover and access to food and moisture (Pearce and Venier 2005). The rapid flush of vegetation and tree regeneration in the study plots here may have provided sufficient amounts of each for the voles.

I cannot rule out the possibility that the increased abundance found here is ephemeral, but the increasing strength of the trend in the second year after harvest paired with the avoidance of the adjacent control plots leads me to suspect that it is of a more lasting nature. I saw a difference in vole abundance between the harvested treatments begin to emerge as the study progressed. While the estimated abundance of red-backed voles is generally higher in the slash-retention than the slash-removal treatments, the difference was not a statistically significant one when 2007 and 2008 are considered together. Pairwise comparisons between slash-retention and slash-removal treatments were statistically significant when 2008 is considered on its own. I take this to be evidence of an emerging trend, and it is possible that the two treatments will continue to diverge.

Post-harvest counts of *Peromyscus* spp. do not differ significantly from pre-harvest counts for the CFC sites—though five of six treatments show small, non-statistically significant increases. There are also no statistically significant differences between the treatments within a site either before or after harvest. This result does not support the hypothesis that *Peromyscus* spp. numbers would increase post-harvest.

The stark and statistically significant drop in P*eromyscus* numbers on the Carlton site was likely evidence of the known cyclicity of these animals and unrelated to the harvest. Since this drop also occurred to the same magnitude in the adjoining, non-harvested control plots, it seemed clear that the harvest was not the cause of the decline, and any treatment effect was masked by the background cycle. The Carlton site was the largest of the harvested sites, and the control traps were over 100 meters from the nearest harvested edge, making edge effects of the harvest unlikely. There was not a similar decline noted on the Cloquet sites, arguing against a weather related cause. Given the very high population density in the fall before harvest, likely some density-dependent cause of mortality—possibly disease—was at work.

I did not find a clear trend for shrew response to harvest. Shrews—both *B. brevicauda* and *S. cinereus*—declined contrary to expectations in harvested treatments, but this decline was also present in the paired control plots leading to the absence of a statistically significant difference between the harvested treatments and the paired controls. Other work has shown common shrews—like the masked shrew (*Sorex* *cinereus*) and the northern short-tailed shrew (*Blarina brevicauda*), do not differ in a consistent way between clearcut forests and unharvested control plots (Sullivan et al. 1999; Zwolak 1999; but see Kirkland 1990 for support of shrew increases via logging).

It is possible that both species were experiencing normal population cycles, and that this trend may have been exacerbated by harvest in the cut treatments, at least for *B. brevicauda*. Fuller et al. (2004) finds similar decreases in shrew numbers on regenerating clearcuts (11-20 years post-harvest), which they attribute to lower densities of arthropods in response to decreased CWD. Kirkland (1990) reviewed 19 studies that included shrew responses to forestry, and reported that 13 showed a positive response of soricids to clearcutting.

Shrews may also be impacted by moisture levels, and avoid or are at a disadvantage in the drier conditions associated with clearcuts (Fuller et al. 2004). As the sites age, it will be worth continuing to monitor these species to see if differences begin to emerge as CWD decays out of the slash-removal system.

Similarly to *B. brevicauda, S. cinereus* shows a statistically significant decline in MKA after harvest in two of the three treatments, though here the treatment that does not show a statistically significant drop is the slash-removal treatment. As with the northern short-tailed shrew, the masked shrew has not been shown to respond either positively or negatively to logging in other studies.

Shrews of both species are mostly insectivorous, though the short-tailed shrew will consume other vertebrates (George et al. 1986). Although the insect community may change after logging—and was not studied here—it is reasonable to assume that both a mature forest and a recent clearcut would both house sufficient prey for shrews. Given that captures of these animals are infrequent, and background population cycles short, we should be cautious in inferring substantial meaning in the findings here. The species were not eradicated from any of the sites, and in light of their ubiquity in the region and high potential for population growth it does not appear that harvest of CWD will have a major impact on either local or landscape level abundance of these animals. This finding agrees with that of others who have also found shrews to be resilient to anthropogenic habitat disturbance (Sullivan et al. 1999; Zwolak 1999).

Although not predicted, chipmunk abundance declined on the harvested treatments to a greater degree than on paired controls, and also declined more on the slash-removal than slash-retention treatments. This finding is further supported by an apparent low density of chipmunks on harvested areas in the Superior National Forest, where surveys find few chipmunks and where slash is often piled or removed and CWD rare (M. Rentz and J. Erb, unpublished data).

I recorded few captures of rare species, with only a handful of bog lemmings (*Synaptomys cooperi)* and no captures of rock (*Microtus chrotorrhinus*) or heather (*Phenacomys ungava*) voles, nor any captures of smokey shrews (*Sorex fumeus*) and least weasel (*Mustela nivalis*). The heather vole is classified as a species of special concern in Minnesota, and is known from less than 20 records in the state (www.dnr.state.mn.us, last accessed 11-21-2013). The rock vole, while having no official status in Minnesota is tracked by the Minnesota Department of Natural Resources, and is known from only a few locations, with fewer than 100 captures between 1990 and 2006 (Minnesota

Department of Natural Resources 2006). The center of population for this vole species is Cook County—the tip of the "Arrowhead" and well north of this research—but within the scope of woody biomass harvest plans for the State (Christian, 1982a; Christian 1982b; Daniels 1982; Jannet 1990; Timm et al. 1977). There is not a great deal known about this animal in the state, though Jannet (1990) suggests that it may be adversely affected by clearcutting. Rock voles are also known to be associated with bunchberry (*Cornus racomosa*) and blueberries (*Vaccinium* spp.), notable because *Vaccinium* species were negatively affected by CWD harvest in the vegetation work accompanying this research (Rentz 2013). Land management that negatively affects these rarer species warrants caution, and further work targeted at these species is warranted.

Although differences between harvested plots were slight after a single year post-harvest, there are clear emerging differences in small mammal abundance and summed biomass by the second year, with slash-retention treatments outperforming the slash-removal treatments. While it is encouraging that overall numbers and biomass are higher post-harvest in the slash-removal treatments than both pre-existing levels and paired controls, the differences relative to slash-retention plots and the decline in captures of squirrels and chipmunks is worrisome. Each chipmunk weighs the equivalent of 5 mice or voles and the larger squirrels have an even higher ratio, providing predators with considerably more meat per capture. Other work has noted the importance of CWD for enabling mustelid predation (Andruwski et al. 2008). While harvesting CWD is unlikely to have severe impacts on overall abundance of small mammals and most common species, the lower density of biomass, the shift of the community to one with fewer sciurids, and the decrease in structural complexity of the forest floor may combine to reduce the suitability of the habitat for small carnivores such as weasels and martens.

The work also only investigates the immediate impacts of harvest. It is possible that the small differences immediately post-harvest may converge, or may continue to diverge. It is important to note that while the removal of CWD from removal plots was not complete, the remaining debris tended to be in more advanced decay classes. As this wood decomposes out of the system, it will not be replaced by harder logs in the system, and will not be supplemented with natural CWD until the regenerating trees begin to die. There will, therefore, be a period to come when CWD will be even less common in these stands. With this decline differences in abundance of saproxylic insects and fungi—important food sources for small mammals—between the treatments may be exacerbated.

Chapter 6

Live trap efficiencies for boreal forest small mammals: a comparison of Sherman, Ugglan, and Fitch-style traps.

Introduction:

Ecological research on small mammals, especially mark-recapture (M-R) studies, requires effective trapping methods to accurately assess both the species present and their relative abundance. Trapping methods that do not capture all species with equal success will yield inaccurate estimates of population size, and may even miss some less common species completely.

M-R studies are built on several assumptions, including the assumption that all species are equally catchable, that both sexes and both sub-adults and adult animals for a given species are equally catchable, and that initial capture does not affect an individual’s chances of subsequent capture (the animal does not become “trap happy” or “trap shy”). These assumptions may not be equally valid for all traps, and indeed comparisons of traps have shown that different models of traps do vary in performance (Anthony et al., 2005; Astúa et al., 2006; Chitty and Kempson, 1949; Dizney et al., 2008; Jacob et al., 2002; Lambert et al., 2005; Lambin and Mackinnon, 1997; McComb et al., 1991; O'Farrel et al., 1994; Rose et al., 1977; Slade et al., 1993; Whittaker et al., 1998; Wiener and Smith, 1972; Ylonen et al., 2003). Given differences in trap performance, some authors have suggested that a combination of traps be used to fully assess the small mammal community (Anthony et al. 2005, Dizney et al. 2008; Holdenried, 1954; McComb, et. al. 1991). I wanted to be sure to capture as representative sample of the community as I could, and at high enough rates for good data integrity. For these reasons I chose to use a combination of traps: Sherman, Ugglan, and Fitch-style traps.

No studies have simultaneously compared the Sherman, Ugglan, and Fitch-style traps, but these traps have been used in other comparison research. Results have varied considerably between traps and between different authors for the same traps. Studies in Australia (Jacob et al., 2002) and in Israel (Ylonen et al., 2003) found both Sherman and Longworth traps out performed Ugglan traps, while Lambin and MacKinnon (1997) found Longworth and Ugglan traps performed equally well. Anthony et al (2005) found differences between Longworth and Sherman traps indicating that these traps favored different species and may not be interchangeable. Morris (1968) found a greater number of species and individuals in Sherman traps than paired Longworth traps. Similarly, Astua et al (2006) also found that the Sherman trap was more efficient at capturing smaller mammals, especially rodents, than the paired Longworth traps in eastern Brazilian forests.

Unlike the traps above, which are of solid, opaque construction, the Fitch trap is constructed of open mesh. O'Farrel et al. (1994) found homemade and commercially available mesh traps outperformed the common folding Sherman trap, as did Rose et al (1977) and Strauss et al. (2008). Conversely Dizney et al. (2008) found these traps to perform very poorly in heavily vegetated areas in the Pacific Northwest.

In sum, it is probable that trap effectiveness varies with habitat and species assemblage, and possibly even within species in differing habitats. If this is the case, reliance on a single style of trap may yield misleading results. Researchers would be wise to either pilot different traps on site before beginning research, or employ a variety of trap designs.

Another consideration for researchers is whether to use single capture or multiple capture traps. Multiple capture traps may be especially appropriate for research on the interactions between individual animals, but it remains to be shown that researchers do not sacrifice overall capture rate when shifting to multi-capture traps. Belant and Windels (2007) found a decrease in overall capture rate in Tin Cat traps (an intentional multi-capture trap). Some species captured in Sherman traps were never captured in the paired Tin Cat traps, yielding different species diversity estimates. Similar concerns have been raised for Ugglan traps (Jacob et al. 2002; Ylonen et al 2003). Of the traps used in this study, the Sherman is a single capture trap, the Ugglan trap is intentionally multi-capture, and the Fitch trap, while not intentionally multi-capture, does not preclude additional animals from entering the trap once it is closed.

In this research I aimed to: 1) compare the three traps for overall effectiveness including both capture and failure rate, 2) compare the capture profiles of the three traps, and 3) determine whether a multi-capture design reduces overall trap effectiveness.

Methods

Sites.—I conducted this experiment from May-November 2006, 2007, 2008 on four sites: CFC1, CFC3, Carlton, and St. Louis.

Trapping.—I used a combination of Sherman folding traps (Figure 6.2), Fitch-style traps (Figure 6.3) and Ugglan Special #3 (Figures 6.4 and 6.5). Sherman traps (H. B. Sherman, Inc., Tallahassee, Florida) are a commonly used style in the United States and worldwide. Sherman traps are a box trap (single compartment with no nest box) with a treadle/door system (animals entering the trap step upon a depression plate releasing the door). The traps are available in a number of styles and sizes; the model used here (LFATDG) is 8 x 9 x 23cm. Fitch-style traps are a "do it yourself" style trap using shaped hard wire cloth to form a tube (Rose 1994). A tin can is inserted into one end as a nest box and the open end is fitted with a gravity door which is triggered as the animal passes a swinging treadle. I purchased these traps from Bob Rose, Old Dominion University. The traps measure 6.5 x 6.5 x 39 cm. Despite some evidence that they are an effective trap (Rose et al 1977, Nichols, et al. 1984, O'Farrel et al. 1994, Hayes, et al. 1996, Bob Rose, personal communication) they are not widely used. Ugglan traps are manufactured in Sweden by Grahnab and are widely used in Europe. I used the model Special #3 (7.5 x 6 x 24 cm) with the Grahnab supplied long roof cover. The design consists of a gravity trip plate leading to a nestbox, and is reset automatically after capture allowing for multiple captures.

I ran traps for four consecutive nights in May, June, August, September, October, and November on four sites in 2006 and 2007 and three sites in 2008[10]. I did not trap in July to avoid capturing animals in what is typically the hottest and most humid month. For

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[10] The site in St. Louis County was dropped from the research during the winter of 2007-2008 when the lease holder of the logging sale gave back the lease rather than cut harvest. As the larger study focused on the impacts of harvest, without the logging the larger question was moot and the site was dropped.

Figure 6.1 Photograph[11] of a Sherman trap. Note the wooden shingle to shade the trap and the micro-positioning of the trap adjacent to the CWD to maximize capture probabilities. Typically the wooden shingles overlapped the edges of the traps.

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[11] All photographs in this Chapter were taken by the author.

Figure 6.2 Photograph of a Fitch-style trap. Note the thin metal rod inserted through the trap to prevent the trap from flipping on its side.

Figure 6.3 Photograph of an Ugglan trap. Note the protective cover partillly slid back to show fit and the mesh trap underneath.

Figure 6.4 Photograph of a *Peromyscus* in an Ugglan trap. Note the consumed bait, the nest box, and the gravity plate through which the mouse entered the nest box.

Figure 6.5 Bait ball and polyfill bedding inside a Sherman trap. Hanging the contained bait from the back door decreased the chances of bait fouling the trapping mechanism and clearly showed evidence of an animal eating the bait.

Figure 6.6 Photograph of a pitfall trap, showing the last few meters of the fencing and the pit with elevated cover.

analysis I consider each four night run as its own primary period. I treated all traps equally, with common bait, bedding, and placement.

Each site was broken into three treatments as part of the larger research and each treatment therefore can be thought of as its own trapping site. Each of the three trapping sites on a larger study site were assumed to be independent of each other, but were trapped on the same nights. On each trapping site I laid out a 4 by 5 trapping grid of 20 trapping stations with 20 meter spacing. At each trap station two traps were placed independently within three meters of the trap center. I intentionally maximized capture probabilities by utilizing best available microhabitat within this three meter radius. The three types of trap were cycled between the stations in alternating pairs (Sherman/Ugglan; Sherman/Fitch; Fitch/Ugglan), with each type of trap missing from every third station. At the first and last trap station all three types were used, for a total of 14 traps of each type per grid, 42 traps total.

Traps were baited with “bait balls” consisting of cooked oatmeal and a chunk of raw potato (source of water) wrapped in waxed paper (Figure 6.5). Bait balls were hung from the back door of the trap in Sherman and Ugglan traps or placed in the nest box of the Fitch trap. I replaced bait balls when they had been consumed or if they became moldy. I chose bait balls over loose bait such as oats or bird seed to prevent gumming up of trap mechanisms, to ensure more consistency in bait volume and quality between traps, and to determine if bait had been disturbed or not. Examination of the bait ball allowed me to determine if an animal had entered the trap and consumed bait without tripping the trap. I added a small volume of polyfill batting (roughly the same volume as a mature mouse) in each trap for bedding, and covered traps with cedar shakes to prevent the metal traps from overheating. Traps were checked once daily in the morning.

Ideally, traps would have been checked twice daily or locked open during the day, however limited field personnel and the remoteness of some sites made this difficult. Traps were checked twice a day in 2005, and captures during the day were minimal (63 captures over 48 calendar nights of trapping). In light of the low rate of capture in 2005 and the greater remoteness of the sites added in 2006 (which made two trips a day prohibitive) I made the decision to drop the afternoon check and treat each of the sites alike. I did take care to minimize thermal stress on captured animals by shading traps under existing vegetation where possible and covering all traps with cedar shakes to further minimize solar gain on the metal traps. Chunks of raw potato were added to the bait balls as a source of water, and these were frequently chewed on by trapped animals. Finally, trapping periods were deliberately chosen to avoid the hottest periods of the year (late June through early August here), and during trapping season I would delay opening traps for a particular run if the forecast indicated unusual heat.

Upon checking, each trap was recorded as either: 1) capture (and species noted), 2) no action, or 3) a failure. Failures were identified as either disturbance to the bait indicating an animal entered the trap but was not caught nor was the trap sprung, a sprung trap with no capture and undisturbed bait, or an assumed escape (closed trap with disturbed bait but no animal). For analysis, a “trap night” or “uncorrected trap night” is one 24-hour period that a trap was open and baited, and is not corrected for closed but empty traps. For “corrected trap nights” I scored closed but empty traps as a half trap night (Beauvais and Buskirk, 1999), but scored other failures in which the trap remained open as a full night.

Marking and measuring of animals.—Upon initial capture animals were identified, weighed, sexed (rodents only), and marked. I marked rodents with a single ear tag (Monel #1005-1, National Band and Tag Company) and shrews (*Blarina brevicauda* and *Sorex* spp.*)* with a Visible Implant Elastomer tag (Northwest Marine Technology, Shaw Island, Washington). In response to frequent losses of ear tags in *M. gapperi* I began tagging this species with both ear and VIE tags in the third and fourth seasons (2007 and 2008). For all species I applied the VIE tags through injection of the elastomer in the underside of the tail with a 29 gauge insulin syringe. A small (<0.01CC) amount of the mixed elastomer was injected sub-cutaneously. On subsequent captures animals were identified to individuals and weighed. On occasion mice would escape prior to weighing. When missing, weights for mice were estimated from captures of the same individual within the same primary period.

These methods meet the ASM guidelines established for research with live animals (Sikes and Gannon 2011) and were reviewed and approved by the Institutional Animal Care and Use Committee at the University of Minnesota (Protocol Number: 0703A03590).

Analysis.—Trap success, failure, and species specific comparisons were analyzed using tests for equality of proportions. Comparisons in capture proportions of the most common animals were analyzed with Chi Square tests. Differences in body masses were tested through analysis of variation with a Tukey Honest Significant Difference correction. Differences in number of animals recaptured and average number of recaptures for each animal by trap were analyzed with an equality of proportions test and a *t* Test respectively. I used R (R 2.13.1 for Mac OS X GUI 1.40) for all statistical analysis.

Results

Over the course of the three years each trap type totaled roughly 7,700 uncorrected trap nights (Table 6.1). The traps differed considerably in trap success, with Fitch traps recording the lowest rate of capture (4.9%), and Ugglan the highest (26.7%. $X^2 = 1464.44$, *d.f.* = 2, $P < 0.01$) (see table 6.1). In total 12 species were captured by the traps, though deer mice (*Peromyscus* spp) and red-backed voles (*Myodes gapperi)* accounted for 93% of all captures. Four species—*Peromyscus* spp., *M. gapperi,* Eastern chipmunk (*Tamias striatus*), and northern short-tailed shrew (*B. brevicauda)*—were found with enough frequency to allow for statistical analysis. This low diversity of small mammals with a dominance of *M. gapperi* and *Peromyscus* is common in upland habitats in northern Minnesota (John Erb and Michael Rentz unpublished data; Wilson, 2011).

Overall capture.—Both Sherman and Ugglan traps caught more animals than Fitch style traps regardless of the time of year. Ugglan traps shows a significantly higher capture

Table 6.1.—Total trap nights, corrected trap nights, failures, fatalities, overall captures, and common species capture totals and captures/trap night for three small mammal live traps Fitch, Sherman, and Ugglan in northeastern Minnesota. Corrected trap nights are adjusted by assigning accidently tripped traps a 0.5 trap night, as per Beauvais and Buskirk (1999). Rates are calculated based upon total trap nights, not corrected nights. Differences between traps within columns are noted if $p < 0.01$ (d.f.1).

	Fitch	**Sherman**	**Ugglan**	**Total**
Trap Nights	7697	7700	7749	23216
Corrected T.N	7458	7422.5	7747.5	22628
Failures	835 (10.8% a)	943 (12.2%b)	119 (1.5% c)	1897 8.20%
Fatalities	28 (7.3%a)	57 (3.0%b)	80 (3.9%b)	165 (3.8%)
All Captures	380 (4.9%a)	1903 (24.7%c)	2072 (26.7%c)	4355 (18.8%)
Peromsycus sp.	233 (3% a)	1204 (16% b)	1372 (18% c)	2809 (12%)
M. gapperi	110 (1.4% a)	538 (7% b)	628 (8% c)	1276 (6%)
B. brevicauda	8 (<1% a)	32 (<1% b)	43 (<1% b)	83 <1%
T. striatus	16 (<1% a)	99 (1% b)	3 (<1% c)	118 (<1%)

rate than Sherman when based on uncorrected trap nights (26.7% versus 24.7%, X^2 = 8.17, *d.f.* = 1, $P < 0.01$). The difference is not statistically significant when the correction of assigning tripped but empty traps a value of a half trap night (Beauvais and Buskirk 1999) is used (26.7% for Ugglan and 25.6% for Sherman, X^2 = 2.34, *d.f.* = 1, $P = 0.13$).

The traps caught specific species at different rates as well. A greater percentage of Ugglan trap nights recorded a capture of *Peromyscus* spp. than either Sherman (X^2 = 11.9, *d.f.* = 1, P <0.001) or Fitch traps (X^2 = 894.1, *d.f.* = 1, P <0.01). Ugglan traps also recorded more captures of *M. gapperi* than Sherman (X^2 = 7.63, *d.f.* = 1, $P < 0.01$) traps or Fitch traps (X^2 = 387.5, *d.f.* = 1, $P < 0.01$). Sherman traps similarly caught more mice (X^2 = 722.79, *d.f.* = 1, $P < 0.01$) and voles (X^2 = 298.28, *d.f.* = 1, $P < 0.01$) than Fitch traps. Capture rates for *B. brevicauda* did not differ between Ugglan and Sherman traps (X^2 = 0.8306, *d.f.* = 1, $P = 0.36$) but both traps recorded this species at a greater rate than Fitch traps (Sherman: X^2 = 13.25, *d.f.* = 1, $P < 0.001$; Ugglan: X^2 = 20.75, *d.f.* = 1, P <0.001). Sherman were the most effective traps for capturing *T. striatus*, with a greater proportion of trap nights recording a chipmunk capture (n=99) than either Ugglan (n= 3; X^2 = 89.68, *d.f.* = 1, $P < 0.01$) or Fitch (n=16, X^2 = 7.6695, *d.f.* = 1, $P < 0.01$). Fitch traps out-caught Ugglan traps for chipmunks as well (X^2 = 58.88, *d.f.* = 1, $P < 0.01$).

Long-tailed shrew (*Sorex* spp) captures were low for all 3 traps, with no animals captured in Fitch traps, and 7 and 8 in Sherman and Ugglan respectively. This stands in contrast to 294 captures of *Sorex* species in nearby pitfall (Figure 6.6) traps run concurrently with the box traps.

Capture profile.—Each trap caught the three most common animals—*Peromyscus* spp, red-backed voles, and short-tailed shrews—in approximately the same proportions (Chi-squared= 0.69, *d.f.*=2, $P = 0.71$), though as noted above at rates that differed by a statistically significant degree. (Table 6.3). Voles (*M. gapperi and Mictotus pennsylvanicus*) and *B. brevicauda* make up the same proportion of all captures for each trap (voles= $X^2 = 2.63$, *d.f.* = 2, $P = 0.27$, *B. brevicauda*= ($X^2 = 0.62$, *d.f.* = 2, $P = 0.73$). *Peromyscus* spp. account for a greater fraction of all captures in Ugglan traps than Sherman traps (66.3% of all captures versus 63.3%, $X^2 = 3.77$, *d.f.* = 1, $P = 0.05$) but this difference disappears when squirrel captures—common in Sherman traps but rare in Ugglan traps—are removed (66.4% of all captures in Ugglan, 67.1% Sherman, $X^2 = 0.15$, *d.f.* = 1, $P = 0.70$).

Ugglan traps captured statistically significantly fewer chipmunks and squirrels than either Sherman or Fitch traps. Only three Eastern chipmunks and no red squirrels (*Tamiascurius hudsonicus*) were captured in Ugglan traps (0.1% of captures), while these two species made up roughly 5 percent of captures in both Fitch (19 captures, 5%) and Sherman traps (106 captures, 5.5%). The ratio of squirrels to all captures was not different between these traps ($X^2 = 0.104$, *d.f.* = 1, $P = 0.747$).

Fatalities.—Summed fatality rates for all species were below 10% for each trap, though Fitch traps had a significantly higher fatality rate than either Sherman ($X^2 = 15.70$, *d.f.* = 1, $p < 0.001$) or Ugglan traps ($X^2 = 8.56$, *d.f.* = 1, P = 0.003) (Table 6.1). Although Ugglan traps recorded more fatalities than Sherman traps, the difference was not statistically significant (X^2 = 1.98, *d.f.* = 1, P = 0.15) for all lumped captures. There was a significant difference between the two traps when only initial captures of *Peromyscus* are considered: 2% of first-caught animals died in Sherman traps, whereas just over 7% died in Ugglan traps (X^2 = 9.66, *d.f.* = 1, $P < 0.01$).

Multiple captures.—Of the 1,276 *Peromyscus* capture events in Ugglan traps 82 involved more than one mouse, including 12 instances of three individuals and two instances of four individuals. This multiple capture rate of 6.4% for Ugglan traps is significantly higher than the 1.3% rate for Fitch traps (X^2 = 8.66, *d.f.* = 1, P = 0.003), and 0.3% for Sherman (X^2 = 66.73, *d.f.* = 1, $P < 0.001$). Sherman traps had four captures of two individual *Peromyscus*, and Fitch traps had three captures of two *Peromyscus*. Another three *Peromyscus* in Ugglan traps were captured with—and presumably killed by—a *B. brevicauda*. No *Peromyscus* were caught with *Blarina* in either Fitch or Sherman traps, though one mouse was caught with a *Sorex cincerus* in a Sherman trap; in this instance the mouse survived, but the shrew did not.

Recaptures.—An analysis of initial trap capture and recaptures in *Peromyscus* does not show a relationship between the trap an animal was first caught in and its likelihood of subsequent recapture. The number of animals re-captured did not differ significantly with the trap in which the animal was first caught (X^2 = 0.92, *d.f.* = 2, P = 0.63), nor did the number of total re-capture events differ between the cohorts of animals based on trap of first capture (X^2 = 0.62, *d.f.* = 2, P = 0.73). The recapture rate of Sherman initial captures (1.88 recaptures/initially caught animal) was not significantly different than the rate for Ugglan (1.72 and $t = 0.95$, $d.f. = 715$, $P = 0.34$).

Trap fidelity.—The traps show some variation in trap loyalty (the likelihood that future captures will be in the original trap style). *Peromyscus* first caught in a Fitch trap were more likely to be caught in a Fitch trap again (24.7% of all recapture events) than those first caught in Sherman or Ugglan for whom the Fitch traps represented only 9.0% and 7.1% respectively of future recapture events. Alternatively, although *Peromyscus* first caught in a Fitch trap were more likely to be re-captured in a Fitch trap than those first caught in either of the other two traps, they also demonstrated the least trap fidelity, with 75.3% of subsequent captures occurring in Sherman and Ugglan rather than Fitch traps. Those *Peromyscus* first caught in either an Ugglan or a Sherman did not show such an avoidance of their first trap, and were as likely to have future captures in their original trap as the alternate (48.5% loyalty for Sherman and 51.2% loyalty for Ugglan). The loyalty scores for Sherman and Ugglan traps were not significantly different from each other ($X^2 = 0.90$, $d.f. = 1$, $P = 0.34$) or from chance, when Fitch traps are excluded (Ugglan: $X^2 = 0.40$, $d.f. = 1$, $P = 0.52$, Sherman: $X^2 = 0.50$, $d.f. = 1$, $P = 0.48$).

Sex ratios.—All three traps show a slight male population bias for *Peromyscus* when considering only first captures, ranging from 51 males /100 animals for Sherman to 64/100 for the Fitch (neither of which is different from 50:50 by a statistically significant degree). The Ugglan male bias (58/100) and overall bias (55/100) are both statistically significant when compared to parity (Ugglan: $X^2 = 9.22$, *d.f.* = 1, $P < 0.01$; all traps: $X^2 = 9.23$, *d.f.* = 1, $P < 0.01$).

Body mass.— Mean weights for captured mice and voles are in Table 6.3. There were statistically significant differences in mean weight between the traps for initial *Peromyscus* captures ($F_{2,792} = 6.90$, $P = 0.001$), all *Peromyscus* capture ($F_{2,2431} = 23.13$, $P < 0.001$), all *M. gapperi* captures ($F_{2,1037} = 3.52$, $P = 0.03$), but not for *M. gapperi* when only initial capture weights are considered ($F_{2,360} = 0.91$, $P = 0.4$). Mean weights of mice and voles were highest in Sherman traps for both animals in their first capture and for all captures. Differences in weight were never statistically significant between Fitch and Sherman traps, but were between Sherman and Ugglan traps for all *Peromyscus* captures ($P < 0.001$), *Peromyscus* on first capture ($P < 0.001$), and all *M. gapperi* captures ($P = 0.02$), but not for *M. gapperi* initial captures ($P = 0.51$). Weight differences between Ugglan and Fitch traps were only statistically significant for all *Peromyscus* captures ($P = 0.006$).

Failures.—All three traps failed on at least some trap nights, but the rates varied considerably between the trap types. Sherman traps failed at a rate of 12.2%, Fitch at 10.8% and Ugglan at 1.5%. Pairwise comparisons of difference between all of the traps were statistically significant ($P < 0.01$ for all iterations)

Table 6.2—Mean weights (in grams) and standard errors (in parenthesis) for 3 small mammals by trap type in northeastern Minnesota 2006-2008. Superscript indicates significance at a $p < 0.05$.

	Fitch	Sherman	Ugglan
Peromyscus spp.			
all captures	20.44 (0.29) a	20.74 (0.14) a	19.28 (0.12) b
initial captures	18.93 (0.48) c, d	19.2 (0.25) c	17.93 (0.20) d
M. gapperi			
all captures	17.65 (0.44) e, f	18.18 (0.19) e	17.44 (0.16) f
initial captures	16.52 (0.65) g	17.35 (0.29) g	16.85 (0.31) g
B. brevicauda	19.75 (0.65)h	20.23 (0.49) h	20.40 (0.48) h

Table 6.3—Capture profiles of three common traps evaluated in northeastern Minnesota. Data were collected 2006-2008. Capture totals and capture percentages (in parenthesis) for each species. Capture percentages are calculated as the fraction a given species represents of all captures for that trap. In removing squirrels from the analysis the ratio of the three most common species is not different between the traps overall, or in any pair-wise comparison (Chi-squared= 0.69, *d.f.*=2, $P = 0.71$). There was a statistically significant difference in proportion of capture attributable to squirrels between both Fitch and Ugglan ($X^2 = 79.76$, *d.f.* = 1, $P < 0.01$) and Sherman and Ugglan ($X^2 = 96.78$, *d.f.* = 1, $P < 0.01$), though not between Fitch and Sherman ($X^2 = 0.1$, *d.f.* = 1, $P = 0.747$)

	Peromyscus spp.	*M. gapperi*	*B. brevicauda*	*Sciuridae*
Fitch	233 (61.3)	110 (28.95)	8 (2.11)	19 (5.00)
Sherman	1205 (63.3)	543 (28.53)	32 (1.68)	106 (5.57)
Ugglan	1374 (66.3)	639 (30.84)	41 (1.98)	3 (0.14)

The nature of the failure varied by trap. The most common failure in Sherman traps was tripped but empty traps (n= 555, 60% of failures), the most common for Fitch traps was animals entering the trap and raiding the bait without triggering the trap (n=433, 53% of failures), while the two most common for Ugglan were animals escaping from a tripped trap and traps which were opened by another animal (n=47 (48%) and n=40 (41%), respectively).

Escapes or traps that were tripped but showed signs of the bait having been eaten were seen in all three traps, but again varied by trap; 14 percent of all capture events ended in escape for the Fitch trap, compared to 5% for Sherman and 2.5% for Ugglan. All pair-wise comparisons for escape rates were statistically significant ($p < 0.01$).

Discussion

The three trap types varied greatly in capture and failure rates, with Ugglan traps having the highest capture rate and lowest failure rate of the three traps. Sherman traps had a similarly high capture rate to Ugglan traps (24.7% versus 26.7%) but a much higher failure rate. All three traps caught the three most dominant species in the same ratios, leading to similar estimates of the diversity of the community. The species mix reported here is typical of forests in northeastern Minnesota (Erb and Rentz, unpublished Data, Timm 1975).

Use of the two high-success traps (Sherman and Ugglan) yielded similar capture rates for this community, though the Ugglan is simpler to use and will allow for multiple captures, as well as having a higher catch rate, and greater proportion of initial captures.

The Fitch trap had a similar capture profile, catching *Peromyscus* and *Myodes* at a similar ratio as the other traps, but at a far lower rate. Other researchers have had success with this trap, though often in prairie or meadow habitats, and targeting voles (O’Farrel et al. 1994; Rose et al. 1977). The traps do not seem to function well in forest environments with this species mix, similar to the findings of Dizney et al. (2008). The literature and the findings here seem to suggest that mesh traps such as the Fitch are appropriate for open grassland habitats and for some species, but are not effective in forests. Both Rose et al. (1977) and O’Farell et al. (1994) did capture *Peromyscus* spp. in their mesh traps at greater rates than in Sherman traps, Dizney et al. (2008) found results more similar to those here, where mice were caught less frequently in mesh traps than Shermans.

The species mix here differed from those in the Rose, O’Farrel, and Dizney papers, but *Peromsycus* species were present in all four studies. *Peromyscus* captures were higher in mesh than solid traps for O’Farrel et al. (1994) and Rose et al. (1977) but not for Dizney et al. (2008) leading to the assumption that habitat type—in the case of Dizney forests with closed canopies as opposed to the open prairie in the others—is the driving factor reducing the effectiveness of mesh traps. It would follow from this reasoning that something about the open nature of the trap is off putting to small mammals in forested conditions.

It is interesting to note that mice first caught in Fitch traps will re-enter the traps more often than animals first caught in either of the other two traps, but even they enter the trap less frequently than they are recaptured in the other traps.

Another potential factor for the lower than expected performance of the Fitch traps may lie in the bait and bedding used in the traps; previous tests of these and similar traps have not always treated the traps equally. Rose et al (1977) varied the bedding between the traps, using cotton in the Sherman traps and hay in the Fitch traps, and ascribing this difference as one of the potential reasons for the increase in capture rate of the Fitch over the Sherman traps. In addition, Rose et al. (1977) left the Fitch traps on site throughout the season, and cycled the Sherman traps through the various sites. While the ability to use loose natural bedding like hay may be an advantage to the Fitch trap, it is difficult to separate the effects of the bedding from the effect of the trap itself. While we would not expect animals to choose traps based upon bedding, the hay used in there would likely smell more natural than cotton or polyfill bedding, and may also have been viewed as a food source. It is important for study design to maintain uniformity between designs. Further, the Fitch traps may have benefited from remaining on site throughout the season, allowing the animals to become comfortable with the traps in the landscape.

As noted above, other studies have found Sherman traps to out-perform Ugglan traps (Jacob et al. 2002; Ylonen et al 2003). Jacob et al (2002) did not use the cover shield on Ugglan traps in their Australia work, and hypothesized that the open mesh may have been a deterrent to house *Peromyscus* entering the trap. Fitted sheet metal covers (purchased directly from Granhab) were always used in this study. If this cover improved the function of the Ugglan traps it is possible that similar covers could have improved the Fitch traps, though O'Farrel et al. (1994) found that covering a mesh trap lowered its capture rate.

One of the advantages of the Ugglan trap is its ability to catch multiple animals in 1 night, a useful feature for research questions on interactions between individual animals. The data presented here show that researchers will not sacrifice overall capture numbers in shifting to this trap from the more common Sherman. Importantly the Ugglan traps did not appear to suffer from the loss of species and lowered capture rate that Belant and Windels, (2007) experienced with Tin Cat traps. Additionally, the Ugglan trap is easier to use, and is more appropriate for use by less experienced researchers or when large numbers of people are used in field work, such as in citizen science or undergraduate field exercises.

Another factor in considering the Ugglan trap is its inability to capture squirrels. This may be a benefit in studies focusing on *Peromyscus* or other small mammals in which a squirrel capture would be a lost trap night for the targeted taxa, but is a distinct disadvantage for broader ecological investigations in which squirrels are a desired component of the capture.

Trap failures. —All traps will fail at some times, either through researcher error, chance, or a particularly clever or lucky animal. The traps here, however, show markedly different failure rates. Use of the bait ball allowed for more precise tracking of bait disturbance than loose bait, and clearly shows that this type of event should not be discounted. Animals entering the trap without triggering the trap accounted for 5.6% of trap nights in Fitch traps, and 3.5% in Sherman. Experienced researchers who take the time to check the treadle condition each day may see a lowering of failure rate, though it is worth noting that few published studies control for animals that enter the trap without setting it off as presented here. One exception is Holdenried (1954) who reported rates of Sherman traps “entered, not sprung” of 7% and “sprung without capture” of 10%. These values for this study are 3.5% and 7.2% of all trap nights respectively. Since traps failed on both ends of the spectrum—set too loose (closed with no capture) and too tight (open with evidence of bait raiding)—there does not appear to be a strong bias of consistent mis-setting.

Although not commonly reported, trap failure should be a consideration when choosing a trap. Sprung but empty traps and raided but not sprung traps represent lost capture opportunities, and can increase trap saturation, effectively lowering the proportion of a local population that can be captured on any given night. The failure rates for Sherman and Fitch traps were statistically significantly higher than that for Ugglan traps, meaning a greater proportion of Ugglan traps were “in service” on any given night. The 12.2% failure rate for Sherman traps found here is actually lower than that reported by Holdenried (1954) of 17 percent. It is worth noting that Ugglan traps outperform Sherman traps when based on simple nights in the field, but that the 2 traps score equally well when the trap night statistics are corrected by assigning failed traps a 0.5 trap night rather than a full trap night.

The Fitch and Sherman traps are not as simple to set as the Ugglan, and it is reasonable that with increasing experience the capture rate would improve and the failure rate (mis-traps) decline. This did not bear out in practice however; both Fitch and

Sherman traps had the lowest failure rate (mis-traps) in the first year of the study, with higher failure rates in subsequent seasons.

Fatalities.—Ugglan traps recorded a higher (though non-statistically significant) fatality rate than Sherman traps. This was especially true for *Peromyscus*. Captured *Peromyscus* would often attempt to chew their way through the mesh under the trap door and the mesh was sized such that their snouts became stuck in the mesh and the animals were almost invariably unable to back out. I found 35 animals in this situation and the animal was either dead or required euthanasia in 15 of these cases. Five *Peromyscus* were caught with injuries that suggested they had become so entangled but freed themselves, though 1 of these was found to have died anyway. In the remaining 20 cases the animal was freed and released after they were found to be in good enough health that survival was deemed likely. Six of these *Peromyscus* did survive and were caught subsequently. Only *Peromyscus* were found to get their snout stuck in the mesh, I never found a single individual of other species trapped in this way.

In conclusion, although Fitch, Sherman, and Ugglan traps all caught the same species in the same ratio, the capture rates varied greatly between the traps. From these results it seems clear that the Fitch trap is not an appropriate trap for northern hardwood forests of the upper Midwest. Ugglan and Sherman function very similarly when the failure rate of the Sherman trap is accounted for, though with its lower fail rate the Ugglan trap does out-perform the Sherman. The conclusion of previous authors—that researchers use a variety of traps in an attempt to catch a complete representation of the small mammal community and to avoid reliance on a trap that may not be the most effective for the species mix and habitat (Holdenried, 1954; McComb, et al. 1991)—seems valid, though not as important for low diversity sites such as these. Researchers do not sacrifice overall capture rate or incur a skewed species profile with the multi-capture Ugglan trap relative to the more common Sherman trap.

Chapter 7

Development of a novel marking technique for small mammals.

Introduction:

One important technique to assess the small mammal population on the study sites is Mark-Recapture (M-R) analysis. M-R studies help researchers shed light on community dynamics and demographics. M-R studies generate unique histories for each individual in a population; in turn these histories can be used to calculate capture probability, infer mortality likelihood, and make population and demographic estimates. These unique capture histories require effective, species appropriate marking of animals, and the assumptions that marks are not lost and can be correctly identified throughout the life of the animal (Cowen 2005; McDonald et al. 2003; Seber 1986; Williams et al. 2002). To be useful a marking system needs to remain detectible for the life of the study, allow for individual identification, and not impact the animal's overall health. If the marking leads to increased morbidity and infection, decreased ability to secure food, or increases an animal's likelihood of predation by increasing its visibility or limiting its physical capabilities, the inferences drawn from the data will be in error (McDonald et al. 2003; Seber 1986). If the mark is lost or mis-read, inferences will likewise be affected (Cowen 2005; McDonald et al. 2003; Seber 1986).

Multiple methods have been employed to mark mammals including leg bands, toe clipping, hot branding, freeze branding, ear tags (Rudran 1996), passive integrated responder (PIT) tags (Animal Care and Use Committee 1998), and even radioactive tags allowing animals to be tracked with Geiger-Muller counters (Godfrey 1954). Each marking technique has advantages and disadvantages, and each may violate the above assumptions in some way.

In recent years PIT tags have become increasingly popular, though their relative expense makes them a less than ideal option, especially for animals with low survival and recapture rates. Ear/flipper tags remain an easy and widely used method, applicable to a variety of animals from small mammals such as mice, to larger animals like bears (Diefenbach and Alt, 1998), and seals (Siniff and Ralls, 1991). These tags are especially useful for marking mammals with large pinnae. Marking of animals lacking sufficient pinnae, especially fossorial animals, is more problematic.

Considerable research has been done on the health impacts of marking systems. Although once widely practiced, toe clipping has lost support in recent years. Murray and Fuller (2000) caution that all types of mutilation marking in mammals should be avoided, though their review of the literature identifies numerous studies that show that toe clipping does not have significant effects (see also Pavone and Boonstra, 1985; Wood and Slade 1990). Perhaps most important to field researchers in mammalogy, the current guidelines of the American Society of Mammalogsists caution against its use (Sikes and Gannon 2011), noting it is specifically addressed in The National Research Council's "Guide For the Care and Use of Laboratory Animals." The NRC's "Guide" further requires the use of aseptic conditions, something difficult to maintain in the field (National Research Council, 2011). Mutilation tagging can also be prohibited by management agencies for species of conservation concern, such as the endangered Pacific pocket mouse (*Perognathus longimembris pacificus*) (Debra Shier, personal communication).

Loss of the mark is also a concern (Cowen 2005; Fokidis, et. al. 2006; Morley, 2002; Siniff and Ralls 1991; Wood and Slade 1990). Ear tags may snag on vegetation, in burrows, or may be lost via torn ear pinnae during fighting or grooming (Gannon and Sikes 2007). Some markings may be appropriate for some species, but less useful for others: Fokidis et al. (2006) studied tag loss in flying squirrels and murine rodents, and found that flying squirrels lost ear tags at a greater rate than PIT tags, while the opposite was found in the rodent species. Social factors may also affect tag loss, if individuals of a species interact with frequency—especially through allo-grooming, to which Fokidis et al. (2006) ascribe at least some of the blame for tag loss in the squirrels in their study.

As part of my work on small mammal response to CWD harvest, I initially chose ear tags (Monel #1005-1, National Band and Tag Company) to mark rodents, but they were not suitable for marking shrews (Soricidae) because of the animals' small size and reduced pinnae. I anticipated capturing masked (*Sorex cinerus)* and northern short-tailed shrews (*Blarina brevicauda*) and needed a method to determine whether an animal had been previously captured. Visible Implant Elastomers (VIE, Northwest Marine Technology, Shaw Island, Washington) presented a possible, more permanent alternative to fur clipping or painting of fur. These tags are a 2-part elastomer that cures into a pliable, non-bioactive florescent mark and have been used with success for herpetofauna (Bailey 2004, Davis and Ovaska 2001; Nauwelaerts et al. 2000), fish (Bonneau et al. 1995), and crustaceans (Jerry et at. 2001, Linnane and Mercer 1998). To date, no published study has used these tags in mammals. Northwest Marine Technology encouraged its use and provided assistance in securing permission for the tag to be used in this novel manner from the University of Minnesota's Institutional Animal Care and Use Committee (UMN-IACUC).

Ear tag loss in red-backed voles (*Myodes gapperi)* was a serious problem during the first two seasons of the research, prompting a desire for a back-up or alternative system. Anecdotal evidence (tags found in traps) suggested voles were removing tags through either self grooming or allogrooming. VIE tags were chosen as a back-up.

This research presents the first use of VIE tags for wild mammals. Objectives include determining: 1) whether the tagging technique can be applied by a single researcher in the field; 2) if the tags remain visible for at least as long as alternative tags; and 3) if the VIE tags aid in identifying animals who have lost tags, thereby allowing one continuous capture history, rather than two separate histories.

METHODS

Sites.— I used all four sites for this research: CFC1, CFC3, Carlton, and St. Louis. I trapped voles from May-November between 2005 and 2008, but did not begin using VIE tags until 2006. One site (St. Louis County) was dropped from the research during the winter of 2007-2008. Tagging in this manner began prior to logging, and continued after harvest.

Trapping.— Each of the four sites contained three independent 4 x 5 trap station grids with 20 meter spacing and an adjacent 4 bucket pitfall array. Live traps were a combination of three styles: Sherman (8 x 9 x 23cm, H.B. Sherman, Tallahassee, Florida); Ugglan (Special #3: 7.5 x 6 x 24cm. Grahnab, Sweden); and modified Fitch (6.5 x 6.5 x 39cm; Bob Rose, Old Dominion University). Live traps were laid out in alternating pairs, with each trap type missing from 6 of the 20 trap stations for 14 live traps of each type per grid. Each treatment also had a single 4 bucket pitfall array with 5 to 10 meters of aluminum flashing as fencing between the buckets. Traps were opened for 4 consecutive nights monthly from May through November (hereafter referred to as a primary period). Live traps were left in the field for the entire season and locked open when not in use. Pitfalls were locked closed with a tightfitting plywood lid between trapping periods.

Live traps were baited with bait-balls of cooked oatmeal and raw potato wrapped in waxed paper and hung from the back of the trap. Traps were also stocked with polyfill as a bedding medium. Live pitfalls were 35cm square and 33cm deep and constructed of durable plastic. Pitfalls were baited with whole oats and live bee moth larvae (*Galleria mellonella*), stocked with 10-20 cm of coarse wood sawdust and had an overhanging plywood cover suspended over the trap to provide shelter from sun, rain, and meso-carnivores. All traps were checked once daily in the morning. Lumped trap nights across all sites were roughly uniform across logged and unlogged sites, though more voles were captured in clearcut than forested sites.

Marking animals.—I began tagging shrews with the VIE system in May 2006, and started using it on voles in May 2007. During this period I captured and marked N = 365 individual voles (157 males, 181 females, 27 unsexed) with ear tags and VIE tags. On initial capture I weighed, sexed and marked each vole with a standard, uniquely numbered ear tag (Monel #1005-1) as well as a single VIE tag. Shrews were identified to species, weighed, and marked with VIE tags. I used three fluorescent colors to tag animals: orange, red, and blue. Use of three colors allowed for greater individualization of marks. For all species the VIE tags were injected in the underside of the tail with a 29 gauge insulin syringe. A small (<0.01CC) amount of the mixed elastomer was injected sub-cutaneously. Syringes were dipped in rubbing alcohol and allowed to dry after use to reduce transmission of disease or infection between animals, but needles were used continuously until the product became unusable or the needle dulled. All marking was done by a single researcher (see Figure 7.1).

On all subsequent captures, I identified the animal captured, weighed it, and documented the condition of the two marks. I inserted a new ear tag for animals that had lost them but retained the VIE tag. I re-individualized animals that were identified as losing an ear-tag at the end of the study by examining capture data to determine their original identification number using a combination of sex, site, grid location, weight, capture history, and VIE tag color. Parsimony was the ruling factor, and in cases where multiple animals fit the criteria, the most recently tagged animal was chosen. In this way

Figure 7.1 Single Researcher placing VIE tag

tag retention estimates for VIE are the minimum possible, and subsequent use of the date for population analysis through Program MARK would assume these shorter individual life histories. I did not remark animals that were missing a VIE tag. Voles that were tagged but not re-captured were dropped from analysis.

These methods meet the ASM guidelines established for research with live animals (Sikes and Gannon 2011) and were reviewed and approved by the Institutional Animal Care and Use Committee at the University of Minnesota (Protocol Number: 0703A03590).

Artificial illumination.—The VIE tags are designed to be visible in daylight conditions, but have the added feature of fluorescing under ultraviolet lights. This feature is intended for use in low light or night time surveys. Sunlight was typically sufficient to view the tags, and the artificial light source was rarely needed.

Analysis.—I assessed tag performance through a survival analysis (Klein and Moeschberger 2003), considering confirmation of tag loss as the event in question. For analysis, only animals re-captured at least once after tagging were considered. In cases where a vole received more than one ear tag as a result of loss, each ear tag was considered independently, and only included in the analysis if the animal was recaptured at least once after additional ear tags were placed. Exact dates of tagging and recapture were recorded for each animal, but the lack of data between primary trapping periods precludes this level of precision, as it was impossible to know when a tag was lost between trapping runs. Instead, I tracked tag survival within and across primary periods. A tag that was verified as lost within a primary period was assigned a tag survival length of 0, tags that survived to the next primary period were assigned a tag survival length of 1, increasing by 1 for each subsequent primary period.

I examined the effects of tag type (ear vs. VIE) and sex on tag loss rate using an equality of proportions test. I tested differential tag survival using an Asymptotic Logrank Test. I generated Kapplan-Meier plots to show tag survival/failure. All analyses were conducted in R (v. 2.13.1 for Mac OS X GUI 1.40).

RESULTS

Voles.—Over the course of trapping, 365 individual voles were captured and tagged with both methods; of this total, 237 (130 females, 91 males, 16 unsexed) were recaptured at least once, allowing for analysis. I applied a total of 310 ear tags to these 237 voles. Five voles were not recaptured after the final ear tag was applied, and this last tag is not included in the analysis.

Tag losses are presented in Table 7.1. Significantly more ear tags (27.7%) were lost compared to VIE tags (4.6%) (equality of proportions test: $X^2 = 47.31$, *d.f.* = 1, $P < 0.001$). Another 18 animals were captured with torn ears and no visible VIE tags; it is possible that these animals lost both tags, increasing VIE losses to 29 instances (12.2% loss rate). It is also possible that the ear was injured in another manner that resembled ear tag loss, and the animals were initial captures.

Survival analysis shows the VIE tags survived longer and were lost at a lower rate than the standard ear tag regardless of which level of VIE tag loss is assumed (n=11: $Z = 6.70$, $P < 0.01$; n=29: $Z = 4.41$, $P < 0.01$). There was not a statistically significant difference in ear tag loss between the sexes for either number of individuals to lose tags ($X^2 = 0.95$, *d.f.*= 1, $P = 0.33$) or for total number of lost tags ($X^2 = 0.46$, *d.f.*=1, $P = 0.50$).

Using minimum survival lengths—the number of days between the tag being placed and the last confirmation of the tag being visible—the 217 ear tags with calculable minimums survived an average of 15 days, versus 36 for the VIE tags. The longest surviving example for each tag was 468 days on the same animal. Six VIE tags lasted more than one year, versus only one for ear tags

Table 7.1.—Tag loss and retention for two small mammal tags—Visible Implant Elastomers (VIE) and standard ear tags—in red-backed voles (*Myodes gapperi*). A total of 365 animals were tagged at least once with each tag. Of these, 237 animals were recaptured at least once, allowing for analysis. The numbers below represent the 237 animals on which analysis was possible. Only confirmed tag losses (loss of one tag while retention of the other) are included below; an additional 18 animals were captured with torn ears and no VIE, indicating either the animals lost both tags or injured their ears in another manner.

Tags	**Visible Implant Elastomers**	**Ear**
Tags applied	239	305
Tags lost	11	85
No. voles that lost		
0 tags	228	166
1 tag	11	59
2 tags	0	11
3 tags	0	0
4 tags	0	1

Individual identification.—I was able to positively determine the individual identity of a vole with a lost ear tag in 47 instances. I applied the least-possible length parsimony rule and was able to assign individual identity in an additional 28 cases in which ear tags were lost. In a further eight cases minimum survival lengths of the VIE tag could be inferred, but it was not possible to identify a potential previous identity with sufficient confidence for use in a mark-recapture analysis. In five cases it was not possible to gather any inference about the animal's previous identity.

Weight loss.—On initial capture mean (standard deviation) weight of red-backed voles was 17.3 grams (±4.3). Since all animals received both tags, it is not possible to differentiate what impacts the two tags may have had on health of the animal independently, but weight changes for animals caught at least once during the same trapping period as initial VIE tagging were not statistically significant (Table 7.2). Mean weight difference of paired captures from initial tagging to capture the following day showed animals gained on average less than 0.1 grams (t = -0.74, *d.f.* = 147, *P* = 0.46). From initial marking to the second day animals lost on average 0.15 grams (t = 1.07, *d.f.* = 99, *P* = 0.29). By three days post tagging animals lost on average 0.4 grams (t = 1.33, *d.f.* = 39, *P* = 0.19). None of the differences in weight were statistically significant.

There were slightly more instances of day-day weight loss (n=111) than weight gain (n=99).

Table 7.2.—Weight changes in red-backed vole (*Myodes gapperi*) 1, 2, and 3 days post marking with both an ear tag (Monel #1) and Visible Implantable Elastomer. Number of individuals to gain, lose, or show no weight change, and mean weight change in grams. Mean differences in weight are not statistically significant (*P*>0.1).

	Day 1	**Day 2**	**Day 3**
Weight increase	50	30	15
Weight decrease	49	46	16
No Change	49	16	9
Mean	<0.1	-0.15	-0.4

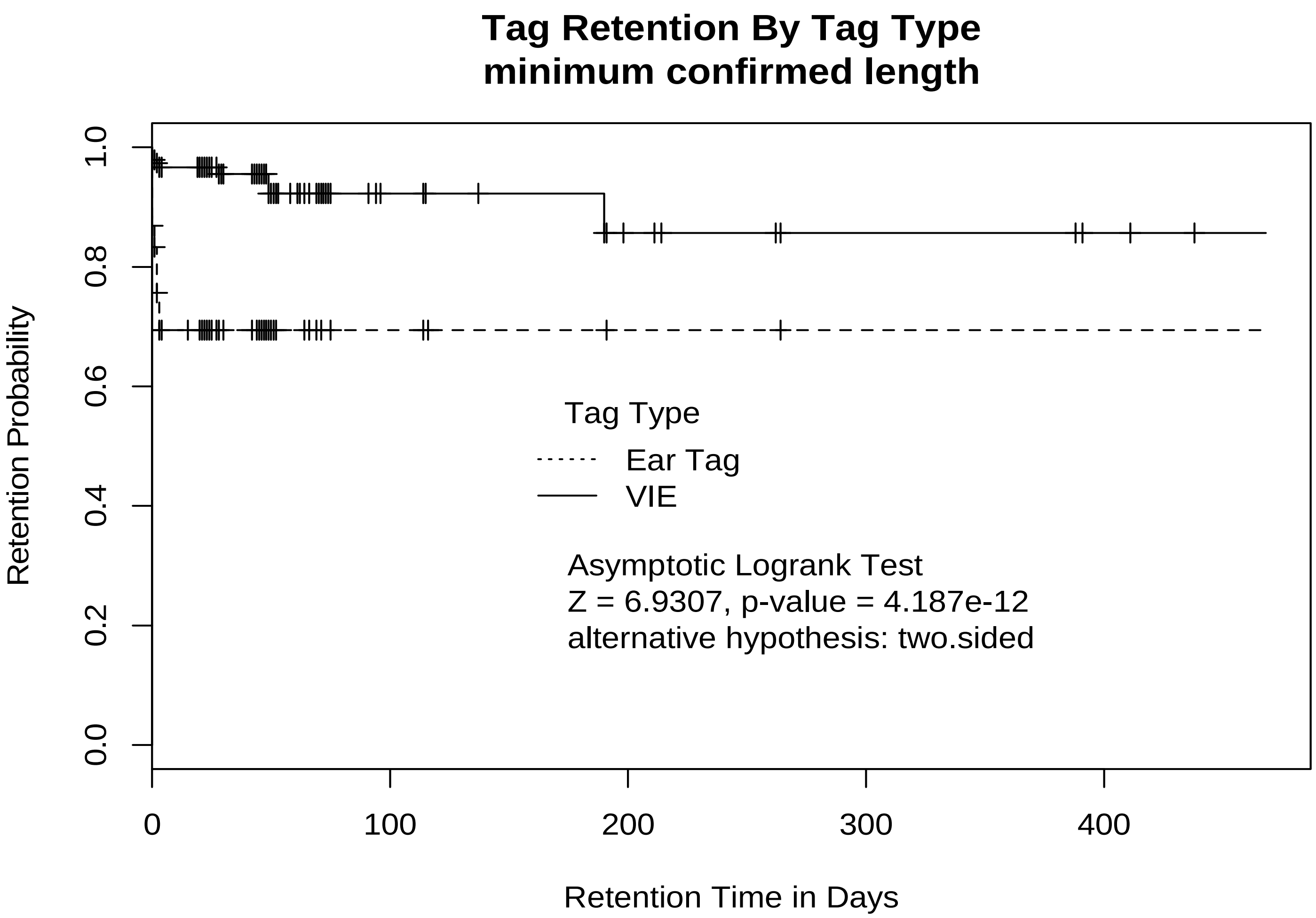
Shrews.—I marked 19 northern short-tailed shrews with VIE, of which five were recaptured (~25% recapture rate). I tagged 68 marked masked shrews with the VIE, of which ten were recaptured at least once (~15% recapture rate).

Tag movement.—Movement of the VIE tag within the animal has been identified as a problem in some taxa (Davis and Ovaska 2001). No tags appeared to change position during the study, although the field marking and limited personnel precluded precise marking and notation of mark location beyond general area of tail (base, mid-tail, or tip). Only one animal appeared to have shed the elastomer through its skin based on a visible scar. Movement of the tag completely out of the tail into the body or shed outside of the body would be recorded as lost tags, and it is reasonable to infer that tag shedding was the cause of tag loss in most cases.

DISCUSSION

VIE tags appear to be a safe, useful, and effective tagging technique for small mammals and show evidence to be a superior marking method for red backed voles when compared directly with ear tags, and an effective method for marking shrews. VIE tags lasted considerably longer than the ear tags in voles, and were lost less frequently. Figure 7.2 presents a Kaplan-Mier survival plot for the two tags, and shows the clear "survival" difference between the two systems.

Figure 7.2 Kaplan-Mier survival plot comparing retention and longevity (survival) of VIE and standard ear tags.



All ear tag loss occurred within the first primary period or between the first and second primary period, most in the first few days after initial marking. Ear tags that survived to the next primary period appear to be retained at a high level. By comparison, VIE tag losses were lower and spread out over time. Only 4.6% of VIE tags were lost, of which under half (5 out of 11) occurred during the initial primary period. VIE tags continued to be lost at a low rate across four months (i.e. until the 4th primary period).

Individual identification.—As noted in the introduction, individual identification is integral to M-R studies. Ear tag loss without the VIE back-up would have split an animal's capture history into a series of tag histories. As used here, the VIE tags themselves were not sufficient to identify individuals, however they were invaluable in generating these histories by tracking animals across ear tag losses and linking individual animals before and after tag loss events. Capture histories for these animals were only possible because of the extra marking, and without the VIE these animals would likely have been lost for mark-recapture analysis or would have diluted the quality of the data, and made inferences of animal survival and movement difficult.

Individual identification with VIE tags is possible, however; developing a system of alternating colors or number of marks would allow for greater individualization using the technique. Indeed, other researchers who have begun using the technique in endangered Pacific pocket mouse (*Perognathus longimembris pacificus*) did use a system to allow for individual marks (Debra Shier personal communication). Individual marking in the Pacific pocket mouse study required at least two personnel (Debra Shier personal communication), which was not possible for this research.

Artificial light.—The fluorescent tags used here are visible under normal light conditions, but are designed to fluoresce when exposed to ultraviolet light. This feature is especially useful for low light and dark conditions, such as night surveys. All animal handling in this study was outdoors during daylight hours, and the fluorescent feature of the tags was not required, but was chosen as a safety margin in the event that the skin obscured visualization. As expected, natural sunlight was sufficient for identification of the VIE tag, and use of the artificial UV source was discontinued after several months of use. In only 3 of the 480 observation were VIE tags recorded as lost on cloudy days, only to re-appear on subsequent sunny days, and it is possible that continuous use of the light may have lowered the reported tag loss rate, but in general the bulky light was not needed in this setting. The manufacturer notes: “VIE is available in six fluorescent and four non-fluorescent colors. The fluorescent colors are highly visible under ambient light and provide the option of greatly enhanced tag detection when fluoresced with the VI Light” (http://www.nmt.us/products/vie/vie.shtml, accessed 12-11-12). Non-fluorescent tags may also work in this application.

MANAGEMENT IMPLICATIONS

The intent of this endeavor—to verify tag loss and to re-individualize voles that had lost tags and to separate initial captures from re-captures in shrews—was realized.

Researchers who work with these or similar animals would be well served to use these tags, particularly as a back-up to another tag in species known to lose tags at a high rate.

Chapter 8

Conclusion

Forest managers across North America, including those in Minnesota, are considering harvest of logging residue (CWD) to develop a source of alternative fuel for electricity generation, provide another economic opportunity for the timber industry, or manage forests to lower fire risk. Such harvest is not without ecological consequences. To investigate some of these potential consequences I analyzed the impacts of woody biomass harvest on vegetation and small mammals in three forests in northeastern Minnesota. I divided each forest into three roughly equal parts: 1/3 of each site was clearcut with slash retained on site and no harvest of downed coarse woody debris (CWD) or snags (retention treatment); 1/3 was clearcut with whole-tree skidding, no dispersal of slash, and intentional harvest of snags and CWD ("removal treatment"); and 1/3 remained uncut to serve as an unharvested control. I surveyed CWD and the shrub, small mammal, and ground floor vegetation communities before and after harvest.

Harvest with and without slash-removal had a considerable impact on CWD. All plots averaged about 400 cm^2/m^2 of CWD prior to harvest. After harvest, the slash-retention plots averaged over 800 cm^2/m^2 while the slash-removal plots contained only 320 cm^2/m^2. The majority of the slash increase in the retention plots was newly dead wood created by the logging, while this type of wood decreased on the slash-removal plots due to harvest. Remaining CWD on both slash-retention and slash-removal harvest plots were on average shorter than the naturally occurring CWD of the pre-harvest condition. Shorter CWD may decay faster and be less useful as travel corridors for small mammals than natural CWD.

In chapter 1, I laid out several hypotheses about the likely impact of harvest on vegetation and small mammals. Here I summarize the findings in regard to these hypotheses.

Hypothesis 1: *Harvest following either of the two treatment protocols will increase overall small mammal community abundance, but lower the community diversity of the slash-removal plots relative to pre-harvest and control plots.*

This hypothesis was supported by the results, though not strongly on all sites. Community populations increased with harvest as predicted on CFC1, declined on all treatments on Carlton due to the crash in *Peromyscus* abundance, and increased by a non-statistically significant amount on CFC3. Diversity measures decreased by a statistically significant margin in two of the three slash-removal treatments, but not on the Carlton Site, where measures of diversity increased as the *Peromsycus* abundance declined to more even levels with that of red-backed voles.

Hypothesis 2: *Total small mammal abundance and diversity post-harvest will be higher in the slash-retention than the slash-removal treatment.*

This hypothesis was also weakly supported by the data. Indices of diversity were higher for slash-retention treatments than slash-removal treatments, though this difference was only statistically significant on CFC3. The gap in diversity values for the treatments began to widen by the second year.

The small mammal community responded similarly across the harvested treatments within the sites, though the magnitude of response differed. Generally captures of shrews and chipmunks declined post-harvest, estimates of *Peromyscus* remained more or less consistent with pre-harvest levels, and the abundance of red-backed voles increased by a statistically significant amount following clearcutting. Paired control sites also showed declines in shrews and chipmunks, and although the harvested sites generally exhibited steeper declines, post-harvest populations are not statistically different between the control and harvested plots for these species. Population estimates for *Peromyscus* also responded similarly within sites across treatments. This is not the case with the other dominant small mammal, the red-backed vole. Vole numbers increased post-harvest on all but one harvested treatment (voles declined the CFC3 slash-removal treatment by a non-statistically significant level) and declined on all control sites. Vole numbers increased more in the slash-retention treatment than the slash-removal treatments across the sum of sites, and the difference between the two treatments appears to be growing after two years.

Harvest had a generally positive effect on small mammal biomass and total abundance relative to paired controls, and on each of the three sites the slash-retention harvest had the highest mean biomass and community population size/sampling session. As was the case for the red-backed voles, the difference between the harvested treatments increased from year 1 to year 2 post-harvest.

Hypothesis 3: *Removal of logging slash and CWD will lead to a less diverse plant community and decline of more sensitive species.*

Species richness increased as non-native species and some native disturbance species spread to the sites. Shannon's Index of diversity remained steady on both control and harvested treatments. Some individual species such as members of the Liliales did decline in abundance. Few species, however, were completely extirpated from the harvest plots, and those that were had limited presence (<10 plots) prior to harvest. Other native species such as bigleaf aster (*E. macrophylla*) responded positively to the harvest.

Hypothesis 4: Removal of logging slash and CWD will open the forest to more invasion by non-native species.

I found good support for this hypothesis. Prior to harvest bare ground (exposed mineral soil) was rare in all plots, and where it did exist was a product of natural processes such as windfall of large trees or animal burrowing. After harvest exposed soil increased on both harvest treatments to cover 6% of slash-retention plots and almost 12% of slash-removal plots. The slash-retention plots recovered to average on 2.5% plot coverage by the second year, while on the slash-removal plots bare ground still accounted for nearly 10% of plot area.

Prevalence and richness of non-native plants increased in both harvested treatments. By two years after harvest non-native species richness between the two sites was similar, but these plants made up a larger percentage of the slash-removal plots then the slash-retention plots. This was especially true for the CFC1 site, where non-native plants covered 14% of slash-removal plots, twice the 7% rate of the adjacent slash-retention plots. No non-native species were counted in any control plots on any site. There is likely a causal relationship between the bare ground and non-native species; exposed mineral soil improves seed germination, both of desired and undesired species. Although exposing mineral soil is also done for the benefit naive tree species such as red oak (M. Cornett, personal communication, Eric Zenner, personal communication) the potential expansion of non-native species may be a negative consequence.

Prior to logging only two non-native species were found in the sites, a richness that increased to 15 species by the second summer after harvest. Some of the new species were found only on the margins of the stands (adjacent roadsides and ATV trails) while others were more widespread. There are several possible explanations for how non-species spread to the sites. Non-native species could have spread from adjacent existing seed sources, or could have been transported to sites on logging equipment. No requirement was made to clean equipment prior to entering these sites. Determining the origin of the non-native species was beyond the scope of this dissertation.

Harvest of both types resulted in a similar flush of overall plant biomass, as shown by the summed coverage of all taxa. As noted above, a desire to minimize research impact on the plots required only passive, visual observation of plots, rather than an active, destructive sampling that would have allowed for direct measures of total biomass through dried weight. Overall summed coverage averaged over 150% of pre-harvest coverage across both harvested sites, while remaining essentially constant on the control plots (a non-statistically significant reduction of ~20%). Coverage of native forb species declined across both harvest treatments in the year immediately following harvest, and remained below pre-harvest levels after two years. The reduction in native forb coverage was not significant for the slash-retention plots after two years, but remained below pre-harvest levels by a statistically significant margin on the slash-removal plots.

Much of the flush in vegetation across the harvested treatments was of grasses. Overall graminoids (grasses, sedges, and rushes) increased on slash-retention plots from an average of 12.5% plot coverage prior to harvest to 37% afterwards; and by a similar margin in slash-removal plots (14% to 36%). Grasses declined over the same period in the control plots. Although I was not able to identify the grasses to species, there are known invasive grasses in the region, and high grass density is known to inhibit tree growth. I observed the phenomena of a flush in graminoids following harvest across the region, and in some cases have noted that the grasses may persist in the understory for years or decades post-harvest.

Hypothesis 5: Whole tree skidding will inhibit regeneration of hazel.

This hypothesis was well supported by the data. Hazel densities were lower on the slash-removal treatment post-harvest (2.66 stems/m^2 after two years post-harvest) compared to the slash-retention treatment (4.87 stems/m^2). Hazel on the slash-removal treatments were predominately of the smallest size class, with nearly all shrubs more than

0.5m removed via harvest. Shrubs of this size fared better on the slash-retention treatments.

Hypothesis 6: There will be a noticeable drop in stem density for shrubs commonly found in the interior forest in the slash-removal but not the slash-retention plots.

This hypothesis found little support, with the exception of a decline in *Vaccinium* spp. (only a single stem post-harvest) on the slash-removal treatment. Although 17 genera of shrubs and vines were recorded in at least one plot, only a few species (such as *Acer* spp. and *D. lonicera*) were found in great enough numbers to infer any effects from the harvests. Conversely, shrubs less strongly associated with shade such as *D. lonicera* and *Salix* spp. increased in counts after harvest on both slash treatments, and by increased more in slash-removal than slash-retention plots.

Hypothesis 7: Rubus spp. stem density will increase under both harvest treatments, but by a greater degree on the slash-removal treatment.

This hypothesis was not supported; *Rubus* spp. responded similarly in the two cut treatments. *Rubus* stems increased from 0.93 stems/m^2 in slash-retention plots and 0.53 stems/m^2 to 2.71 and 2.95 stems/m^2 after two years respectively. Increases in stem density in each treatment were statistically significant relative to pre-harvest conditions, but were not different from each other.

Hypothesis 8: Retention of coarse woody debris will lead to lower densities of aspen suckers relative to the removal treatment.

Density of CWD inhibited aspen regeneration, but only at the highest densities observed, moderate levels of CWD did not appear to impact aspen sucker density in a meaningful way. In fact of the four plots with a hard CWD load greater than 900 cm^2/m^2 only two had aspen stem densities at levels below that considered adequate for successful regeneration, while four of the remaining 37 plots (average hard CWD density of 150 cm^2/m^2) also fell below this level.

Another commonality between the two harvests was the response of the overall shrub community, including aspen stem regeneration. Summed across all shrub species, total stem count nearly doubled from ~8 $stems/m^2$ on both harvested treatments to ~16 $stems/m^2$ after two years. Stem counts also grew in the adjacent control, but by a statistically significantly smaller margin. Increases in *Rubus* and *Populus* spp. were not statistically different between the two harvest types, but beaked hazel (*C. Cornuta*) was. Slash-retention plots averaged ~3.5 more $stems/m^2$ than slash-removal plots and just over four more than control plots.

In considering the sum of differences between the two harvest types, it is clear that the disturbance due to harvest itself plays a larger role in determining the trajectory of the stand than does the removal or retention of CWD. In comparing the two treatments, parameters such as increase in graminoid cover, fate of most shrubs, change in *Rubus* stem density, and native plant species richness, and aspen regeneration did not differ. However those metrics for which there was a difference deserve consideration. In all cases where there were statistically significant differences between the two treatments, the slash-removal treatment yielded a less favorable outcome for wildlife and native conditions than did the slash-retention treatment. This is especially true for the loss of native species cover, the increase in bare ground and non-native species cover, the loss of CWD, the decline in *C. cornuta,* and the emerging difference in small mammal biomass.

Of these differences, I am most concerned with the increased bare ground and non-native plant presence. Not all of the species encountered on my plots have been identified as posing an invasive threat, but some have, and others may emerge as problematic species. The case against invasive species is well known, and activities which exacerbate their spread should be avoided where feasible. The greater levels of exposed mineral soil created by the CWD removal could continue to provide an opportunity for non-native plants to become established. Although all of the non-native and potential invasive species identified in this study were forbs, there are invasive grasses, sedges, and rushes that could potentially invade sites such as these, as well as potentially invasive shrub species. Northern Minnesota is currently threatened by common buckthorn (*Rhamnus cathartica*), introduced honeysuckles (*Lonicera* spp.), and the shrub-like Japanese knotweed (*Fallopia japonica*) (personal observation, and Molly Thompson, personal communication). The absence of these species on my sites was likely due to a lack of a seed or other propagule source on the sites, and not an inherent resistance of the systems.

It is important to note that these are only immediate impacts one to two years post-harvest. It is not known if these differences will persist. The sites should be revisited in the future to better understand longer-term impacts.

As this study progressed the MFRC developed a set of voluntary guidelines for biomass harvesting, "Voluntary Site-Level Forest Management Guidelines for Landowners, Loggers and Resource Managers." The Guidelines call for the retention of pre-existing CWD and snags, and for the scattering of 20% of generated slash back onto the site (MFRC 2007). This study tested the two extremes of complete slash-removal or slash-retention. It did not test the ecological impacts of 20% retention. The impact of intermediate levels of slash-retention is left for future research. Since the Guidelines remain voluntary in most instances and may not be routinely followed, a test of the extremes is still valuable, since it is reasonable to assume that some harvests may come closer to the full slash removal here than the voluntary goal of 30% slash retention. If there are differences in key indicators between the two slash treatments, future research could compare intermediate levels of slash-retention to determine if the impacts increase in a linear fashion, or if there are tipping points or phase shifts. Further, the work here seems to indicate that it is the *act* of harvest, rather than the level of harvest that affects the forb community and opens the site to increased non-native species invasion.

The future of woody biomass harvest for energy remains in doubt, at least in Minnesota. Initial interest was high, especially among land managers, foresters, energy producers, and those who advocate for the timber industry (Arnosti et al. 2008). Despite numerous research reports, conferences, logger and land owner workshops this enthusiasm has not translated into reality on the ground. A 2012 survey of logging operations in Minnesota revealed that the number of firms that owned the equipment necessary to prepare the wood for shipment to a power plant had actually declined since the development of the biomass guidelines in Minnesota (MLEP 2012).

Whether woody biomass harvest plays a large role in Minnesota forests in the future will largely be determined by economic forces and distance of the forest to market. Currently, woody biomass harvest is only economical in conjunction with traditional harvest, and not as a stand-alone operation (D. Chura, personal communication). Additionally, the equipment needed to shred or grind the biomass into the form the powerplants desire and then deliver the product are expensive to purchase and costly to operate. Not a single operation reported to the Minnesota Loggers Education Program that they intended to purchase new equipment in the near future, and only five pieces of used equipment were being considered for purchase (MLEP 2012). The economic and transportation logistics meant that only about 19.5% of harvest volume in Minnesota was chipped or ground for any use (energy production or other) (MLEP 2012).

With these concerns noted, however, the ecology of the system does not depend on the logging residue being sold, only that it is removed from the forest. As I noted in the introduction, whole-tree skidding and central harvesting—the technique used in my research to attain the slash-removal treatments—is likely now a reality for how our forests will be managed. The slash piles created in this process may never be sold, but instead burned or left to rot in the landings where they were created. Whether it is removed from the forest, left in a single pile, piled in windrows, or burned, it is not available across the entire stand, as would be expected from natural mortality, my slash-retention treatment, or the intentional redistribution called for in many guidelines. Removing slash has the potential to lower small mammal community diversity, reduce the density of small mammal biomass, and negatively affect the abundance of some species relative to harvest with slash-retention. These differences are slight immediately post-harvest, but could increase with time.

Recommendations:

1. Woody biomass harvest should only involve generated logging slash such as tops and limbs from harvested trees, and not pre-existing snags and CWD.

In the research here, loggers were allowed to, and did, remove existing, naturally produced CWD and snags. I did not test the removal of generated slash with and without the concurrent removal of natural CWD, but given the importance of these features as noted previously in this thesis, and the increased harvesting effort needed to secure these, I recommend that only the generated slash be considered for harvest.

2. Forest managers should take care to reduce the introduction/expansion of non-native species, which may benefit from biomass harvest. Stands with adjacent sources of non-native species (such as roadways or adjoining stands) may be more vulnerable to increased colonization. Sites with no non-native species may benefit from cleaning logging equipment prior to harvest to remove a potential source of seeds and propagules.

Prior to harvest there were no identified non-native plants on the Carlton site, while there were on the CFC1 location. Post-harvest the increase in invasive species on the CFC1 was greater than that on the Carlton site by a statistically significant margin. Harvest of forests that are not already colonized by these species may be able to better able to resist invasion. The increased mechanical presence and subsequent erosion as well as the direct removal of CWD seems to lower the resistance of the system to expansion of invasive species, as demonstrated by the increased levels of these species on the removal versus the retention plots.

Even sites with assumed low-invasion risk may not be safe; the more remote Carlton site still experienced an increase in non-native species. While I am not aware of any studies that have shown invasive species spread to be inherent to harvesting of forests, I have found non-native plants in nearly every harvested stand I have visited in the region. If foregoing woody biomass harvest and retaining the slash on site boosts the resistance of the system to invasion, then it may be reasonable to proceed with caution on harvesting in all currently non-invaded forests, or at least forests of higher ecological value. In this scenario, currently invaded forests may become sacrifice zones where woody biomass harvest is allowed even knowing it may worsen the problem, while it is limited in non-invaded forests.

3. The current MFRC Voluntary Guidelines for Biomass Harvest (MFRC 2007) should be investigated to determine if they are beneficial.

As noted above, this study attempted to test the extremes—complete harvest of CWD and no harvest—and not intermediate harvest with some retention. It is possible that there are tipping points and that the system is resilient to some level of harvest. Alternatively if it is the increased machine use and whole-tree skidding that results in impacts then partial harvest may engender similar impacts to full harvest.

4. The sites here should be re-visited in the future to ascertain whether the differing impacts between the two harvest treatments were ephemeral or more longer-lasting. Future sampling may illustrate whether the two harvested treatments are progressing through ecological succession in similar ways, or whether the extra disturbance imposed by CWD harvest has shifted the succession trajectory of the slash-removal treatments.

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